Chapter 1. Introduction

1.1 Mala

Mala or the rufous hare-wallaby *Lagorchestes hirsutus* (literally 'hairy dancing-hare', Gould 1844) was first described from specimens collected from the Western Australian wheat belt (Lundie-Jenkins 1995). The species is part of the Superfamily Macropodoidea (members of this Superfamily are referred to as 'macropods' throughout the thesis), and belongs to the Family Macropodidae and Subfamily Macropodinae (Van Dyck and Strahan 2008). Three species of *Lagorchestes* were once extant on mainland Australia: *L. asomatus*, now extinct; *L. conspicillatus*, extinct in the southern part of its former range but still found across northern Australia; and *L. hirsutus* (Van Dyck and Strahan 2008). Four subspecies of *L. hirsutus* are currently recognised; *L. hirsutus hirsutus* (mainland south-western Australia), *L. hirsutus* subsp. (unnamed central mainland species), *L. hirsutus bernieri* (Bernier Island, Western Australia) and *L. hirsutus dorreae* (Dorre Island, Western Australia; Johnson and Burbidge 2008). However, a study of mala genetics has questioned this taxonomy, and may lead to future reclassification of the species (Johnson and Burbidge 2008). This thesis is concerned with the unnamed central mainland species, hitherto referred to as 'mala'.

Mala are a small wallaby standing around 300mm in height (Lundie-Jenkins 1995; Figure 1.1). Slight sexual dimorphism is present in the taxon, with females (900-1750 grams) being marginally larger than males (800-1600 grams; Johnson and Burbidge 2008:31). The species has thick, rufous fur, with the chest and abdomen being a paler, sandy buff in colour (Lundie-Jenkins 1995, pers. obs.). Hair length increases towards the lower back, giving the species a shaggy appearance (Lundie-Jenkins 1995, Figure 1.1).

In the central part of its former range, mala occurred in spinifex (*Triodia sp*.) dominated sandplain and dunefield habitat (Bolton and Latz 1978, Johnson and Burbidge 2008). The species is nocturnal, spending the daylight hours sheltering in a nest which

Figure 1.1 Mala *Lagorchestes hirsutus*. This animal is one of the 24 founders of the Uluṟu - Kata Tjuṯa National Park population, photographed early on the morning of its release

may take a variety of forms (Lundie-Jenkins 1993, Johnson and Burbidge 2008). At its most basic, the nest comprises a shallow scrape made beneath a spinifex clump, shrub or dead plant material (pers. obs., Johnson et al. 1996). However, mala may also excavate a short burrow (again under vegetative shelter), with a narrow entrance terminating in a broader chamber (pers obs, Johnson 1988, Figures 1.2 and 1.3). At Uluṟu, these

Figure 1.2 Entrance to mala nest amongst spinifex, UKTNP

Figure 1.3 The same nest as Figure 1.2, but with spinifex lifted showing narrow entrance and burrow beyond

burrows are up to approximately 40cm in length (pers. obs.). At dusk, mala emerge from their nests and travel to feeding areas (Lundie-Jenkins et al. 1993). These locations are generally open habitat adjacent to the mature spinifex communities in which they shelter, such as areas regenerating from fire or saline flats (Johnson and Burbidge 2008, Lundie-Jenkins et al. 1993). Mala browse a broad range of plants, of which grasses make up the bulk of the diet (Lundie-Jenkins et al. 1993). Mala do not require the presence of surface water to survive (Johnson 1988).

When moving slowly, such as during browsing behaviour, mala move pentapedally using all four limbs and the tail in a fashion similar to larger macropods (Lundie-Jenkins 1993, pers. obs., Figure 1.4). Faster movement is bipedal, with the tail held aloft (Lundie-Jenkins 1993, pers. obs., Figure 1.4).

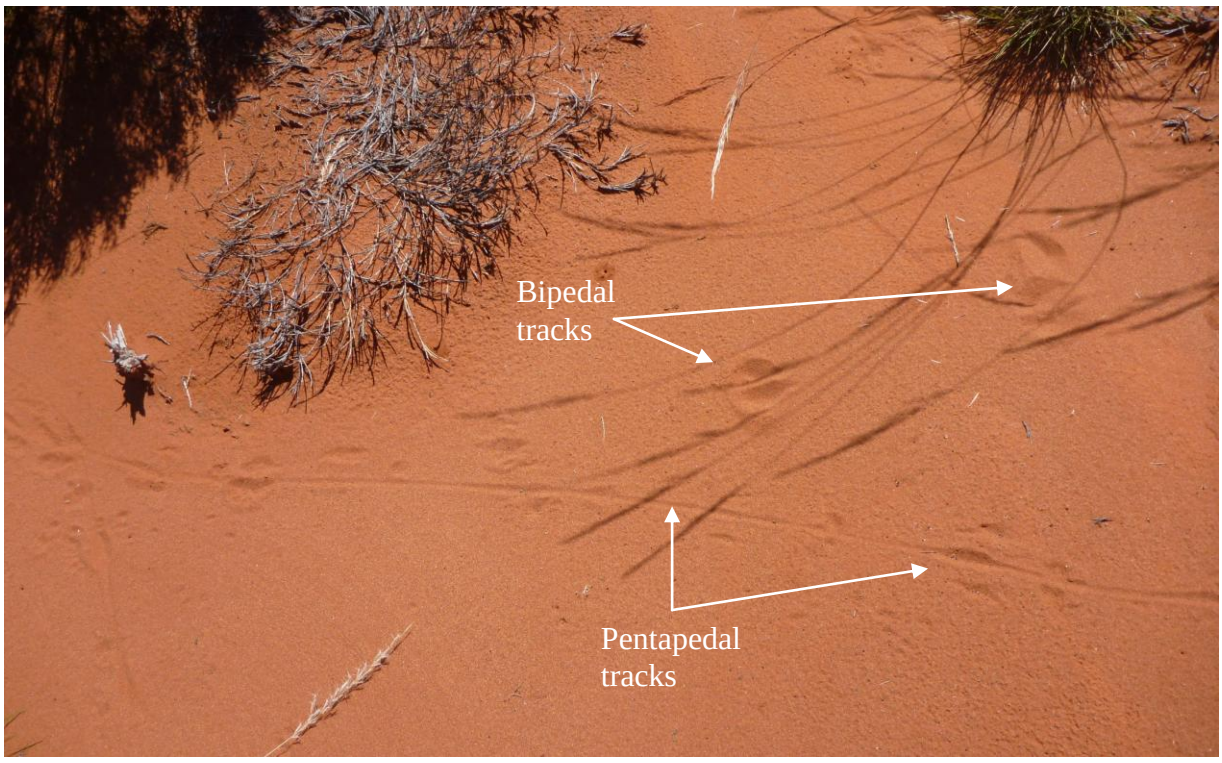

Figure 1.4 Pentapedal and bipedal mala tracks at UKTNP. A clear tail drag is visible when mala move slowly, however this is absent when the species moves more rapidly

If startled from their nests during the day, mala flee at high speed in zig-zag fashion, and often emit a short, high-pitched squeak (pers. obs., Johnson 1988). In captivity, mala

breed continuously throughout the year, and similarly no seasonality of breeding had been identified in a study of free-ranging animals (Lundie-Jenkins 1995). Males reach sexual maturity at around 14 months, and females between 5 and 18 months (Lundie-Jenkins 1995). Young remain in the pouch for about 124 days (Johnson and Burbidge 2008), and mala have been known to live as long as 13 years in captivity (Northern Territory Department of Natural Resources, Environment, The Arts and Sport unpublished data).

Mala were once found across a vast area of arid and semi-arid mainland Australia (Johnson and Burbidge 2008, Figure 1.5). However, the species began to decline in the

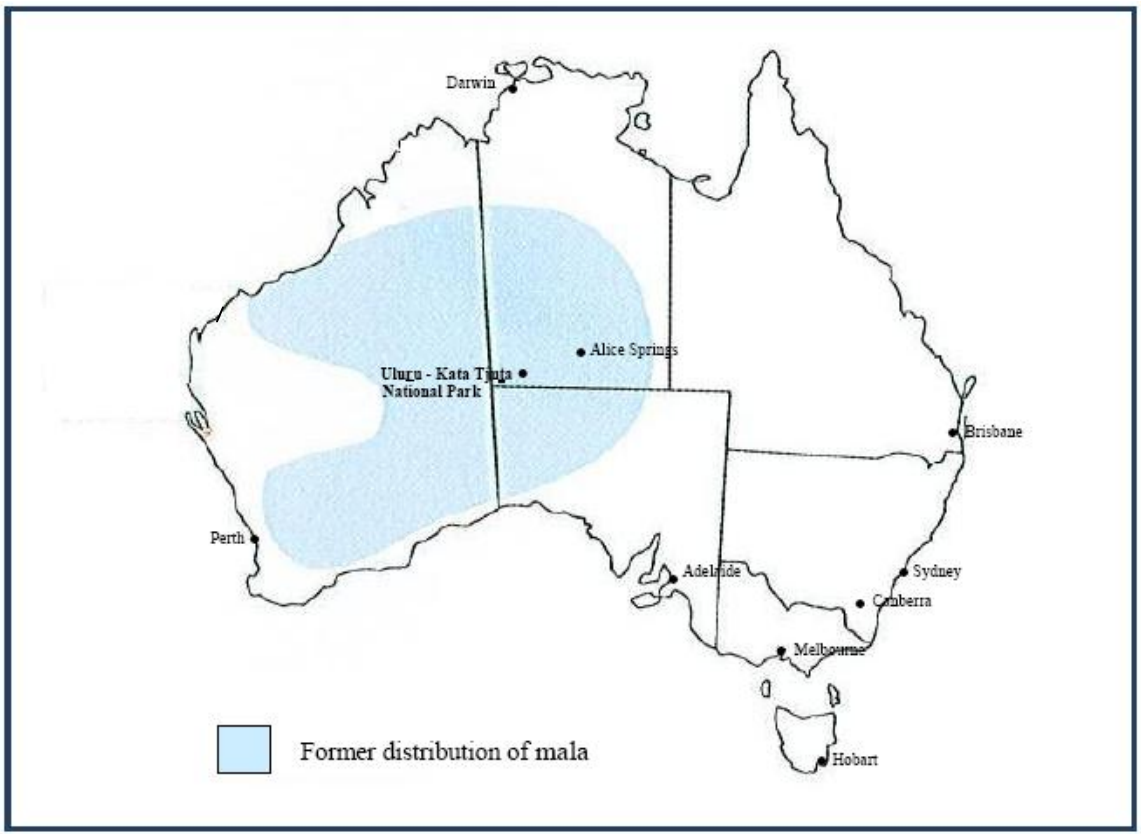

Figure 1.5 Former mainland distribution of mala (after Johnson and Burbidge 2008:318)

middle of last century, and is believed to have become extinct in the southern part of central Australia in the 1950s (Johnson and Burbidge 2008, Langford 2000). It has been proposed that the dramatic decline of mala was driven by habitat alteration (caused by

introduced herbivores and frequent, extensive bushfires) and/or direct predation by introduced carnivores (Bolton and Latz 1978, Burbidge and Pearson 1989, Johnson 1988, Richards 2005, Richards et al. 2008).

Surprisingly, mala did not become extinct on mainland Australia entirely, and two small populations of the species were discovered in 1959 near Sangsters Bore in the Tanami Desert (approximately 450km north-west of Alice Springs; Langford 1999a). These populations persisted until the late 1980s / early 1990s, when fire extirpated one colony, and the other succumbed to fox predation (Langford 2000). However, the establishment of a captive breeding colony prior to the extinction of the last wild mala population permitted the gradual recovery of the species (Gibson et al. 1994). This conservation work has been documented in detail in a series of scientific and general interest publications (Friend et al. 2002, Gibson et al. 1994, Hardman and Moro 2006a, Hardman and Moro 2006b, Johnson et al. 1996, Johnson 1988, Langford 2000, Langford 1999a, Langford 1999b, Langford 1993a, Langford 1993b, Langford and Burbidge 2001, Lundie-Jenkins 1998, Lundie-Jenkins 1996, Lundie-Jenkins 1995, Richards 2005, Figure 1.6).

1980	Conservation Commission of the Northern Territory (CCNT) staff removed animals from the last wild mala populations in the Tanami Desert to commence a captive breeding program at the Arid Zone Research Institute (AZRI) in Alice Springs.
1984-1985	Mala bred at AZRI were released into the wild at Lake Surprise in the Tanami Desert, Northern Territory. Post-release monitoring failed to locate any of the released animals.
1986-1989	CCNT and Willowra Aboriginal Community members built a one square kilometre enclosure at Lander River to protect reintroduced mala from predators. Mala bred at AZRI were released into the enclosure to create a breeding population. In 1988, a colony of mala was established at Western Plains Zoo (NSW) in order to support the captive breeding of the species. The program was discontinued in 2001.
1990-1992	Mala bred/held at the Lander River enclosure were released into the surrounding environment. Although individuals survived outside the enclosure for over a year, the reintroduction was ultimately unsuccessful.
1993	A national Mala Recovery Team was established to coordinate conservation action for the species.
1995	The Mala Recovery Plan was completed.
1998	In order to establish additional breeding colonies, 20 mala were translocated from the Lander River enclosure to the Dryandra breeding compound in WA.
1999	30 mala were translocated from the Lander River enclosure to Trimouille Island WA, where a population has successfully established. A workshop on faunal reintroductions was held at UKTNP, and Anangu voiced a desire to see a suite of species, including mala, returned to the Park. Revised Mala Recovery Plan for the period 1999-2004 was completed.
1992-1995	A colony of mala was established at Monarto Zoological Park (SA) in order to support the captive breeding of the species. The program was discontinued in 2004.
2000	The remote nature of the Lander River enclosure made fence maintenance and population monitoring logistically difficult. The Parks and Wildlife Commission of the Northern Territory (PWCNT, formerly CCNT) decided to decommission the Lander River site and build a new predator-proof enclosure at Watarrka National Park. Mala from Lander River were released into this 120 hectare enclosure in 2000.
2001	Protected from introduced predators by a fence across the isthmus, 16 captive bred mala bred on-site (animals originally sourced from Lander River enclosure) were released onto Francois Peron Peninsula. The fence ultimately proved ineffective and the reintroduction failed due to cat predation. Captive bred mala were released by the Australian Wildlife Conservancy into a predator-proof enclosure at Scotia Sanctuary in western NSW. The translocation has been deemed a short-term success.
2003	After meetings with agencies involved in mala recovery, Anangu and UKTNP decided that mala would be the priority species for reintroduction to Uluru.
2004	UKTNP staff and Mutitjulu Community members commence construction of a predator-proof enclosure at Uluru.
2005	Mala from Watarrka National Park were translocated to the newly completed 170 hectare enclosure at UKTNP.

Figure 1.6 Overview of mala recovery action 1980-2005

1.2 Uluṟu - Kata Tjuṯa National Park

Uluṟu – Kata Tjuṯa National Park (UKTNP, also referred to as 'Uluṟu' throughout this thesis) is an internationally recognized, World Heritage Area located 320 km south-west of Alice Springs (Figure 1.5). The Park, which covers 1325 km^2 within the historic range of mala, receives approximately 300,000 visitors annually from Australia and around the world (UKTNP unpublished data). The Uluṟu massif, and the domes of Kata Tjuṯa, are the central natural features of UKTNP (Figures 1.7 and 1.8). The Muṯitjulu Aboriginal Community is located within the Park, and is home to around 150-200 people. Aṉangu (Pitjantjatjara and Yankunytjatjara Aboriginal people) state that they have always been associated with Uluṟu, as they are descendants of the spirit beings which created the land (Uluṟu - Kata Tjuṯa Board of Management and Parks Australia 2000). *Tjukurpa* (Aṉangu Law and culture) explains the creation of the natural world, and dictates the relationships and responsibilities Aṉangu have to it (Uluṟu - Kata Tjuṯa Board of Management and Parks Australia 2000). Thus Aṉangu are inextricably linked to the Park spiritually. Historically, both Uluṟu and Kata Tjuṯa were important economic locations for Aṉangu (Layton 1989). Reliable water sources dictated traditional travelling routes, and

Figure 1.7 Uluṟu

available resources within the Park were sufficient to permit large gatherings of Aṉangu during favourable seasons (Layton 1989). Senior Aṉangu now residing at Muṯitjulu and further afield recall journeys to Uluṟu during traditional times as young children with their family group (pers. obs.).

After the Commonwealth Government amended the *National Parks and Wildlife Act 1975* and the *Aboriginal Land Rights (Northern Territory) 1976* in 1985, the legislative

Figure 1.8 Kata Tjuṯa

framework was in place to permit granting of inalienable freehold title to the Park to Aṉangu (Uluṟu - Kata Tjuṯa Board of Management and Parks Australia 2000). As part of this undertaking, Aṉangu agreed to lease the land back to the Commonwealth to be run as a national park (Uluṟu - Kata Tjuṯa Board of Management and Parks Australia 2000). A Board of Management, with a majority of Aṉangu traditional owners, was formed, and continues to manage UKTNP to ensure the protection of its natural and cultural values (Uluṟu - Kata Tjuṯa Board of Management and Director of National Parks 2010).

1.2.1 UKTNP natural environment

UKTNP is situated within the Amadeus Basin geological region (English 1998a, Sweet and Crick 1992). The Uluṟu and Kata Tjuṯa inselbergs, rising around 340 metres and 500 metres above the surrounding plain respectively, are the Park's most significant topographic features (Uluṟu - Kata Tjuṯa Board of Management and Parks Australia 2000). About 550 million years ago, the erosion of ancient mountains to the west of what is now UKTNP provided the material that would form the Uluṟu arkose and Kata Tjuṯa conglomerate (Sweet and Crick 1992). Although Uluṟu and Kata Tjuṯa visually dominate the landscape, dunefields and sandplains cover the majority of the Park (English 1998a, Johnson and Burbidge 2008, Reid et al 1993, Uluṟu - Kata Tjuṯa Board of Management and Parks Australia 2000). The dunes are believed to have started forming around 30,000

years ago, and reach heights of up to 13 metres (Sweet and Crick 1992). Two patterns are present: long, curvilinear dunes; and more randomly oriented 'chicken wire' configurations (English 1998b:265). In addition to these sand deposits, clay-rich red earths are found in both outwash areas associated with Uluru and Kata Tjuta and dune swales across the greater Park (English 1998a, English 1998b).

Consistent floristic patterns are found within the Park's sand dune and sandplain habitat (Buckley 1981). Vegetation is sparse on dune crests, and thick stands of thryptomene (*Aluta maisoneuveii*) occur on the midslopes (Buckley 1981, English 1998b). Lower dune slopes are dominated by spinifex (*Triodia sp*.), whilst the clayey sands of the outer swales support tussock grasses (Buckley 1981). Dense stands of mulga (*Acacia aneura*) are found on the heavier soils of the dune swales (Buckley 1981, English 1998b). The relatively higher soil moisture and microhabitat associated with the Uluru massif supports greater plant diversity than the surrounding sandplain and dune environments (Kerle 1995). Perennial grasses and sedges are found at semi-permanent waterholes, and water courses are lined with bloodwoods (*Corymbia opaca*; Kerle 1995). Kata Tjuta, where the complexity of the domes provides a wide variety of habitat types, is yet more floristically diverse than Uluru (Kerle 1995).

A total of 21 native and five introduced mammal species are found at UKTNP. These include three species listed under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act): the recently reintroduced mala (endangered), the southern marsupial mole (*Notoryctes typhlops*, endangered), and the mulgara (*Dasycercus blythi*, threatened; Commonwealth of Australia 2009a). Further, 178 bird species have been recorded within the Park (Commonwealth of Australia 2011). With 77 species present, UKTNP supports a richer diversity of reptiles per unit area than any other location within the Australian arid/ semi-arid zone (Reid et al. 1993). Included is the great desert skink (*Egernia kintorei*), which is listed as 'vulnerable' under the EPBC Act (Department of Environment and Heritage 2006).

1.2.2 UKTNP climate

Although the long-term annual average is approximately 290mm, rainfall at UKTNP is highly unpredictable (Director of National Parks and Uluṟu – Kata Tjuṯa Board of Management 2010, Reid et. al. 1993). Further, rain events may be very localized (Reid et al. 1993, pers. obs.). Significant rainfall can occur at any time of year, however the majority of heavy falls occur in the warmer months and are associated with monsoonal weather patterns to the north (Reid et al. 1993). Regular historic rainfall data for UKTNP is only available from 1965 to 1982 (Reid et al. 1993). However, Yulara Airport (located approximately 20km north of the Uluṟu massif) has recorded rainfall data from 1985-87 and 1995 to the present (Bureau of Meteorology 2011). Although complete records are not available, the existing data clearly show the irregular pattern of rainfall at UKTNP (Figure 1.9).

Unlike rainfall, temperature at UKTNP is relatively predictable (Reid et al. 1993). The summer months experiences hot days and mild to warm nights, with an average of 43 days above 40 degrees Celsius between October and April (Bureau of Meteorology 2011, Commonwealth of Australia 2011, Figure 1.10). Typically, cool to mild days and cold nights occur in winter, and frosts are not unusual (Bureau of Meteorology 2011, Reid et al. 1993, Figure 1.10).

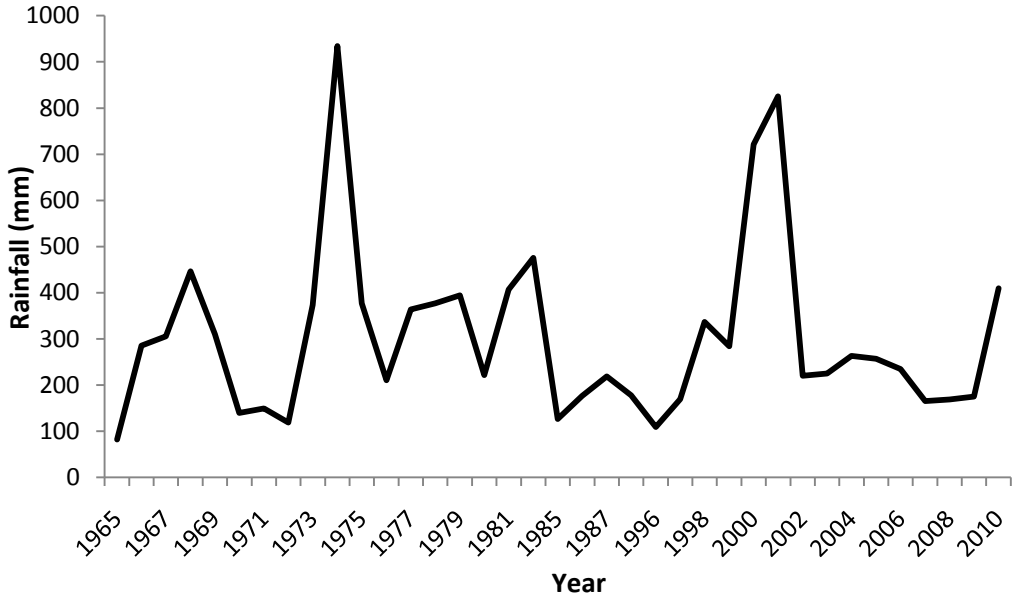

Figure 1.9 Available annual rainfall records (mm) for UKTNP (1965-1982; Reid et al. 1993) and Yulara Airport (1985-2010; Bureau of Meteorology 2011)

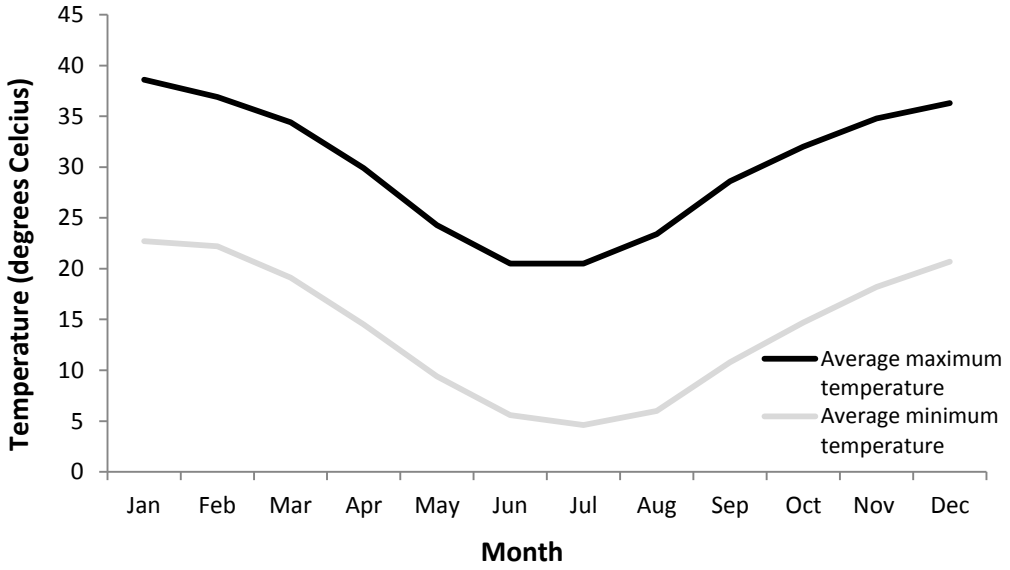

Figure 1.10 Monthly average maximum and minimum temperatures recorded at Yulara Airport 1985-2010

1.3 Importance of mala to traditional owners

Mala are considered an important taxon by Anangu for two reasons. First, older Anangu remember the species, and its economic importance, before it became locally extinct. These senior Anangu are very keen for younger generations to learn about mala and understand its historic significance within traditional society. Second, mala hold an important place in Anangu spiritual culture, and *Tjukurpa* stories tell of the presence of mala spirit beings at Uluru during the creation time.

1.4 Reintroduction of mala to UKTNP

Anangu voiced a strong desire to see locally extinct species reintroduced to Uluru for many years (pers. obs., Gillen 2000). This desire, shared by non-indigenous National Park staff and scientists, culminated in a workshop held in 1999 to investigate the feasibility of reintroducing a suite of locally extinct taxa to UKTNP. Part of the workshop focused on which particular animals were considered priority taxa for reintroduction by Anangu (Gillen et. al. 2000). Anangu women selected mala as their first priority, and Anangu men ranked the species in second place with the brush-tailed possum (*Trichosurus vulpecula*).

A second reintroduction meeting was held in Alice Springs in 2003. This forum brought together some of the attendees of the 1999 workshop with new UKTNP staff and ecologists from the (then) Northern Territory Parks and Wildlife Commission. The aim of the meeting was to identify any new developments in reintroduction science since the 1999 workshop, and subsequently decide how to progress with the return of animals to Uluru. For several reasons, it was decided that mala would be the first species reintroduced to UKTNP. First, mala had been identified as a priority species at the 1999 Workshop. Second, at the time of the meeting only one large breeding population of mala existed on mainland Australia, located at Watarrka National Park in the Northern Territory. The national Mala Recovery Team recognised the need to create additional populations in case of catastrophe at Watarrka, and thus a reintroduction of mala to Uluru would have direct conservation benefits. Third, a source population (Watarrka National Park) existed only three hours away, and staff skilled in mala husbandry were based at

Alice Springs. The meeting acknowledged that no effective technique existed to control feral cats in semi-arid Australia, and consequently the return of mala to Uluru would require the construction of a predator-proof enclosure.

Senior Anangu who had been familiar with mala prior to the species' local extinction selected a suitable site for the Uluru enclosure. The predator-proof fence design was based on the Watarrka National Park enclosure, which had proved successful in excluding introduced predators. Clearing of the 5.6 km fenceline began in April 2004, followed by construction of the fence itself. Continuing the central role of Anangu in species reintroduction work at Uluru, Mutitjulu Community members were employed to work with UKTNP staff to build the fence, and install supplementary food and water systems. The 170 hectare predator-proof enclosure was completed in August 2005 (Figure 1.11), and mala were released into the enclosure on September 25th 2005.

Figure 1.11 The UKTNP mala enclosure predator-proof fence

1.5 Study site

The Uluṟu mala enclosure is located two kilometres south-south west of the Uluṟu massif, and is dominated by irregularly oriented dunes. A total of seventy-three plant species have been recorded inside the enclosure, with a relatively small number of these taxa contributing to the majority of biomass. Dune field and swale vegetation is typical of that found across the Park (Figure 1.12).

Figure 1.12 View westward from dune crest within the Uluṟu mala enclosure, with Kata-Tjuṯa on the horizon. The photograph shows typically sparse vegetation on the dune crest, flanked by a band of dark green thryptomene (*Aluta maisoneuveii*) in the middle distance. Beyond the thryptomene, pale-coloured spinifex (*Triodia* sp.) dominates, with scattered desert oak (*Allocasuarina decaisneana*) and mulga (*Acacia aneura*)

As fire is the most likely phenomena to cause sudden and catastrophic loss of mala, significant time is devoted to managing this threat. A burnt firebreak of 100 metres depth is maintained around the external perimeter to ensure vegetation is never of sufficient density to carry fire. Further, a four metre, vegetation free perimeter track is maintained

both inside and outside the fence. Burning within the mala enclosure itself has been guided by senior Anangu, who determined a balance of approximately 50% of spinifex. in mature state (large hummocks to provide nesting habitat) and 50% in regenerating stages after fire (small clumps with other grass and herb food species present) should be maintained. To progress towards this objective, small burns have been conducted annually since 2004 (Figure 1.13).

Figure 1.13 This photograph, taken in 2011, shows a small management burn conducted within the UKTNP mala enclosure in 2008. A clear ecotone is present between the mature spinifex and the regenerating area. Two grasses found to be particularly important to mala diet, woolybutt (*Eragrostis eriopoda*) and kerosene grass (*Aristida contorta*), grew in abundance after the burn

In addition, strategic firebreaks have been burnt to split the enclosure area into smaller sections, thereby minimising the possibility of catastrophic loss of habitat, and potentially loss of mala, if a lightning strike was to cause a fire inside the fenceline. As one of the primary aims of the reintroduction project was to rapidly create a second, large mala population, food and water supplements have been provided since the release in an effort

to create good breeding conditions independent of climatic flux. Commercially manufactured macropod feed ('Roo Food' Laucke Mills, Adelaide) is supplied *ad libitum* via four steel feeders (Figure 1.14). A 2,400 litre water tank (Hills Industries, Adelaide) gravity feeds 13 drippers (Bellsouth Pty Ltd, Narre Warren) which provide water on

Figure 1.14 Four feed stations have been placed within the UKTNP mala enclosure

demand when a valve is depressed by mala (Figure 1.15). The external perimeter fence is checked daily to ensure feral camels have not caused damage during the night. Entry to the enclosure by staff is kept to a minimum to avoid mala becoming habituated to humans. Staff enter fortnightly to check the electric fence system, feeders and drippers, and to monitor the internal perimeter fence for introduced predator sign. Further, mala trapping is conducted annually to provide demographic and animal health data.

Figure 1.15 Thirteen drippers supply water for mala within the Uluṟu enclosure

1.6 Thesis objectives

My thesis has four objectives as detailed below:

1. To provide practical management guidance for the Uluṟu – Kata Tjuṯa National Park mala population, through demographic, behavioural and dietary studies of mala at Uluṟu.

As discussed above, the Uluṟu – Kata Tjuṯa National Park mala reintroduction is a culturally important program providing direct conservation benefit for the species. Fundamental to the ongoing success of the program is the adoption of best management practice for the Uluṟu population. As further investigation of mala ecology, particularly within predator-proof enclosures, was required to identify such best practice protocols, I conducted studies of population demography, mala behaviour and mala diet.

2. To provide practical management guidance for future translocations of mala.
The potential for threatened species translocations to progress the global science of translocation biology can only be realized if programs are critically assessed and the results made public (Stanley-Price 1991). To provide practical management guidance for future translocations of mala, which have been identified as necessary to ensure the survival of the species (Richards 2005), this thesis presents i) a review and analysis of Australian macropod translocations undertaken over the period 1969-2006, and ii) an assessment of the Uluru mala reintroduction, including the investigation of a suite of ecological and management issues pertaining to the program.

3. To assist with the design of reintroduction protocols for other Australian macropod species.
Translocations continue to be an important part of the recovery of a broad range of threatened Australian macropods. My review and analysis of the history of Australian macropod translocations provides the basis for management recommendations for future programs. Further, the Uluru mala reintroduction case study offers guidance with particular reference to translocations to predator-proof enclosures.

4. To document the Traditional Ecological Knowledge (TEK) of mala held by Anangu.
The documentation of TEK pertaining to mala is important for two primary reasons. First, knowledge held by senior Anangu who were familiar with the species prior to its local extinction has the potential to assist with contemporary conservation efforts. Second, the recording of TEK ensures the conservation of Indigenous ecological and cultural information, and encourages the passing of such knowledge from senior Anangu to younger generations.

1.7 Thesis overview

In an effort to achieve the objectives listed above, I have prepared two review chapters, six data chapters, and a conclusion as detailed below:

Chapter 2. The theoretical, methodological, biological and social aspects of threatened mammal translocations – a review

This chapter provides an explanatory study of the science of threatened species translocation. The multi-disciplinary aspects of undertaking such translocations are presented in detail to introduce the reader to the concepts investigated throughout the thesis.

Chapter 3. Review and analysis of Australian macropod translocations 1969-2006

Chapter 3 presents an examination and assessment of Australian macropod translocations undertaken for species conservation. Data and analysis of 109 translocations are provided in topical sections, each of which examines a different suite of translocation variables. The success or failure of Australian macropod translocations is investigated, and conclusions presented regarding appropriate translocation protocols for macropod species. This chapter provides a contextual basis for considering the reintroduction of mala to Uluru.

Chapter 4. Demography of the Uluru – Kata Tjuta National Park mala population from establishment to five years post-release 2005-2010

This chapter provides an assessment of the Uluru mala reintroduction program through the investigation of population demographics, including comparisons with the reintroduced Watarrka mala population. It concludes with recommendations for both release and ongoing management protocols for future mala translocations.

Chapter 5. A study of the home range of mala at Uluru – Kata Tjuta National Park

Chapter 5 presents a study of mala ranging behaviour at Uluru, which considers the suitability of home range size as an indicator of predator-proof enclosure carrying capacity. The importance of supplementary feed stations to study animals is also investigated. Further, the chapter compares home range size estimates at Uluru with those calculated for mala reintroduced to the wild to assess the perceived benefits of housing threatened species in large, semi-natural enclosures.

Chapter 6. A study of mala diet at Uluru - Kata Tjuta National Park including comparisons with previous analyses of free-ranging mala diet in the Tanami Desert

This chapter presents an investigation of Uluru mala diet and the importance of particular dietary components. Comparisons with previous studies of free-ranging mala are also provided. The chapter concludes with recommendations regarding how to assess enclosure carrying capacity at Uluru, and identify future release sites with regard to mala dietary requirements.

Chapter 7. A study of the behaviour of mala at Uluru – Kata Tjuta National Park, and comparisons with behavioural observations from the Arid Zone Research Institute, Alice Springs

Chapter 7 presents a study of mala behaviour within the Uluru predator-proof enclosure, and compares this to behaviour observed in the species when held in a small, intensively managed captive breeding facility. Conclusions are then drawn regarding the potential benefits, with regard to behaviour, of keeping mala within semi-natural, free-ranging enclosures.

Chapter 8. Cultural aspects of the reintroduction of mala to Uluru – Kata Tjuta National Park

This chapter documents the TEK of mala held by senior Anangu, and explores Indigenous attitudes towards both the return of mala to Uluru and threatened species recovery work in general.

Chapter 9. Recommendations for the management of existing, and establishment of future, mala populations in arid / semi-arid mainland Australia

The thesis concludes with a summary of management recommendations as identified by my study for present and future mala translocation programs. Certain recommendations may also be relevant to the translocation of other threatened macropod species.

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2. The theoretical, methodological, biological and social aspects of threatened mammal translocations – a review

2.1 Introduction

The earth is experiencing a human-induced species extinction crisis (Baillie, Hilton-Taylor and Stuart 1994, Magin et al. 1994, Isaac et al. 2007, WWF 2007). The natural environment is being altered drastically resulting in both the decline and loss of biodiversity (Dobson et al. 1997, Kleiman, et al. 1994, Seal et al. 1994). Australia provides a tragic case study, having experienced the highest modern mammalian extinction rate of any continent (Burbidge 2008). Continuing pressures facing Australia's mammals threaten to add to the 22 species, which include four macropods, that have become extinct (Burbidge 2008, Van Dyck and Strahan 2008). In an effort to address global biodiversity loss, floral and faunal translocations have become an important and increasingly popular conservation tool (Allen 1994, Atkinson et al.1994, Gittleman and Gompper 2001, Leaper et al. 1999, Mathews et al. 2006, Atkinson et al. 995, Morell 2008, Saunders and Friend 1993, Seddon 1999a, Seddon 1999b, Seddon et al. 2007, Sigg, et al. 2005, Stuart 1991, Ye et al. 2007). This chapter provides a review of published literature concerning the theoretical, methodological, biological and social aspects of undertaking threatened mammal translocations. It has been prepared, primarily, to provide a foundation and context for Chapter 2, which investigates 109 Australian macropod translocations.

2.1.2 Methods

In order to research mammal translocation biology, I searched a variety of sources, including: 40 peer-reviewed journals; books; both government and non-government conservation agency websites; translocation proposals; unpublished reports; protected area management plans; species management/recovery plans; unpublished research

theses; conference proceedings; magazines; media releases; newsletters; and tourist information brochures. Seven broad categories were subsequently created within which to discuss the body of available information:

- species chosen for translocation
- use of captive bred animals
- release area
- use of Population Viability Analysis (PVA)
- release methodology and founder group composition
- use of post-release monitoring
- evaluation of translocation success

2.1.3 Definitions

In this review I have used and expanded Cunningham's (1996:350) definition of 'translocation': '…any assisted animal movement, including introductions, reintroductions...' to also include augmentations (the release of animals into extant populations). I have adopted the IUCN (1998:6) definition of 'reintroduction': 'an attempt to establish a species in an area which was once part of its historical range, but from which it has become extirpated or become extinct'. I also use the IUCN (1998:6) definition of 'conservation/benign introductions': '…an attempt to establish a species, for the purpose of conservation, outside its recorded distribution but within an appropriate habitat and eco-geographical area'. I refer to such actions simply as 'introductions'. The term 'threatened' is used as an umbrella term for all faunal management classifications such as 'vulnerable' and 'endangered'.

2.1.4 Translocation as a tool for threatened species conservation

Translocations may assist in the recovery of threatened species populations facing a variety of extinction pressures. For example, as the natural areas available to support faunal species are degraded/reduced, population sizes will be limited by the carrying capacity of the remaining suitable habitat, thus exposing populations to increased extinction risks (Caughley and Gunn 1996, Kleiman et al. 1996, Seal et al. 1994). Such

risks are exacerbated by the restriction of natural migratory behaviour due to the disjunct nature of remnant habitat 'patches'. Thus, survival may increasingly rely upon human-induced movement of threatened species to alleviate demographic and genetic extinction pressures (Chivers 1991, Ginsberg 1994, Griffith et al. 1989, Hedrick 1995, Hein 1997, Kennedy 1992, Kleiman et al. 1994, Miller et al. 1999, Morell 2008, Shrader-Frechette and McCoy 1993, Ostermann et al. 2001, Stuart 1991). Translocation may also assist species conservation where climate change threatens existing populations (Richardson et al. 2009).

In some cases, extant populations may decline to a point where recovery by natural means (positive population growth, emigration), even when supported by in-situ management, may not be possible (Lacy 1994, Tenhumberg 2004). Translocation programs may provide an opportunity for species recovery in such drastic circumstances (Lacy 1994). Ideally, however, translocation would be used as a management tool prior to populations reaching such critically low levels (Griffith et al. 1989).

When a species' range contracts to a single population, translocation can provide an opportunity to increase its distribution through the establishment of additional populations, thus insuring against the loss of the species through catastrophe (Griffith et al. 1989, Lundie-Jenkins 1998, Reading et al. 1996a, Short et al. 1992, Seebeck 1990). Further, for taxa surviving only in captive breeding facilities, translocation provides the only option for re-establishing these species in the wild (Lundie-Jenkins 1998).

In addition to directly assisting species preservation, translocation programs may provide broader conservation benefits. For example, translocations may be undertaken in an effort to identify the causes of species decline (Short et al. 1992). This may be achieved through the experimental release of animals into areas where a range of threatening processes occur (Armstrong et al. 1994, Short et al. 1992). Translocations also provide a unique opportunity to study animal behaviour, population characteristics and ecology (Sarrazin and Barbault 1996). Further, translocations can provide tests for population

viability analysis results and theories regarding minimum viable population size (Stanley Price 1991).

Although essentially a species-specific conservation tool, translocations can also have positive effects on a broader, ecosystem-wide scale (Balmford et al. 1996). For example, the control of threatening processes through pre-release site management may benefit a variety of extant species. Further, the re-establishment or augmentation of animals considered 'keystone species' may have considerable positive affects on overall environmental health by restoring ecosystem processes (Miller et al. 1999, Seddon 1999a). The release of threatened species may also act directly upon threatening processes, such as assisting to control introduced predators (Wroe and Johnson 2003).

Translocated animals may act as flagship species, thus playing an important role in the formation of favourable public attitudes to, and thus support for, biodiversity conservation (Burgman and Lindenmayer 1998, Kleiman 1989, Lindburg 1992, Ostermann et al. 2001, Seddon et al. 2005, Serena and Williams 1994). These benefits can be maximised if the community is directly involved in translocation programs (Parker 2008). As well as contributing to the conservation of biodiversity, translocations may also be undertaken to preserve cultural links between people and the natural environment (Gibson et al. 1994). Animals considered culturally significant may therefore be the focus of translocation programs (Ausband and Foresman 2007, Bremner-Harrison 2007 pers. comm., pers. obs.)

Despite these potential uses and benefits of translocation, the technique has attracted criticism regarding its value as a conservation tool. Translocation programs have been described as risky, expensive undertakings (Burgman and Lindenmayer 1998, Miller et al. 1999, Seddon et al. 2005, Snyder et al. 1997) that should not be considered a 'conservation panacea' (Earnhardt 1999:282). Further, it has been argued that if recovery programs such as captive breeding are established with the future goal of translocation, it may lead to a false sense of security regarding the species' conservation status (Snyder et al. 1996). Consequently, resource allocation, protection of habitat and the active

management of wild populations may decline (Burgman and Lindenmayer 1998, Rahbek 1993, Snyder et al. 1996, Snyder et al. 1997). Further, translocation may not be an effective conservation initiative for all threatened species. A lack of suitable source population and/or release habitat (Stuart 1991), inability to control threatening processes (Langford and Burbidge 2001), insufficient funding (Kleiman 1989, Maguire et al. 1990, Wilson and Stanley Price 1994), logistical problems (Miller et al. 1999), legal considerations (Miller et al. 1999), lack of public and political support (Ginsberg 1994, Shrader-Frechette and McCoy 1993), and biological/ecological unsuitability of the species for translocation (Christensen and Burrows 1994, Groombridge 1992, Miller et al. 1999) may all conspire against the use of translocation for particular threatened species.

Considering the perceived limitations of threatened species translocation, authors have stressed that translocations should be adopted as part of a suite of recovery actions, and cost-benefit analysis undertaken to identify appropriate management priorities (Clark et al. 1990, Kleiman 1989, Snyder et al. 1996)

2.2 Species chosen for translocation

2.2.1 Introduction

In 2006, the World Conservation Union (IUCN) recognized over 7725 threatened animal species worldwide. While a daunting figure in itself, the IUCN stressed that only a minute percentage of known taxa had been assessed, and consequently the number of species in decline may be significantly higher (IUCN 2006).

The human value assigned to biodiversity will ultimately determine its level of preservation (Jacobson and McDuff 1998). Currently, sufficient funding is made available to conserve only a fraction of the world's threatened species (Isaac et al. 2007). As a result, decisions made to allocate scarce resources for the preservation of ecosystems or taxonomic groups will be at the cost of others considered less valuable (Rahbek 1993, May 1990). There is a growing body of work that is developing criteria to prioritise the species which require conservation action. Further, a number of management tools to prioritise recovery action among multiple threatened species have

been proposed (Hughey et al. 2002, Isaac et al. 2007, Marsh et al. 2006, Possingham 2001, Strahan 1986, Weitzman 1998). As with all conservation projects, a combination of inter-related biological, political, emotional, economic, and cultural factors will influence which species are chosen for translocation programs. Broadly, such factors may be categorized as 'biological' and 'social'.

2.2.2 Biological factors affecting the choice of species for translocation

There are many biological factors which influence the decision to select a particular species for translocation, the most significant of which are discussed below.

Conservation status

Theoretically, the level of threat to a particular taxon is a critical factor in determining the amount of management effort and funding committed to its conservation. Authors have stated that the majority of scarce resources are spent on conservation programs targeting a small number of critically endangered species (Mills et al. 1993, Possingham 2001). Such programs include translocations, and substantial costs have been reported for individual threatened species translocation programs worldwide (Fischer and Lindenmayer 2000). In reality, however, the situation may be somewhat different. Within Australia, for example, a large amount of resources is expended on species in 'lower risk' categories (eg the greater bilby *Macrotis lagotis* is listed by the federal *Environment and Biodiversity Conservation Act 1999* as 'Vulnerable', however the species is a 'flagship' for conservation throughout much of Australia and has its own national 'day' in the calendar), whilst other species in more peril receive less funding (eg *Potorous gilbertii* and *Miniopterus schreibersii bassanii* are both listed as 'Critically Endangered' and receive limited funding). Further, the expenditure of large sums of money in attempts to conserve locally rare, but globally secure, species has been recognised (Hunter and Hutchinson 1994). In such cases, social factors (presented below) may be exerting a greater influence on management effort and resource allocation than conservation status.

Mawson (2003) proposes a number of reasons why non-threatened species should also be considered for translocation. First, reintroduction of such species into their former range may further guarantee the preservation of the taxa, and contribute to overall ecosystem health. Second, the result of translocation attempts involving common taxa can provide an indication of habitat suitability for other more threatened species. Further, translocation of non-threatened taxa also provides training for conservation agency staff and community involvement opportunities.

Protection of unique taxa

It has been stated that the phylogenetic uniqueness of a taxon is one criterion which should afford priority when selecting species for conservation (Isaac et al. 2007, Moritz 1999, Morowitz 1991). Thus, the preservation of a monotypic lineage is of greater importance than conserving one of a number of closely related species. The distinctive characteristics of unique species often give them broad appeal, and result in widespread public support for their conservation. Further, the peculiar evolutionary history they represent, and consequent diversity they contribute, are also identified as compelling justifications for allocating scarce resources for their preservation (Isaac et al. 2007, Krajewski 1991). Conversely, it has also been argued that non-radiating endemic forms are ineluctably headed for extinction, and effort should therefore focus on 'evolutionarily dynamic lineages' (Erwin 1991:753). The conservation of such taxonomic groups will permit the ongoing natural development of biodiversity (Mace et al. 2003), and is of particular importance considering the increasing extinction pressures resulting from human activities (Erwin 1991). However, it has been acknowledged that no method exists for predicting future speciation, therefore designating certain species as evolutionary 'dead-ends' may be premature (Krajewski 1991). Nee and May (1997) conclude, though, that economic and social factors, rather than decisions based upon the phylogenetic status of taxa, are likely to ultimately dictate conservation priorities.

Conservation of keystone species

It has been proposed that the ecological role performed by a species should be considered when selecting taxa for recovery programs (Moritz 1999, Miller et al. 1999, Westman 1990). Keystone species are taxa identified as undertaking pivotal roles within an ecosystem, and without whom major perturbations within the system would occur (Paine 1969). Such species include top-order predators, prey species, soil 'engineers', seed dispersers and pollinators (Cox et al. 1991, Cox 2000, Dennis 2004, Kotliar et al. 1999, Miller et al. 1999, Carter 2004, Quin et al. 1996). In situations where such taxa become locally extinct, their re-establishment is essential for restoring original ecosystem function. The reintroduction of keystone species, and the resultant benefits afforded to the system as a whole, may thus avert criticism of translocations as expensive, single-species exercises (Miller et al. 1999). However, Mills et al. (1993) identify several problems with the keystone species concept, including the broad and varied definitions of what constitutes a keystone species, and the difficulty in objectively identifying such taxa. Although reaching a consensus on keystone species' definition may be difficult, further research into species' ecology will assist in the recognition of such critical taxa.

Likelihood of success

The likelihood of recovery success should be considered when selecting a species for conservation action (Moritz 1999). The medical term 'triage' has been applied to the discussion of biodiversity conservation (Noss 1996, Scheuer 1993, McIntyre et al. 1992, Baskin 1994), and acknowledges three categories of threatened taxa. First, there are some species doomed to imminent extinction regardless of recovery efforts. Second, there are taxa that have a reasonable chance of avoiding extinction if appropriate management action is undertaken. Finally, some species have a reasonable chance of persisting whether management intervention occurs or not (Noss 1996). Advocates of such an approach suggest resources should be concentrated on species assigned to the second category (Possingham 2001). However, the 'triage' concept has been criticised as providing threatened species managers with a pretext for not attempting species recovery when it would require politically unpopular action (Noss 1996).

Physiology

Differences in translocation success rates between species suggest that some taxa may be more easily re-established than others (Short et al. 1992). Physiological and behavioural characteristics may determine whether a species is suitable for translocation (Wolf et al. 1996). For example, certain physiological traits, such as relatively slow locomotion, may make particular species more vulnerable to predators, and therefore less suitable for release. Hardy, readily adaptive species may be preferable translocation candidates (Christensen and Burrows 1994). Small populations are particularly susceptible to genetic, demographic and stochastistic extinction pressures (Frankham 1998, Seebeck et al. 1990, Shaffer 1987). Highly fecund species, with the potential to rapidly increase population size, may therefore prove good candidates for reintroduction (Richards and Short 2003).

Behaviour

Many aspects of species' behaviour may directly, or indirectly, affect the success of translocation attempts. Animal behaviour can influence the effective population size, which in turn may affect the species' suitability for translocation. Wedekind (2002:16) defines effective population size as 'the size of an ideal population that loses genetic variance at the same rate as does the real population'. Consideration of effective population size is critical to small population management, as genetic extinction pressures act upon the effective, rather than census, population size (Frankham 2005). Within a given population, the number of breeding animals will be lower than the actual population size (Robinson et al. 1990). Caughley (1994) suggested that for mammals, the total population size will be twice to four times as large as the effective population size. Mating systems such as polygamy create small effective population sizes relative to the total number of potential breeders (Franklin 1980). Other behavioural traits such as infanticide may also reduce effective population size (Anthony and Blumstein 2000). Species displaying such behaviour may require large populations to achieve the necessary effective population size to maintain genetic variability and population viability (Frankham 1995, Soulé 1980). Consequently, for translocation programs, the release of large numbers of animals may be required to allow the population to grow effectively.

This may preclude some species from translocation programs, if removing relatively large numbers of animals to create founder groups would risk the viability of remaining captive/wild populations.

Trophic level may also affect the outcome of translocation attempts. Carnivore translocation has proved problematic, with substantial dispersal from the release site and strong homing behaviour contributing to program failure (Miller et al. 1999). Further evidence of trophic level affecting translocation outcome is provided by Griffith et al. (1989), who found that herbivores were the most likely group of species to be successfully translocated. Trophic level may also require consideration if a program aims to re-establish a suite of species. Carnivores, for example, should be released after the successful translocation of potential prey species (Morris 2000), to increase the chances of both successful predator and prey establishment.

As mentioned above, some species may be prone to dispersing large distances from the point of release, thereby compromising translocation success. Such post-release movement may result in increased stress and loss of condition due to excessive movement, inability to locate potential mates, moving beyond the boundary of suitable habitat, inability to take advantage of post release support (for example the provision of supplementary food and water) and entering areas outside predator control zones (Biggins et al. 1995, Hardman and Moro 2006, International Wolf Centre 2004, Stamps and Swaisgood 2007, Richards and Short 2003). Consequently, unless intensive post-release management can be undertaken, species prone to such dispersal patterns may prove unsuitable for translocation. Conversely, problems may also be encountered with species which display little movement after translocation, as sedentary behaviour may result in a build-up of wastes and associated odours. This may be problematic for species targeted by predators which rely on olfactory signals to locate prey. It has been suggested that, in certain species, high mortality rates of newly released founders may be attributable to this process (Banks et al. 2002).

Behavioural traits may also have a more direct influence on translocation success. Characteristics such as regular, predictable movement patterns, and the establishment of colonies, may leave a species particularly susceptible to predation (Christensen and Burrows 1994). The re-establishment of such species may therefore require the complete control of critical predators.

Suitable source population

Availability of animals is critical when selecting species for translocation. Sufficient animals must be available to create an effective founder group, without endangering the survival of the source population(s) (Stanley Price 1991). For some critically endangered species, extant populations may simply be too small. However, intact island populations protected from key extinction threats, successful captive breeding programs, and wild population size increases resulting from intensive management, may provide a sufficient source of animals (Friend 1991, Richards and Short 2003).

Availability of suitable release habitat

Habitat loss continues to be a fundamental cause of biodiversity decline worldwide (Dobson et al. 1997, IUCN 1994, Kleiman et al. 1994, WWF 2007). As suitable habitat has been identified as crucial to reintroduction success (Kleiman 1989, Griffith et al. 1989, IUCN 1987), availability of release sites may limit the choice of species available for reintroduction. For example, the relatively small size of protected reserves, and their disjunct distribution, may constrain the translocation of species requiring large tracts of land (Miller et al. 1999). Species requiring highly specific habitat may also suffer from a lack of suitable release sites. In addition, an inability to control certain threatening processes within the release area may restrict the selection of some species for translocation (Langford and Burbidge 2001).

Surrogate species

Initial releases of threatened species are particularly risky. Translocations are complex, multi-faceted exercises; many components of which are poorly understood (Serena and Williams 1994). To minimise the loss of valuable individuals of a particular threatened

species, relatively common and closely related taxa may first be chosen for experimental release (Fischer and Lindenmayer 2000, Reading et al. 1996a, Stanley Price and Soorae 2003). If such surrogate species are to be introduced outside their natural range, sterilization prior to release will allay fears of establishing unwanted populations (Miller et al. 1994). Research using surrogate species can also assist in determining appropriate captive breeding strategies, training regimes, release methodology, and post-release monitoring techniques (Miller et al. 1994). However, Fischer and Lindenmayer (2000) note that although the use of surrogate species in experimental translocations may be useful, different taxa are likely to respond differently to the same release conditions. In addition, surrogate programs may only release a small number of animals, thereby providing limited information regarding the likelihood of establishing a viable population of the target species.

2.2.3 Social factors affecting choice of species for translocation

The selection of species for translocation may be influenced, or even dictated, by factors other than 'scientific' considerations. Inter-related political, social and economic factors will affect planning for, and management of, threatened species conservation (Lande 1988). As human values, and subsequent actions, are responsible for the decline of the vast majority of endangered species, logically the converse should also hold. Human values and actions will be central to endangered species recovery (Kellert 1985). Indeed, some analysts have concluded that the selection of species for conservation action, including translocation, is more likely to be decided by social factors than scientific assessment and nomination (Seddon et al. 2005, Erwin 1991, Sarrazin and Barbault 1996).

Human beings inherently value some species over others (Czech et al. 1998, McIntyre et al. 1992, Strahan 1986). Attitudes held towards species are shaped by society and culture, and consequently people living in different regions may have very different perceptions of particular species (Bowen-Jones and Entwistle 2002, Gusset et al. 2008). For example, despite support from conservationists, opposition to translocation programs

for animals considered dangerous to humans may be voiced by residents within/near the proposed release area (Belden and McCown 1996, Responsive Management 1997).

In addition to 'scientific' perspectives, a species may be considered valuable on account of a variety of other characteristics. These include aesthetic appearance; usefulness; ability to provide recreational opportunities; and national, cultural or religious significance (Ausband and Foresman 2007, Gibson et al. 1995, Kellert 1985, Maguire et al. 1987, Morowitz 1991, Reading et al. 2002, Maguire et al. 1987, Strahan 1986, US Fish and Wildlife Service 2007). The value placed upon a threatened species will largely dictate the level of resources committed to aiding its recovery (Czech et al. 1998, Maguire et al. 1987, Maguire et al. 1987).

The perceived benefits of releasing high profile species may result in these taxa being preferred as translocation candidates. Kellert (1985:533) concluded that '…people are most inclined to protect endangered species that are large, aesthetically attractive, phylogenetically similar to human beings, and regarded as possessing the capacities for feeling, thought and pain'. The preservation of such 'charismatic megafauna', with their ability to garner broader support for conservation work, readily receives financial support from both government and non-government sources (Rohlf 1995:88, Simberloff 1998). It is these high profile species that are the focus of most translocation programs (Sarrazin and Barbault 1996, Seddon et al. 2005). In addition to directly assisting the released species, translocations of high profile taxa can raise public awareness of conservation issues and the role of translocation in species recovery (Burgman and Lindenmayer 1998, Kleiman 1989, Serena and Williams 1994, Seddon et al. 2007). Although conceding that such species may not be of the highest conservation priority, translocations of these taxa may also benefit other species through threat abatement at the release site (Seddon et al. 2005). The release of charismatic species may also confer benefits to threatened species management agencies. Such translocations are tangible displays of management action (Seddon et al. 2007), and therefore may generate a positive public opinion of management authorities (Rohlf 1995). However, regardless of these perceived benefits, the value of channeling substantial resources to high profile species at the expense of

others, regardless of their conservation status or ecological significance, has been questioned (Rohlf 1995, Rahbek 1993).

Economic value

Species considered economically valuable to humans may be preferentially selected for translocation. For example, threatened taxa which attract tourist revenue may be given priority as translocation candidates (Stanley Price 1991, Seddon et al. 2007). Although the translocation of certain species may provide economic benefits, the conservation of threatened taxa will often require a restriction on natural resource use within a region, or the need to accommodate the species (and its impacts) on private land (Gusset et al. 2008, Kellert 1985, McKinstry and Anderson 1999, Restani and Marzluff 2002, Shaffer 1981). Consequently, local residents and other stakeholders may actively campaign against the translocation of species considered a threat to income, economy, livelihood or recreational opportunities (Breitenmoser 1998, Reading et al. 2002, Rees 2001, Stanley Price 1999). Economic factors may be particularly relevant in developing countries where a large proportion of the rural population live in poverty (Kellert 1985). Regardless of conservation status, the choice to translocate such politically unpopular species may ultimately lie beyond the control of threatened species managers.

Religious significance

Religious significance may be an important consideration in the selection of a particular species for translocation (Ausband and Foresman 2007, Gillen et al. 2000, Langford 1999, US Fish and Wildlife Service 2007). Religious value placed on a taxon may result in public pressure and support for its conservation, thereby increasing the chance of obtaining program funding (pers. obs.). Indeed, non-government organisations may undertake translocations of such species independent of government funding (Ausband and Foresman 2007).

Management desire for quick results

Wildlife agencies may be politically motivated to fund less-threatened taxa in preference to critically endangered species. Fewer management resources may be required to secure

these species, and success would permit their removal from threatened species' lists. In addition to obvious benefits to the species, relatively fast delistings will also lead to a positive public perception of conservation agencies (Restani and Marzluff 2002).

2.3 Use of captive bred founders for translocation

2.3.1 Introduction

Threatened species translocations worldwide have used wild animals, captive bred animals or a combination of both in attempts to create or augment wild populations. Successful translocations using captive bred animals have been documented, and in some cases animals have thrived in the wild even after considerable time in captivity (Kleiman 1989, Lapidge 2005, Smeeton 2006). However, reviews of global translocations indicate higher success rates when wild caught animals are released (Fischer and Lindenmayer 2000, Griffith et al. 1989). Although the translocation of wild animals may be preferable (IUCN 1998), it is not a management option available to all taxa (Ginsberg 1994). Some species may be extinct in the wild, or be in such decline that removal of animals for translocation may threaten wild population persistence (Badridze 1994, Lundie-Jenkins 1988, Soderquist 1993, Wilson and Stanley Price 1994). Where wild populations appear ineluctably headed for extinction, some or all of the remaining animals may be captured to found captive breeding populations (Kelly 1999, Magin et al. 1994, Maguire et al. 1990). These populations are subsequently managed to provide animals for reintroduction (Dobson and Lyles 2000, Reading et al. 1996b). Propagation and release of captive born individuals may therefore be the only prospect of re-establishing some threatened species in the wild (Groombridge 1992, Lacy 1994, Lapidge 2001).

As the captive breeding environment is free from many of the sources of mortality affecting wild populations, captive breeding has the potential to provide the necessary animals for reintroduction attempts relatively quickly (Lacy 1994, Maguire et al. 1990). In addition to this primary role, captive breeding may also raise awareness of threatened species, and attract funding to conservation initiatives (de Boer 1994, Carbyn et al. 1994, Lapidge 2001). The use of captive breeding to supply animals for reintroduction is widespread (Mathews et al. 2006), and the current global extinction crisis will ensure the

number of species dependent on such 'last resort' management strategies for survival will continue to increase.

2.3.2 Genetic and behavioural considerations

The release of animals both genetically and behaviourally similar to wild conspecifics is likely to maximize the chances of translocation success. Through natural selection, a wild population will adapt, albeit imperfectly, to the biotic and abiotic environment in which it exists (Darwin 1872). Individuals thriving within the natural environment will thus contain genotypes adapted to particular conditions, while simultaneously maintaining genetic heterogeneity to cope with environmental change (Caughley and Sinclair 1994). Further, a suite of behaviours, both inherited and learned, will allow individuals to cope with the dynamic nature of the wild environment.

A population's ability to adjust to changes in the environment will be dictated by its level of genetic diversity (Anthony and Blumstein 2000, Ballou and Foose 1996, Frankham 2005, Lacy 1987). Environmental changes resulting in novel selection forces include the introduction of new diseases, parasites, predators and competitors (Lacy 1987, O'Brien et al. 1985). An example of the relationship between genetic diversity and new disease is the impact of Devil Facial Tumor Disease on the Tasmanian devil *Sarcophilus harrisii*. It has been suggested that the proliferation of the condition in the Eastern Tasmanian *S. harrissi* population may be due in part to relatively low levels of genetic diversity (Loh et al. 2006). Further, the northwestern population of *S. harrissi* differs genetically, and has yet to be affected by the condition (Loh et al. 2006). As the present rate of environmental change may well be without precedent, a lack of evolutionary potential may significantly affect the viability of threatened species (Lacy 1997). Releasing animals without appropriate genetic or behavioural characteristics may thus compromise translocation success, and therefore changes in wild traits within captivity should be avoided.

Low genetic diversity in captive breeding population founders

Inevitably, captive breeding colonies will be considerably smaller than wild populations, primarily due to limited available space, and will thus contain only a sample of the

overall genetic variation present within the wild population (Lacy 1994, Sherwin and Murray 1990, Watters and Meehan 2007). In effect, the establishment of captive population acts as a genetic bottleneck. Further, captive breeding programs may be initiated after a species has declined significantly in range, number of populations and overall number of individuals (Reading et al. 1996a). Consequently, founders taken from the wild to commence captive breeding may be from a single location, relatively closely related, and already displaying reduced genetic variation (Bradley et al. 1999, Bryant and Reed 1999, Lacy 1994). Wild population persistence must not be compromised by such actions (Ballou and Foose 1996); consequently only limited numbers may be available to establish captive colonies, further compromising founder population genetic diversity.

Agents of genetic and behavioural change within captivity

Inevitably, the biotic and abiotic captive environment will differ significantly from that of the wild. Environmental differences, and genetic problems faced by small captive colonies, can result in genetic and behavioural variation between the captive populations and their wild counterparts (Frankham 1994, Hodder and Bullock 1997, McDougall et al. 2006, McPhee 2003). As such variation may compromise the success of future attempts to re-establish animals in the wild, it is acknowledged that the preservation of genetic diversity and wild behaviour is of critical importance to captive breeding programs (Ballou and Foose 1996, de Boer 1994, Burgman et al. 1988, Frankham et al. 2000, Hedrick and Miller 1992, Kleiman 1980, Lacy 1987, Soulé 1980, Woodford and Rossiter 1994). In addition, it has been argued that ignoring behavioural and genetic changes in translocation programs would be ethically questionable (Kleiman et al. 1994, Seddon, et al. 2007).

As a consequence of small population size, captive breeding colonies are particularly susceptible to the erosion of genetic variation through drift (Earnhardt 1999). Genetic drift is defined as '…the random change in allelic frequencies that results from the sampling of gametes from generation to generation' (Hedrick 2000:229). Each successive generation contains only a sample of the alleles present in the preceding generation, and hence by chance particular alleles may increase or decrease in frequency

(Ballou and Foose 1996). The infrequent addition of new alleles through mutation will not be sufficient to counter this loss (Ballou 1984, Franklin 1980). Further, chance may determine that harmful alleles become more prevalent within the population (Frankham 2005). Therefore, as a consequence of drift, a captive bred population may, over time, develop a different genetic composition to that present within the original founder group.

Another consequence of small population size, such as those typical of captive breeding colonies, is a greater likelihood of mating between related individuals, or 'inbreeding' (Caughley 1994, Earnhardt 1999, Falconer 1989, Reading et al. 1996a). In addition to a general loss of genetic diversity, inbreeding may result in deleterious alleles becoming homozygous (Snyder et al. 1985, Amos and Balmford 2001). This is due to the increased chance of an offspring receiving the same allele from both parents when breeding pairs are related (George 1988). A reduction in individual fitness can result (Falconer 1989, Holt and Pickard 1999). When a population displays reduced fitness as a consequence of inbreeding, it is said to be suffering inbreeding depression (Frankham 2005, Lande 1988, Snyder et al. 1985).

Inevitably, animals bred and raised in captivity will display behaviour which differs from their wild counterparts. These differences result from changes in selection pressures, and the confinement of animals to a relatively depauperate environment lacking appropriate developmental stimuli (Carlstead 1996, Waples and Stagoll 1997). Selection for the captive environment may commence during the initial sourcing of animals from the wild, as some individuals may not survive the stress of capture and transportation (Pinder and Barkham 1978). Once within captivity, animals suffering stress may fail to breed, whereas individuals unlikely to compete successfully for mates in the wild may reproduce (Kleiman 1980). Traits strongly selected against in the wild may be unaffected or encouraged by the captive environment (Bremner-Harrison et al. 2004, Carlstead 1996, Snyder et al. 1996). Human involvement in animal management is likely to contribute to this process. 'Artificial selection' for certain traits begins with the initial choice of founders, and may continue into breeding programs, as the effects of human interaction inevitably (although not necessarily consciously) favour animals displaying certain

behavioural traits (Kleiman 1980, McDougall et al. 2006). These traits may include early reproduction, quiet disposition and ease of handling (Campbell 1980, Carlstead 1996). Consequently, captive breeding may result in the loss of wariness inherent in wild-born animals (Bremner-Harrison et al. 2004, Soderquist 1993), including that directed towards humans (Carlstead 1996, Waples and Stagoll 1997, Woodford and Rossiter 1994). The loss of such behaviour may be critical to reintroduction success, as captive bred animals may respond inappropriately to predators (McLean et al. 1994, Pople et al. 2001, Short et al. 1992).

In addition to issues arising from selection differences between captivity and the wild, captive bred animals may not be required to develop and use certain skills and behaviours essential for survival in the natural environment. Consequently, such abilities and actions may be lost from captive populations over time (Campbell 1980, Miller et al. 1999, Shepherdson 1994, Waples and Stagoll 1997). For some species, this scenario may arise in no more than a few generations (Blumstein et al. 2002). Further, as a result of the invariable nature of the captive environment, animals may lose the full behavioural repertoire which permits appropriate responses to environmental variation inherent in the wild (Carlstead 1996, McPhee 2003). For example, the provision of food may erode foraging skills, and the lack of complexity and space within the captivity colony may affect an animal's ability to move and navigate effectively after release (Waples and Stagoll 1997). Further, social interaction between captive animals is also likely to be significantly different to that experienced in the wild (Mathews et al. 2005, Watters and Meehan 2007). Consequently, appropriate interactive behaviour with conspecifics, such as that associated with mating and raising offspring, may be lost (Carlstead 1996). In addition, within certain species, appropriate behaviour for thriving in the wild is learned from conspecifics. Animals raised within artificial environments may have inadequate opportunities to learn such behaviour (Kleiman 1980), and this may affect their suitability as reintroduction candidates (Pinder and Barkham 1978, Snyder et al. 1996).

Thus, as a consequence of being bred in captivity, animals translocated to the wild may be exposed to a largely novel environment, in which certain behaviour must be employed

to facilitate survival. Food and shelter need to be identified and obtained, predators recognized and avoided, social interactions undertaken, successful navigation and movement through the environment achieved, and a breeding partner(s) located (Kleiman 1989, Mathews et al. 2005, Snyder et al. 1996). If captive bred animals are unfit to face such challenges, the reintroduction will inevitably fail (Waples and Stagoll 1997).

2.3.3 Animal health

In addition to genetic and behavioural considerations when captive bred animals are sourced for translocation, the potential for disease transmission, and the level of fitness of individuals, must also be evaluated.

Disease

Animals sourced from wild populations to found captive breeding colonies may be exposed to disease through both intra- and inter-species contact within the captive environment (Woodford and Rossiter 1994). Such conditions may be sub-clinical within captivity, however when coupled with increased stress suffered during transport and release may result in expression of disease symptoms within the founder group (Viggers, et al. 1993). Translocation success may consequently be compromised. Translocating captive bred animals also risks introducing novel diseases to wild taxa (Campbell 1980, Cunningham 1996, Miller et al. 1999, Woodford and Rossiter 1994). A lack of prior exposure may make wild populations particularly susceptible to such novel diseases (Snyder et al. 1996, Viggers et al. 1993).

Impact of limited space on physical fitness

The limited space available within captive breeding facilities may directly affect the wellbeing of animals and their suitability for translocation. Individuals may have insufficient opportunities for physical exercise, and consequently may suffer from problems including poor aerobic fitness and foot soreness upon release (Bradley et al. 1999, Bright and Morris 1994, Mathews et al. 2005, Waples and Stagoll 1997).

2.3.4 Addressing issues relating to the translocation of captive bred animals

Various techniques may be employed to minimize the negative effects of breeding wild animals in captivity, thereby increasing the chance of future translocation success.

Commencement of captive breeding programs

Sourcing animals from the wild before threatening processes cause dramatic population decline can increase the chance of establishing a population with genetically diverse founders. In reality, however, the general trend in threatened species management is to refrain from action until wild species are critically endangered (Lacy 1994).

Establishment of captive breeding colony

Ideally, captive breeding colonies are established with a large number of genetically dissimilar wild-caught founders, thereby reducing the severity of genetic drift and inbreeding (Ballou 1984, Comizzoli et al. 2000, Seebeck 1990, Sigg 2006, Soulé et al. 1986). Obtaining large founder numbers also secures a high proportion of the genetic diversity present in the wild. However, Rahbek (1993) argues that achieving a comparable sample of the wild gene pool is impossible, considering the size of wild populations, and the risk to source population viability from the removal of large numbers of individuals to found captive breeding colonies. Soulé et al. (1986) provide a general rule that founder groups for establishing captive colonies need to be larger than 20 breeding individuals.

As genetic problems primarily impact small populations, increasing the size of the captive breeding colony as quickly as possible, over minimal generations, until the carrying capacity is reached may reduce the initial severity of genetic drift (Ballou and Foose 1996, Lacy 1994). Further, maintaining a large captive population, and maximizing the time between generations, will assist in the retention of genetic diversity (Ballou and Foose 1996, George 1989). A relatively large captive breeding colony is also important in order to withstand the removal of animals for translocation attempts without risking the future viability of the captive population (Kleiman 1989). Lacy (1994) suggests captive colonies should be managed at a sufficient size to ensure that no greater

than 10% of the original genetic diversity present within the founder population is lost over the life of the program. In reality, constraints such as space and funding will restrict the number of animals held at captive breeding facilities (Ballou and Foose 1996).

Minimize time spent in captivity

The longer a species is held in captivity, and the more captive bred generations produced, the greater the potential loss of genetic variability and appropriate 'wild' behaviour (Campbell 1980, Ebenhard 1995, McPhee 2003). Therefore, minimizing time spent in captivity may increase the chance of re-establishing a species in the wild (Frankham 1994). However, many species will require maintenance in captive populations for substantial periods of time, as factors such as a lack of satisfactory release sites and an inability to eradicate threatening processes conspire against re-establishing these taxa in the wild (Ballou and Foose 1996, Soulé et al. 1986).

Metapopulation management

Metapopulation management refers to '…the situation where the total population of a species is divided into a number of separate sub-populations' (Craig 1994:51). These sub-populations are then cooperatively managed for the overall benefit of the species. Notwithstanding the increased chance of inbreeding resulting from splitting the population into smaller groups, such a management structure may confer benefits to retaining genetic diversity (Lacy 1994). Once separated, the individual populations will begin to diverge genetically as a result of drift. Different combinations of alleles, including rare forms, will be preserved by chance within each distinct population (Lacy 1994). Consequently, the combined genetic variation of the individual colonies would be greater than if the animals had been kept as one large, panmictic population (Lacy 1987, Margan et al. 1998). Metapopulation management may also assist in reducing the severity of selection for traits favouring a particular captive environment. This may be achieved through exposing the individuals to different environmental conditions within several breeding facilities (Lacy 1994). Ballou and Foose (1996) acknowledge, however, that similar captive selection pressures may inadvertently exist within each captive

colony, thus risking the fixation of similar alleles in each sub-population. This scenario would thus compromise overall genetic diversity.

Inbreeding pressure within individual sub-populations may be alleviated through the introduction of new genetic stock via the controlled exchange of animals between facilities (Caughley 1994, Frankham 1995). If such movements are infrequent, the unique genetic structure of each population (assuming such uniqueness has developed), may also be maintained (Lacy 1987). Where space is a limiting factor in ensuring captive populations are large enough to avoid loss of genetic variation, such an inter-population management technique provides clear benefit (George 1989). The periodic introduction of new genetic stock from wild populations can also provide considerable benefit to captive breeding colonies. Such movements can avoid divergence in genetic traits occurring between the wild and captive populations resulting from genetic drift (Lacy 1987). In addition, the introduction of new wild individuals can relieve inbreeding within the captive colony (Ballou 1984). The potential to augment captive populations with wild conspecifics is dependent, however, upon the existence of a sufficiently robust wild population able to sustain regular removal of individuals without compromising population persistence. Clearly, this management option will not be available to species extinct in the wild (Lundie-Jenkins 1998), or those persisting at very low numbers.

In addition to genetic management benefits, maintaining threatened species as small disjunct populations can mitigate against the loss of the entire species due to a catastrophic event, such as fire or disease outbreak within a sole population (Shaffer 1987). However, metapopulation management does have its potential drawbacks. Adoption of a metapopulation strategy may incur significant financial cost, and success will require overcoming logistical and legislative barriers which may hamper, in some circumstances significantly, the movement of animals between populations (Comizzoli, et al. 2000, Dixon 1994, Ginsberg 1994).

Controlled breeding

By controlling breeding pairings, and how many offspring are produced per individual, genetic problems associated with captive breeding programs may be ameliorated. Controlling breeding to ensure only unrelated or distantly related individuals are crossed can minimize inbreeding within the population (Ballou 1984, Wedekind 2002). In addition, manipulating breeding to ensure each founder produces the same number of offspring will also confer genetic benefits. A population derived from equal contributions from all founding individuals will display greater genetic variation than one containing descendents from only a limited number of animals (Ballou 1984, Frankham 1995, Hedrick and Miller 1992). As founders will achieve different rates of reproductive success, direct manipulation of breeding individuals will be necessary to equalize founder contributions (Ballou and Foose 1996).

Lacy (1994) acknowledges that the aim of equalizing founder contributions may clash with the desire to increase the population size quickly, as some founders may be poor breeders. However, he concludes that the overall benefits of fast population growth will compensate for the potential problems associated with unequal founder contributions. Moreover, after the carrying capacity of the colony is reached, future pairings can be arranged to redress any imbalance in founder lineages (Lacy 1994). However, intentionally manipulating breeding to ensure equal contribution by all founders may have unwanted consequences. As chance may determine that harmful alleles become more prevalent within small populations (Lacy 1987), captive colony founders sourced from wild populations which have already experienced significant decline may consequently be carrying relatively high levels of deleterious alleles. Ensuring that each founder contributes equally to subsequent generations within the captive breeding colony would serve therefore to encourage the prevalence of such harmful alleles (Bryant and Reed 1999). In reality, many factors may influence the breeding success of individuals, including social structure, individual health and behaviour, and management limitations (Ebenhard 1995, Snyder et al. 1996). These factors may conspire against attempts to equalize founder contributions.

Captive colony managers may seek to eliminate genetic traits considered undesirable by selecting against certain individuals (Ballou and Foose 1996). However, identifying undesirable traits may be problematic, and such breeding techniques may result in the incidental loss of desirable as well as undesirable characteristics (Hederick and Miller 1992, Lacy 1994). A further potential genetic management technique is the deliberate inbreeding of a captive population. Theoretically, selection pressure within severely inbred populations can eliminate deleterious alleles (Amos and Balmford 2001). However, Soulé (1980) advised against intentionally inbreeding captive populations in order to purge harmful alleles, arguing the negative effects of such actions would outweigh any potential benefits.

Reproductive technology and genetic resource banks

Reproductive technologies provide another avenue for maintaining and increasing genetic variation within captive breeding colonies, and reducing the risk of disease transmission. Techniques which may be employed to assist reproduction include artificial insemination, embryo transfer, and in vitro fertilization (Wildt et al. 1997). The establishment of genetic resource banks can provide the necessary material to undertake such procedures (Holt and Pickard 1999).

The preservation of founder alleles can be assisted by the collection and storage of gametes for later use (Ballou and Foose 1996, Dobson and Lyles 2000). This process permits the extension of the breeding span of founders beyond natural limitations, including death (Ballou 1984, Holt 1994). Reproductive technology may also circumvent problems such as incompatibility between individuals targeted for breeding (Seager 1983, Wildt and Wemmer 1999). In addition, the use of reproductive technology can ameliorate animal welfare and logistic issues inherent in the movement of animals by exchanging only germ plasm or embryos between facilities (Wildt et al. 1997). In addition, new genetic material may be added to captive colonies from small wild populations, without the removal of valuable animals (Ballou and Foose 1996, Seager 1983, Wildt and Wemmer 1999). Through circumventing the need to introduce new individuals, reproductive technology can also avoid the transmission of disease between

populations (Seager 1983). Further, reproductive technology may assist with the problem of lack of space within captive breeding facilities by permitting the addition of new genetic material without the need to acquire new individuals (Wildt and Wemmer 1999).

However, the use of reproductive technology requires a thorough understanding of a species' reproductive biology, and such detailed knowledge may not exist for many threatened taxa (Comizzoli et al. 2000, Pukazhenthi 2006, Wildt et al. 1997). In addition, successful cryopreservation techniques for gametes and embryos of some species have yet to be developed (Comizzoli et al. 2000, Holt and Pickard 1999). Such issues currently limit the use of reproductive technology in the management of many endangered species (Holt and Pickard 1999).

Cross-fostering is an additional reproductive technique which may be employed within captive breeding programs. Offspring taken from females and placed with closely related, more common taxa may permit genetically valuable individuals to rapidly produce further young (Department of Environment and Heritage 2006). Population growth of rare captive species may thus be increased (Smith 1998). Animals born in captivity may also be released at a very young age into 'foster' natal groups of wild conspecifics. This procedure can assist in increasing genetic diversity, and avoid individual behavioural problems associated with raising animals in the captive environment (US Fish and Wildlife Service 2004).

Captive colony carrying capacity and timing of releases

The genetic integrity and demographic balance of the captive breeding population should not be compromised by removing animals for translocation, rather only individuals superfluous to the needs of the colony should be released (Kleiman et al. 1994, Lapidge 2001). Therefore, animals should only be released once the captive population has reached carrying capacity, thus providing an important safeguard in case of reintroduction failure (Lacy 1994).

Appropriate source population and individuals for release

Considering the genetic benefits of cooperatively managing a number of smaller captive breeding colonies, it has been suggested that combining animals from each of the populations to produce a founder group can maximize the chance of translocation success (Robert et al. 2002). Further, where possible, releasing a combination of wild caught and captive bred individuals may compensate for a lack of genetic diversity in the captive cohort (Sigg 2006).

Many species occur over large geographic ranges, with some existing in multiple climatic zones. Animals translocated from captivity in one climatic zone to another may cope poorly, regardless of the fact that the release site falls within the species former range (Waples and Stagoll 1997). Further, captive colonies may exist far from the species' former range; in some cases on different continents. Problems associated with translocating species into significantly different climates may thus be exacerbated (Lapidge 2001). Therefore, when selecting founders for translocation attempts, the importance of considering climatic difference between capture and release sites should be considered (Morris 2000).

Presuming that animals may carry adaptations to the local environment, Waples and Stagoll (1997) propose that founder groups should comprise individuals descended from animals captured from, or near, the proposed release site. However, if studies show the potential source and recipient populations are genetically close, the geographical pedigree of the source population may not be significant (Hodder and Bullock 1997).

Studies of captive bred animals may identify behavioural traits considered inappropriate for the wild environment. This can aid in the selection of suitable individuals for translocation (Bremner-Harrison et al. 2004). However, the identification of appropriate behavioural traits necessary for survival in the wild is problematic, particularly when the species is extant only in captive breeding colonies (Lundie-Jenkins 1998). Therefore, it has been suggested that the founder group should include individuals which display a variety of behaviours (Watters and Meehan 2007). This may also ensure sufficient

behavioural flexibility to cope with environmental change (Watters and Meehan 2007). Activity considered abnormal to that observed in the wild may simply be appropriate reactions to the captive environment, as opposed to an indication of unsuitability for release (Carlstead 1996, Mathews et al. 2005). To avoid such erroneous judgment, comparing behaviour of captive bred animals with that of wild conspecifics (where available) in an identical, controlled environment may provide an accurate way of identifying individuals unsuitable for release (Mathews et al. 2005). However, as mentioned above, captive programs inevitably contain a level of artificial selection, and the choice of particular individuals for release risks exacerbating this problem (McPhee and Silverman 2004).

Enrichment of captive environment

Species can be encouraged to maintain, or learn, particular skills and behaviours whilst housed in breeding colonies through the enrichment of the captive environment (Biggins et al. 1999, Mathews et al. 2005). Shepherdson (1994 p.169) states '…enrichment techniques can be used to optimize the levels of social and physical stimulation to maximize reproduction and ensure normal behavioural development'. Animals may thus be more suitably equipped for survival in the wild after translocation (Shepherdson 1994). Various techniques may be employed to enrich the captive environment, potentially yielding a wide variety of benefits. Housing animals in large enclosures which mimic natural environmental complexity may assist in the maintenance of aerobic fitness and ability to move and navigate effectively (Waples and Stagoll 1997). Designing enclosures exposed to ambient weather conditions may prepare animals for the climatic variation experienced upon release, and permit insectivorous species to feed opportunistically (Soderquist and Serena 1994). Animals may be introduced to wild food items and be required to forage/hunt to locate them, thereby better preparing them to recognize and secure food resources after translocation (Biggins et al. 1999, Lapidge 2001, McLean et al. 2000, Waples and Stagoll 1997). This may be particularly important for carnivores, which may need to develop complex hunting techniques (Dobson and Lyles 2000).

For certain taxa, the social environment may be critical for the development of normal intraspecific behaviour (Carlstead 1996, Shepherdson 1994, Watters and Meehan 2007). Ensuring the appropriate social groupings are provided within the captive colony may thus enable individuals to develop behaviours more comparable to wild conspecfics (Carlstead 1996). Further, for species which form social hierarchies, housing animals destined for release within proposed groups may determine the compatibility of individuals prior to translocation (Lapidge 2001). This may avoid intense post-release competition within the founder group which may compromise translocation success (van van Dierendonck and Wallis De Vries 1996).

To permit refinement of environmental enrichment techniques, effectiveness may be assessed through post-release monitoring of behaviour displayed by the founder group (Soderquist and Serena 1994). However, it has been acknowledged that the specific ecological requirements and behavioural characteristics of certain taxa, coupled with space restrictions at breeding facilities, may not permit the effective enrichment of the captive environment (Shepherdson 1994, Watters and Meehan 2007).

Large release numbers

It has been suggested that the greater the number of animals released, the better the chance of translocation success (Fischer and Lindenmeyer 2000, Kleiman et al. 1994). Although animals may have suffered an erosion of genetic variation and 'wild' behaviour during their time in captivity, a large founder size may result in at least some animals adapting successfully to the wild, or displaying appropriate behavioral traits to ensure survival (Campbell 1980, McPhee 2003, McPhee and Silverman 2004). For some species, the successful production of a wild-born generation may be all that is required for the disappearance of behavioural traits resulting from exposure to the captive environment (Sarrazin and Barbault 1996).

Use of acclimatization pens and predator-proof enclosures

Some of the problems identified in captive bred animals which may deem them less suitable candidates for translocation may be ameliorated by the use of acclimatization

pens (which restrict animal movement in the short-term) or predator-proof enclosures (long-term). The advantages and disadvantages associated with the use of acclimatization pens is discussed in section 2.6.7. The protection provided by predator-proof enclosures negates the issue of inappropriate predator response which may be displayed by captive raised animals. In addition, excessive dispersal is also prohibited, and the restriction of animal movement assists in the scrutiny of animal health post-release.

Training

The effects of the captive environment on animal behaviour may be ameliorated through the use of active training techniques. Pre-release training may benefit individual animals, their offspring, and conspecifics to which the behaviour is transferred within the social environment (Badridze 1999, Griffin et al. 2000). In some regions, predation (especially by introduced species) is a common cause of translocation failure (Griffin et al. 2000, Short et al. 1990). Training techniques, such as exposing captive animals to unpleasant stimuli upon presentation of a model predator, may develop desired behavioural responses such as increased wariness and flight (Griffin et al. 2001). Alternatively, the controlled exposure of captive animals to potential predators, or carnivorous species similar to those present in the wild, may overcome prey naivety and assist in the development of appropriate behavioural responses (McLean et al. 2000, Short 1992). In addition to wild predator recognition and avoidance, pre-release training can be used to teach captive bred animals to avoid humans. Captive raised carnivores may also be conditioned not to prey upon domestic stock (Badridze 1999).

The effectiveness of pre-release training, type of training required, and optimum age at which to conduct training is likely to differ between species (Badridze 1994, Griffin et al. 2000, Kleiman 1989, McLean et al. 1994). Some species appear to respond rapidly to training, suggesting the technique may be a cost-effective way of increasing post-release survivorship in some taxa (Blumstein et al. 2002). The level of pre-release training required may depend upon a species' trophic level, social structure, pattern of resource utilization and habitat niche (Kleiman 1989). The retention of training effect, and its

transmission between individuals and over generations, may also vary between taxa (Griffin et al. 2000, McLean et al. 1994). In species where certain behaviour is learned from conspecifics, the introduction of wild individuals may assist with the training of captive bred animals (Griffin et al. 2000). Further, in addition to pre-release training, releasing wild caught and captive bred animals together may assist naïve animals to adjust to the wild environment (Kleiman 1989). However, involving wild animals in training would be resource intensive, and using wild individuals as 'models' for behaviour, either in training or on release, requires that such wild individuals exist and are available (Griffin et al. 2000).

Although successful under some circumstances, problems associated with pre-release training have been acknowledged. For a species that has spent considerable time in captivity, it may be not be possible to fully reinstate the suite of behaviours required to permit the species to survive and flourish after release (Miller et al. 1999). Further, it is unlikely that behaviour which has been lost entirely from a population, such as anti-predator defence behaviour, can be taught (Griffin et al. 2000). For example, learned behaviours that are passed between generations will not be recoverable once lost (Shepherdson 1994). In addition, for certain species, behaviour developed through training may have a limited 'lifespan', and consequently regular reinforcement may be required to ensure the desired behaviour is retained (McLean et al. 1994). In these cases, the timing of training prior to release may be important. Further, as pre-release training is undertaken in an artificial environment, animals may develop inappropriate responses to certain stimuli. During training, individuals can only respond within the limitations of the captive environment, which is inevitably smaller and less complex than the wild. Such responses, for example necessarily short flight distances, may not constitute successful strategies in the wild environment (McLean et al. 1994). In addition, where predator awareness and avoidance are the objectives of the training, replicating the signals that predict the arrival of a predator may be difficult to achieve in an artificial training environment (Griffin et al. 2000).

Lapidge (2001) proposed two reasons why pre-release training in predator avoidance may be ineffectual in increasing survival rates among translocated Australian macropods. First, training will be unable to accurately imitate the hunting techniques of introduced predators, and second, wild-born conspecifics are still susceptible to predation by cats and foxes, despite these species being present in Australia for well over 100 years. As effective predator response behaviour has not been developed by some species over a significant time period, it is thus unlikely to be achieved through artificial, short-term training. Predator control at the release site, rather than pre-release training, may therefore be a more prudent use of resources (Lapidge 2001).

It has been acknowledged that experimental releases to ascertain the benefits of training to post-release survival are required (Griffin et al. 2001). However, conducting robust experiments involving endangered species will be complicated by a lack of available animals, and ethical issues surrounding the release of untrained individuals (McLean et al. 1994).

Amelioration of disease risk

Various techniques exist to minimize disease risk within captive colonies, including: minimizing the time animals are kept in captivity, establishing single-species breeding colonies within the species' former range, care in selection of founder species, strict staff hygiene protocols, and not permitting the public to enter the facility (Cunningham 1996, Snyder et al. 1996). However, high costs are involved in housing captive breeding colonies under these conditions (Snyder et al. 1996).

Appropriate veterinary care prior to release can improve animal health and better prepare individuals for the wild environment. Pre-release testing of relevant taxa inhabiting the release area may assist in determining the risk of disease introduction by translocated species, and contraction of local diseases by released animals (Cunningham 1996, Woodford and Rossiter 1994). Appropriate vaccination may then be administered (Mathews et al. 2006). Treatments may also be administered to reduce parasite load, (Kleiman et al. 1994, Viggers et al. 1993) or alternatively animals may be intentionally

exposed to low levels of parasites known to exist in the release area, but absent from the captive environment (Viggers et al. 1993). In addition, captive bred animals may be quarantined prior to release, to allow sufficient time for disease incubation and the appearance of identifiable symptoms (Viggers et al. 1993, Woodford and Rossiter 1994). Further, monitoring translocated animals may assist in the prompt identification of disease issues after release (Mathews et al. 2006, Woodford and Rossiter 1994).

However, the risk of disease introduction may be minimized, but not necessarily eliminated, by the veterinary screening of animals prior to release. It may be impossible to reliably identify carriers of certain conditions, new diseases may avoid detection, and some translocation programs may lack sufficient funding to permit thorough disease screening (Mathews et al. 2006, Lundie-Jenkins 1998, Snyder et al. 1996). In addition, pre-release testing of taxa inhabiting the release area may prove expensive, impractical, and limited by a lack of knowledge of wild pathogens (Cunningham 1996). Woodford and Rossiter (1994) suggest that in reality, few translocation programs incorporate thorough disease prevention in their methodology. Further, early signs of translocation success typically lead to the abandonment of planned ongoing animal health monitoring.

2.4 Release area

2.4.1 Introduction

The existence of a suitable release area has been identified as critical to successful translocations of mammals and other fauna (Kleiman 1989, Griffith et al. 1989, Groombridge 1992, Lindenmayer 1994, IUCN 1987). Before a release site is deemed suitable, a suite of considerations must be addressed. Proposed reintroduction sites must provide for the biotic (Morris 2000), abiotic (Moseby and O'Donnell 2003), spatial (IUCN 1998) and climatic needs of the species (Lindenmayer 1994). It is clear that without the availability of the basic biological requirements such as food, shelter, sufficient area and suitable climate a viable population cannot be established. Consequently, rapid climate change may have implications for the selection of release sites. It has been acknowledged that changes in climate may require the movement of existing conservation reserve boundaries, and may also render currently suitable

translocation sites untenable (Hayward and Kerley 2009, Lindenmayer et al. 1991, Morell 2008). Further to the provision of the basic environmental needs of the species, the threatening process(es) which led to initial extinction must be absent or effectively controlled at the release site (Burgman and Lindenmayer 1998). Consequently, management to ameliorate threatening processes may be required before a site is deemed suitable for reintroduction. Site preparation may include the control of introduced predators (Christensen and Burrows 1994, Hardman and Moro 2006), the removal of introduced competitors (Short and Turner 2000), and the implementation of specific fire regimes to create appropriate structural habitat (Lundie-Jenkins 1993a). Depending on the particular threats to the re-established population, it may be necessary to continue management actions indefinitely.

In addition to the release site providing the environmental requirements necessary for species survival, maximizing the potential for translocation success may depend upon the fulfillment of other management criteria. These include the need for release sites to be readily accessible to enable long-term monitoring of released animals (Short et al. 1992, Delroy et al. 1986). In addition, land tenure must be appropriate to afford ongoing protection for the released species (Pople et al. 2001). Consideration should be given to areas that could potentially support a suite of translocated species, thereby maximising the efficiency of limited resources (Short et al. 1992). Further, successful translocation actions require the support of local people living on/near the release site, other stakeholders who will be affected by the release and government bodies (Kleiman 1989).

2.4.2 Species introductions

Supporters of the need for introductions argue that they may be necessary when threatening processes contributing to the species decline within its historic range cannot be controlled (Lomolino and Channell 1998). Indeed, the IUCN states that such ‘conservation/benign introductions’, are warranted only when no suitable area for release exists within a species’ former range (IUCN 1998). If introduction under such circumstances is advocated, the potential impact on existing biota must be considered (IUCN 1987). In addition, the introduction of animals into a ‘novel’ environment may

affect the evolutionary path of the species itself (Conant 1988). The restriction of animal dispersal afforded by release onto islands or into enclosures may allay some concerns over releasing animals outside their former range.

Within Australia, introductions are discouraged (Serena and Williams 1994), and may only be permitted by some management agencies under exceptional circumstances (Copley 1994, Department of Conservation and Land Management 1995). However, it has been suggested that in areas which have lost two closely related species, one which is extinct and the other surviving elsewhere, introductions could be used as an acceptable conservation technique. Under such circumstances, the surviving species could be introduced to the area, allowing it to occupy the ecological niche left vacant by the extinct taxon (Mala Recovery Team pers. comm., Stanley Price 1991)

2.5 Population Viability Analysis

2.5.1 Introduction

Population Viability Analysis (PVA) has been identified as a valuable and widely-used tool for threatened species preservation (Asquith 2001, Boyce 1993, Craig 1994, Ellner et al. 2002, Possingham and Davies 1995, Reed et al. 2002). PVA is an assessment of the probability that a particular population will persist for a given time period (Lacy and Clark 1990, Leaper et al. 1999). Although no strict definition of what constitutes a PVA exists (Reed et al. 2002), the term is contemporaneously used primarily to refer to computer-based analytical models (Leaper et al. 1999, Possingham and Davies 1995). Such models calculate the probability of population extinction by randomly calculating the effects of stochastic processes over a given number of simulations (Possingham and Davies 1995). Conservation managers may build species-specific models, permitting the incorporation of parameters identified as important to population persistence. Alternatively, generic PVA software packages are also available, allowing rapid assessment of population viability at relatively low cost (Possingham and Davies 1995). Further, such packages do not require specialist skills in model building (Reed et al. 2002).

PVA can provide an objective analysis of available data pertaining to a species (Brooke et al. 2002), including assisting in the identification of the population parameters critical to population persistence (Possingham and Davies 1995). PVA results can thus provide valuable management guidance (Reed et al. 2002). In addition to estimating the probability of population extinction under ambient conditions, PVA can assist in determining the effect of threats such as loss of habitat size/quality on population viability (Reed et al. 2002). Further, PVA permits the quantitative comparison of potential future population management options (Possingham et al. 2006, Seddon et al. 2007).

However, Ferson and Burgman (1995) identify two factors which prevent the accurate prediction of population viability: environmental and demographic stochasticity, and insufficient knowledge of the species. They state: 'A complete population viability analysis requires the specification of all known parameters and their dependencies...' (Ferson and Burgman 1995:104). Thus, sound knowledge of variables such as survivorship, fecundity, and emigration is required (Ferson and Burgman 1995, Nolet and Baveco 1996, Norton 1995). In addition, the relationships between these factors, their impacts at differing densities, and inter-specific relationships may require consideration in order to more accurately predict population viability (Feron and Burgman 1995, Nolet and Baveco 1996).

2.5.2 PVA and translocations

The potential value of PVA to contribute to translocation success has been acknowledged (Leaper et al. 1999, Lundie-Jenkins 1998, South et al. 2000). Indeed, it has been argued that all proposed translocation programs should incorporate PVA into planning and ongoing management (Seddon et al. 2007).

PVA has the potential to identify translocations that are unlikely to succeed, thus enabling more efficient use of finite resources and avoiding risk to rare species (Lapidge 2001). In addition, PVA permits the theoretical comparison of a range of translocation management options (Lundie-Jenkins 1998, Seddon et al. 2007). For example, PVA

models can provide estimates of population performance under a variety of release strategies (South et al. 2000). Further, through predictive modeling, PVA can assist translocation managers by providing an estimate of the optimum size and demographic characteristics of release groups (Leaper et al. 1999).

Common to the use of PVA in translocation attempts and broader conservation efforts is often a lack of necessary data. For example, feasibility studies for the reintroduction of threatened species may suffer from a lack of ecological and demographic data specific to the proposed release area as a consequence of the historic local extinction. Moreover, a lack of such data is inevitable for introduction programs. Consequently, planners may be required to use data from studies conducted under different environmental conditions from those present at the proposed release site. Consideration of environmental differences, which may significantly affect translocation success, is therefore necessary when interpreting PVA results (Leaper 1999). Further, detailed data regarding major climatic perturbations may also not be available for the proposed release site. Consequently, when relatively uninformed estimations of catastrophes are required, modeling both 'best' and 'worst case' scenarios can permit managers to adopt the mid-point between these extremes (Slotta-Bachmayr et al. 2004).

2.5.3 Limitations of PVA

Although providing many potential benefits, shortfalls with PVA have been identified. Fundamentally, the accuracy of the data used in a PVA model will determine the accuracy of results (Possingham et al. 2006). Consequently, the use of PVA to determine the likelihood of population persistence may be limited by a lack of essential data, such as age-specific fecundity and mortality (Caughley 1994). Such critical population parameters are often unknown for threatened taxa (Boyce 1993, Ferson and Burgman 1995, Leaper 1999, Possingham and Davies 1995), which may necessitate their estimation based on limited knowledge (Asquith 2001). Further, correlations between parameters may not be understood (Ferson and Burgman 1995). Similar issues regarding data shortfalls may arise when attempting to incorporate habitat availability and dispersal behaviour into PVA (Reed et al. 2002). Consequently, the accuracy of PVA will be

compromised (Asquith 2001, Caughley 1994). However, the urgency required to address threatened species decline dictates that uncertainty will be a feature of any technique used to assess population viability, not solely PVA (Brooke et al. 2002, Ellner et al. 2002).

Incorrect assumptions regarding the causal agents of population decline can result in PVA supporting inappropriate management options. That is, the PVA will only answer the questions asked of it, using only the data supplied (Asquith 2001). For example, a PVA based on genetic data, programmed to determine the best course of action to manage a threatened population, assumes that genetic factors are driving the decline (Asquith 2001). Unless they are explicitly incorporated into the PVA, other causal agents will not be considered. The adoption of particular PVA software which concentrates on specific factors may exacerbate this problem of 'threat bias' (Asquith 2001). Although this is not an inherent problem with the PVA process itself, it reinforces the need to use the tool appropriately.

It has been argued that the results of computer-generated PVAs may be accepted without due recourse to the inherent limitations of such programs. Misrepresentations of the accuracy of results may lead to the adoption of inappropriate management actions (Caughley 1994). Although PVA use may be attractive as it produces what appear to be explicit results (Reed et al. 2002), such conclusions must be carefully considered with regard to the accuracy of data entered and the limitations of the model used (Boyce 1993, Reed et al. 2002, South et al. 2000). As PVAs are models, they provide only estimations of population viability, and consequently this must acknowledged in the presentation of PVA results (Reed et al. 2002).

Examples of PVA results being incorporated into management plans prior to independent review have been identified (Asquith 2001). Further, many PVAs do not appear in the published literature, and therefore are not widely available to threatened species managers (Norton 1995). Once again, although these are problems associated with PVA, they are not issues directly related to the technique as such, but rather failings in the way results are managed.

Generic PVA model frameworks may not suit all species, as they will not have the capacity to model complex processes relevant to all situations and taxa (Reed et al. 2002, Slotta-Bachmayr et al. 2004, South et al. 2000, Southgate and Possingham 1995). Managers may, therefore, opt to design PVA models which can encompass the particular ecological characteristics of the species (Reed et al. 2002, South et al. 2000). The use of both generic and species-specific models to estimate the viability of a particular population may be beneficial by permitting the comparison of results, and thus avoiding dependence on a single PVA type (South et al. 2000).

2.5.4 Refining PVA

As PVA is based on the level of understanding of population dynamics for a given taxa, further research yielding new data may require variables within the PVA to be redefined. Consequently, to be an effective management tool for a given species, PVA will need to be frequently updated (Lacy and Clark 1990). Reed et al. (2002) stress the need to validate PVA results with field population studies. In addition, management action will affect threatened population viability, and thus change the context under which previous PVA were completed (Lacy and Clark 1990). Further, environmental change outside management control may require the revision of PVA. Acknowledging the lack of available data regarding the dynamics of threatened species populations, and the difficulties inherent in obtaining such data, studies of common species may assist in refining and testing PVA models (Boyce 1992).

PVA models used in planning translocation programs can be refined using data gathered post-release (Slotta-Bachmayr et al. 2004, South et al. 2000). In a scientifically sound translocation experiment, concurrent studies of population parameters and environmental conditions can assist in predicting population viability (Southgate and Possingham 1995). Results can then be used to refine PVA models for the species, and provide guidance for the management of the translocated population (South et al. 2000). However, logistical and budgetary constraints, and the limited availability of individuals of the threatened taxa, may make the implementation of such ideal experimental releases unfeasible (Southgate and Possingham 1995).

2.6. Release methodology and founder group composition

2.6.1 Introduction

Designing translocation strategies is a complex process, requiring the consideration of many variables including the demographic composition and size of the release group, whether all animals will be released at once or in staggered events, the timing of the release(s), whether acclimatization facilities and post-release assistance will be provided, and the degree of veterinary intervention necessary to ensure animal welfare.

Although human-induced movement of animals has occurred for thousands of years (Johnson 2006, Seddon et al. 2007), translocation as a scientific tool for species conservation is relatively new (Kleiman et al. 1994, Morell 2008). Consequently, there remains a general lack of data regarding appropriate translocation techniques for threatened species, including standardized release methodologies (Kleiman et al. 1994). However, it has also been acknowledged that environmental conditions will be unique to each translocation, thus making the standardization of release methodology, even for similar taxa, problematic (Kleiman 1996, Morrison 2002). For example, the presence of introduced predators at a release site may necessitate a different release protocol for a particular taxon than one to an area free of such pests (Moseby and O'Donnell 2003).

Whilst acknowledging the multi-faceted complexity inherent in animal translocations, the development of a cost-effective release methodology is critical to the broad-scale re-establishment of a threatened taxon (Soderquist 1993). Indeed, Griffith et al. (1989) state that the current level of species decline dictates that the identification of effective release strategies is a matter of urgency. Experimental releases under various protocols are therefore necessary to maximize the efficiency of translocation endeavours (Kleiman 1996).

2.6.2 Size of founder group

The primary aim of threatened species translocations is the establishment of (or in the case of an augmentation, assisting in the preservation of) self-sustaining wild

populations. Therefore, a sufficient number of animals must be released to ensure this goal is achieved.

The majority of reintroduction programs focus on threatened species, which by their nature exist only in small numbers. Releases will, therefore, typically involve relatively small groups of animals (Stanley Price 1991). Small populations are particularly susceptible to extinction pressures, which fall broadly into three interrelated categories: genetic, demographic and environmental (Frankham 1998, Shaffer 1987, Seebeck, et al. 1990, Soderquist 1993). In most cases, a combination of processes will drive the decline or extinction of a population (Gittleman and Gompper 2001, Seebeck et al. 1990). Consequently, reintroduction programs create small populations which are exposed to the same extinction pressures faced by threatened wild populations (Lundie-Jenkins 1998, Stanley Price 1991). MacArthur and Wilson (1967) recognized that small founder populations, before they have a chance to increase in size, are at particular risk of extinction. For example, demographic fluctuations resulting from the release of too few founders may compromise reintroduction success (Moritz 1999). Localized events such as fires, disease outbreaks or predator pressure may also extinguish small release groups. Such impacts may be particularly acute for species displaying low reproductive rates (Stanley Price 1991).

Long-term problems may also result from the release of too few founders. A consequence of small release groups may be low genetic diversity and matings between closely related individuals, resulting in inbreeding depression. Such low genetic variation will reduce fitness, may minimize the chance of adaptation to future environmental change, and ultimately endanger the establishment of a viable population (Miller et al. 1999, Sarre and Georges 2009, Sigg 2006, Smith and Hughes 2008). Franklin (1980) proposed two minimum effective population sizes for both short and long-term population viability. A minimum effective population size of 50 was suggested for immediate population survival, and 500 individuals to ensure the maintenance of genetic variability in the long-term. However, it was acknowledged that

these figures were initial estimates, and that future study would refine minimum effective population sizes (Franklin 1980).

In reality, both the short (50) and long-term (500) estimates of minimum effective population size suggested by Franklin may be too small, and populations of several hundred and several thousand individuals respectively may be required (Shaffer 1987, Reading et al. 1996b). This is due to the fact that the behavioural, demographic and stochastic conditions under which the estimates were made do not exist in the natural environment (Reading et al. 1996b, Lande 1988). For example, a minimum viable population size that may ensure the maintenance of genetic variability may be too small to cope with major environmental perturbations.

As the effective population size is determined by demographic and behavioural parameters, the minimum effective population size is likely to differ between species (Sherwin and Murray 1990). For example, if the species chosen for translocation displays behaviour which reduces effective population size, such as hierarchical mating, relatively large numbers of animals will need to be released (Ginsberg 1994).

A review of translocations undertaken in Australia, Canada, New Zealand and the United States (including Hawaii) between 1973 and 1986 found that reintroductions which released relatively larger groups were more successful (Griffith et al. 1989). Subsequent modeling revealed an asymptotic relationship between the number of animals released and rate of success. For large mammals, asymptotic inflection occurred between 20 and 40 individuals (Griffith et al. 1989). The findings of Griffith et al. (1989) were consistent with the theory proposed by Richter-Dyn and Goel (1972), who suggested a critical population number existed above which a population will successfully colonize. Hence, releasing an adequate number of individuals is critical to reintroduction success; yet releasing animals in excess of this number may provide little benefit (Griffith et al. 1989). However, Short et al. (1992) reviewed 25 Australian macropod releases and found no significant correlation between the release of larger groups and translocation success. Further, they concluded that unless the threatening processes which lead to the extinction

of the original population are removed, the numbers of animals released may have little bearing on the translocation outcome (Short et al. 1992).

Although releasing relatively large founder groups may increase the probability of establishing a viable population, it may not be appropriate or necessary in all situations. For example, the impact of released carnivores on naïve prey populations may limit the number of animals that may be translocated per release (Badridze 1994). Staggering the releases of small groups of animals over longer time frames may permit prey species to adapt to the presence of predators, thus avoiding initial severe population declines (Badridze 1994). Examples also exist whereby successful translocation has been achieved with small founder numbers (Copley 1994, Nelson et al. 1992). Therefore, species should not be dismissed as doomed, and consequently excluded from recovery action, purely due to low numbers (Hedrick and Miller 1992, Soulé 1987). Indeed, Johnson et al. (1989) noted that although hypotheses regarding founder numbers may assist translocation design, the management of the released population will be the critical determinant of success. However, ongoing advances in genetic research will permit a greater understanding of the importance of founder group size to translocation programs.

2.6.3 Release group demography

The demographic composition of the release group may be critical to translocation success (Burgman et al. 1994). When determining the composition of the release group, '…the aim is the combination of animals that will survive best with the least preparation and cost…' (Kleiman 1996:301). Determining such a combination may require a series of experimental releases at considerable expense (Kleiman 1996). Variables requiring consideration include age, gender ratio, and social structure.

Age

The age of individuals selected for release may affect translocation success (IUCN 1987, Moehrenschlager and Macdonald 2003), however Miller et al. (1999) acknowledged that determining the optimal age of animals for translocation may prove difficult. Juveniles of some taxa may record higher rates of post-release survival than adults (Miller et al.

1999, Reading et al. 1996a). Conversely, age may not significantly impact initial post-release survival in other taxa, suggesting the effect of age may be species-specific (Lapidge 2001). The selection of sexually mature animals for translocation can provide the potential for rapid population growth (Lapidge 2001, Miller et al. 1999, Richards and Short 2003). Moreover, if immature animals are released, there is a risk they may not survive to breeding age (Lapidge 2001, Tallmon et al. 2004). However, maximum age limits of animals selected for release may also be set, to avoid issues such as reduced fertility and general fitness (Lee et al. 1990, Tallmon et al. 2004).

The translocation of females nursing non-independent offspring may be problematic. When under stress, females of certain marsupial species may eject pouch young (Delroy et al. 1986, Lundie-Jenkins 1998). In addition to the potential for offspring to fail to survive the translocation process, their presence may compound the stress experienced by the mother, thus compromising her survival (Priddel and Wheeler 2004). Translocation programs involving marsupials may therefore choose to exclude females carrying large pouch young (Christensen and Burrows 1994). In other taxa, however, there may be benefits in translocating pregnant females. These include rapid population increase, and reduced transport costs when compared to releasing the equivalent number of adults (Wacher and Kichenside 1998). Further, if males other than the sires of the offspring are translocated, the founder gene pool will be larger, and subsequent matings with the released males will introduce more genes into successive generations (Wacher and Kichenside 1998).

Gender ratio

The gender ratio within the release group may be an important consideration (IUCN 1987). For certain species, males and females may differ in their abilities to cope with translocation pressures (Moehrenschlager and Macdonald 2003, Teixiera et al. 2007). For example, in their review of Australian macropod translocations, Short et al. (1992) observed a higher post-release survival rate for males. Further, similar patterns have been observed in other taxa (Moehrenschlager and Macdonald 2003). Such results may encourage the translocation of female-biased release groups (Lapidge 2001, Moehrenschlager and Macdonald 2003). Weighting release group gender ratio towards

females may also permit more rapid population growth in some species, or be used to replicate a natural gender bias in wild populations (Lapidge 2001). In addition, when undertaking augmentations of certain species, only females may be selected to avoid potentially fatal aggressive interactions between translocated and resident males (Tallmon et al. 2004). Conversely, an equal sex ratio may be chosen in order to maximise the chance of equal founder representation, thus preserving allelic diversity (Dufty et al. 1994). For certain species, the timing of release of males and females may be important. The simultaneous release of both males and females of certain species may avoid problems of excessive dispersal of males from the release site (Richards and Short 2000). For other taxa, initially releasing females and permitting them to establish home ranges prior to translocation of males may prevent such unwanted dispersal patterns (Miller et al. 1999, Soderquist 1993). Releasing familiar male/female pairs may also increase post-release survival (Gibson et al. 1993).

Social grouping

For certain taxa, the initial release of small groups, rather than individual animals, may avoid excessive dispersal from the release site and assist in establishing a cohesive population (Gibson et al. 1994). Once such a population is established, the size of release groups may be less critical (Gibson et al. 1994). Further, it has been suggested that translocation success for particular species may be increased by releasing established social units (Kleiman 1989, Lapidge 2001).

2.6.4 Number of releases

Releasing animals over a period of time, rather than as a single large founder group, may be advantageous. Such a strategy permits the evaluation of release methodology, level of understanding of threatening processes, and effectiveness of pre-release site management (Caughley 1994, Pople et al. 2001, Short and Turner 2000). The loss of significant numbers of animals due to unforeseen problems (such as high levels of predation or inappropriate dispersal patterns) or chance events (such as environmental or demographic perturbations) may thus be reduced (Short and Turner 2000). Under such a strategy, the relatively small initial release group required could be sourced from a captive breeding

colony (where available) without jeopardizing the viability of the source population (Ginsberg 1994). This would help to minimize the time a species spends in captivity prior to reintroduction (the benefits of which are discussed in 3.4.3) by avoiding the relatively long time period necessary to breed large numbers of animals. Subsequent stock bred within captivity could then be used to supplement the original release (Ginsberg 1994).

2.6.5 Timing of releases

For some taxa, the season in which releases are undertaken may be important to maximize translocation success (IUCN 1987). As discussed in section 2.7.2., the dispersal of animals far from the release site may result in a range of problems. For some species, conducting releases during the breeding season may encourage animals to stay within the release area (Badridze 1994). Further, releasing young animals at the time of year when natural dispersal from natal areas occurs may enhance post-release survival (Carbyn et al. 1994, Reading et al. 1996a).

The effects of both short and long-term climatic conditions in the release area may require consideration when planning release dates (Morris et al. 2004). When translocating species to regions which experience extremes of temperature, it may be necessary to ensure releases coincide with relatively benign climatic conditions (Lee, et al. 1990, Morris 1999). Further, seasonal availability of food may require consideration when planning translocations to ensure sufficient resources are available at the time of release (Bright and Morris 1994, Kleiman 1996, Lundie-Jenkins 1989). In addition, predation pressure may also be seasonal in nature (Morris et al. 2004). Translocation timing may also be important when releasing animals into regions which experience periodic climatic phenomena such as extended droughts (Lapidge 2001, Lundie-Jenkins 1998, Morris et al. 2004). Long-term meteorological data may assist in determining optimum release year and month, thereby assisting to minimize stressors additional to the translocation process itself (Lapidge 2001).

2.6.6 Animal Health

It has been acknowledged that translocations pose significant risks to animal health (Cunningham 1996, Margan et al. 1988, Viggers et al. 1993). The issue of disease transmission between captive and wild populations as a result of translocation activities is discussed in section 2.3.3. In addition to this issue, the potential also exists for wild caught animals to introduce disease to the release area. Translocation methodology may therefore include procedures such as veterinary screening, treatment, and quarantine to reduce the risk of disease introduction, and investigation into the possibility of released animals contracting disease at the release site (Cunningham 1996, IUCN 1998, Viggers, et al. 1993)

In addition to disease risk, stress minimization is an important consideration in animal translocations. The translocation process subjects animals to 'a prolonged and potentially stressful series of events', which may compromise program success (Teixeira et al. 2007:5). For example, stress, and potentially stress-related illness, can result from chasing animals as part of the capture process, when animals are restrained in traps, and during handling (Cole et al. 1994). Therefore, excessive pursuit whilst attempting to capture individuals should be avoided, traps should be checked frequently and designed to protect animals from extremes of temperature, and handling kept to a minimum (Cole, et al. 1994, Nelson et al. 1992). In addition, time spent in transit and ambient noise should be minimized, and care taken to ensure animals are maintained in appropriate climatic conditions during transportation (Cole et al. 1994, Lundie-Jenkins 1998, Nelson et al. 1992). Animals particularly susceptible to stress may be anaesthetised during particular stages of the translocation process (Freegard and Thomas n.d., Teixeira et al. 2007). For other species, potential stress may be reduced by the use of hoods during procedures such as the fitting of radio collars (Hardman and Moro 2006).

Excessive muscular exertion and stimulation of the nervous system, which may result from capture and restraint during the translocation process, may result in capture myopathy (Beringer et al. 1996, Cole et al. 1994). This condition results in muscle necrosis, and may result in death either during the stressful event, or up to one month

later (Shepherd et al. 1988). Australian macropods have been identified as at risk of the condition, and fatalities of translocated animals has occurred (Cole et al. 1994, Hardman and Moro 2006, Shepherd et al. 1988). In addition to general efforts to reduce animal stress, the use of Vitamin E and sedatives may assist in avoiding the onset of capture myopathy (Freegard and Thomas n.d.).

2.6.7 Soft and hard releases

Definitions of the terms 'soft release' and 'hard release' vary, and describe whether or not assistance is provided to animals after translocation. Central to post-release support under the majority of translocations referred to as 'soft releases' is the housing of animals in acclimatization pens at the translocation site. After a designated period of time, the pens are opened permitting the animals to move freely into the wild. Alternatively, pens may be designed to permit the passage of released animals to and from the 'sanctuary' of the enclosure (Soderquist 1993, Waples and Stagoll 1997). Conversely, hard releases involve the direct translocation of animals to the wild with little or no post-release assistance (Hardman and Moro 2006, Morrison 2002, Kleiman et al. 1994).

The use of acclimatization pens permits animals to adjust to local conditions including physical environment, climate and photoperiod (Kleiman 1996, Kleiman et al. 1994, Waples and Stagoll 1997, Nolan 2003). Permitting animals to become familiar with the release site environment may also encourage individuals to remain in the release area, a situation which may be critical to translocation success (Bright and Morris 1994, Morrison 2002, Stanley Price 1991). In some species, the use of acclimatization pens may deter homing behaviour post-release (Moehrenschlager and Macdonald 2003). The use of gates permitting easy exit from pens may also avoid problems with recapturing animals before release, such as flight and excessive dispersal (Lundie-Jenkins 1998). In addition, restricting animals to acclimatization pens also provides the opportunity to monitor individual health and the operation of radio transmitters before release into the wild (Gibson et al. 1994, Morrison 2002). Acclimatization pens also offer flexibility to time releases to avoid inappropriate local conditions, such as severe storms (Lundie-Jenkins 1998). Further, restriction to release pens may also permit recovery from stress, such as that induced by the transport process (Kleiman et al. 1994, Lundie-Jenkins 1998,

Teixeira et al. 2007). In addition, captive bred animals may acquire skills necessary for wild survival whilst housed in the more 'natural' environment provided by acclimatization pens (Shepherdson 1994). Fenced enclosures may also permit animals to 'encounter' predators, resulting in increased predator avoidance behaviour (Nolan 2003). This may be particularly important for animals raised in captivity (Nolan 2003).

However, the use of acclimatization facilities may not be appropriate for all translocation programs. Translocations are expensive exercises, and the additional cost of acclimating animals in pre-release pens requires justification (Hardman and Moro 2006). If trial releases reveal that translocating animals directly to the wild is successful, then program costs may be reduced (Carbyn et al. 1994, Hardman and Moro 2006, Morris et al. 2003, Stanley Price 1991). For certain species, artificial confinement to acclimatization pens may result in animal stress (Morrison 2002). For example, the confinement of animals, or grouping of an inappropriate demographic mix/number of individuals, may lead to intraspecific aggression and associated injuries (Nolan 2003, Richards and Short 2003). Further, the use of acclimatization pens may lead to a loss of condition (Hardman and Moro 2006), and in certain taxa injuries may result from collisions with enclosure fences (Christensen and Burrows 1994). Thus, for particular species, hard-release protocols may be more appropriate.

In order to minimize the stress of the translocation process, post-release assistance in the form of food, water and artificial shelter may be provided (Kleiman 1996, Richards and Short 2003, Soderquist and Serena 1994, Waples and Stagoll 1997, Teixeira et al. 2007). Such post-release support relieves individuals of the need to immediately locate essential requirements, and may also assist in controlling dispersal from the release site (Kleiman 1996, Lundie-Jenkins 1998). However, it has been suggested that in some circumstances, a small number of 'dominant' animals may monopolize supplementary food and water resources (Hardman and Moro 2006). Further, some species may choose not to utilise artificial shelter, food and water (Christensen and Burrows 1994, Richards and Short 2003, Short and Turner 2000). Therefore, monitoring is critical to determine the efficacy of post-release support.

Although post-release support may benefit animal survival and assist in their management, the nature and duration of assistance must be carefully considered. This may be particularly important for captive bred animals, as adaptation to wild conditions must be encouraged (Kleiman 1996).

2.6.8 Study of wild populations

As discussed above, determining the appropriate release protocol for translocation programs is a complex exercise involving a multitude of factors. The study of individual animals and extant wild populations, yielding biological and ecological knowledge, can provide valuable assistance in designing release methodology (Kleiman 1996, Kleiman et al. 1994, Lapidge 2001, Soderquist and Serena 1994, Waples and Stagoll 1997).

2.6.9 Factors limiting translocation methodology options

The identification of 'ideal' release methodologies for threatened species translocations will likely require the undertaking of release experiments. As threatened species survive, by nature, only in small numbers, experimental translocations to ascertain optimum release protocols may be restricted in design and thus validity of results (Hardman and Moro 2006). Further, such research may be very expensive (Kleiman 1996). In addition to experimental releases, computer modeling has the potential to provide valuable data regarding the comparison of various release strategies (Burgman et al. 1994).

Even if the 'ideal' release methodology can be identified for a particular taxa and circumstance, problems may conspire against the realisation of the protocol. For example, a lack funding may eliminate certain release alternatives (Delroy et al. 1986). Further, some translocation programs will not have a captive bred population from which to source founder animals. Consequently, animals must be procured from the wild, resulting in difficulties in obtaining the desired demographic. In addition, threatened species may simply not exist in sufficient numbers to provide relatively large release cohorts (Stanley Price 1991). Inevitable logistical constraints, such as the time available to carry out trapping, may also mitigate against obtaining the ideal founder group number and composition (Priddel and Wheeler 2004).

2.7 Post release monitoring

2.7.1 Introduction

Post-release monitoring involves the active collection of data relating to the fate of translocated animals. Monitoring may also study the impact of the translocated species on the release site environment. The need for comprehensive post-release monitoring is widely recognized as necessary to maximize the chance of translocation success (IUCN 1998, Kleiman 1989, Groombridge 1992, Seddon 1999a, Stanley Price 1991). Not only is it impossible to determine the success of a translocation without monitoring (Seddon 1999a), the establishment of a viable population may depend upon intervention after the initial release (Seddon 1999b). In spite of this, many translocations programs are subject to little or no post-release monitoring (Fischer and Lindenmayer 2000, Short et al. 1992). This is a serious flaw in the design of translocations.

The low rate of translocation success worldwide suggests a lack of understanding of the complex, interrelated factors affecting threatened species releases. In addition to the importance of post-release monitoring to the success of individual translocations attempts, advancing the science of translocation biology depends upon the critical assessment of animal releases. Without such assessment, opportunities to learn from the increasingly large number of translocation attempts will be lost (Stanley Price 1991). Such a situation risks the replication of mistakes, compromising threatened species survival and wasting scarce resources. Exacerbating this problem is the paucity of published reviews of the results of animal translocation programs (Mathews et al. 2005, Fischer and Lindenmayer 2000, Stanley Price 1991, Short et al. 1992, Saunders and Friend 1993). Reasons given for this deficiency include lack of resources and difficulties associated with post-release monitoring (Seddon 1999a). Further, failed translocations are rarely reported, yet understanding the reasons for program failure is critical to the success of future translocation attempts (Short 1992).

2.7.2 Post-release monitoring

Post-release monitoring can be used to gain a variety of information important to the assessment of translocation success. Such data include the dispersal patterns of the

founder group, animal health, population demographic, cause of death of individuals, impact of the translocation on the release site environment, and the evaluation of release strategies. The use of post-release monitoring to obtain such data is described below.

Dispersal patterns

Post-release monitoring can assist in identifying the dispersal patterns of released species (Hardman and Moro 2006, Priddel and Wheeler 2004). Such information is important (section 2.2.6), as excessive dispersal may result in a lower survival rate for those individuals who disperse long distances from the release site (Moehrenschlager and Macdonald 2003). The identification of such problems through post-release monitoring permits management intervention, and the undertaking of procedural reviews to reduce the chances of similar situations arising in subsequent releases. The fitting of radio-tracking devices to individuals, identification of signs of animal presence (for example tracks, scats, evidence of feeding) and reporting of sightings by local people may assist in monitoring founder movements (Biggins et al. 1995, Bellchambers 2001, Carbyn et al. 1994, Hardman and Moro 2006, Morris et al. 2004, Pople et al. 2001).

Animal health

The survival and vitality of released animals is critical to translocation success, and this may be monitored through the post-release capture of founders (Hardman and Moro 2005). Indicators such as body mass and breeding condition can be assessed to determine whether the animal has adapted to the wild environment (Priddel and Wheeler 2004, Richards and Short 2003). Animals clearly failing to thrive may be taken into captivity (Clark 1994). Disease, parasite load and wounds may also be identified, and medical aid administered where required (Carbyn et al. 1994, Viggers et al. 1993). Radio transmitters fitted to animals may be inspected, and any discomfort to the individual discovered (Bellchambers 2001, Priddel and Wheeler 2004).

Management authorities may demand detailed plans for proposed translocation attempts prior to granting approval to undertake such programs (New South Wales National Parks and Wildlife Service 2001, Western Australian Department of Conservation and Land

Management 1995). Translocation managers have an ethical obligation to monitor the wellbeing of animals post-release (Waples and Stagoll 1997), and 'translocation proposals' may require proponents to build in comprehensive post-release monitoring actions (ANZECC 1994, Western Australian Department of Conservation and Land Management 1995). Authorities may also require translocation proposals be assessed by an ethics committee, demanding appropriate safeguards are in place to ensure animal welfare (New South Wales National Parks and Wildlife Service 2001, Western Australian Department of Conservation and Land Management 1995).

Ascertaining cause of death

The prompt identification of the cause of death of released animals is important for the protection of surviving founders (Viggers et al. 1993). In some cases, the factors leading to the death of the released animals may be manageable, thereby mitigating against further losses, and avoiding repeated failures with subsequent releases (Friend 1991). Post-release monitoring techniques, such as the use of radio-tracking, may aid in the location of deceased animals (Short et al. 1992). Radio transmitters fitted with mortality sensors may be particularly useful in promptly locating animals which have not survived post-release (Hardman and Moro 2006). Animal remains, signs of predation and autopsies may provide evidence as to cause of death (Priddel and Wheeler 2004). In addition, inspections of the stomach contents of predators killed during control operations may reveal the remains of reintroduced animals (Dufty et al. 1994).

Demography

As founder populations are likely to be relatively small, and therefore susceptible to a range of extinction pressures (section 2.6.2), the growth of the release cohort is critical to the establishment of a viable population (Stanley Price 1991). Post-release capture of animals can identify new individuals born after release, thereby allowing population growth and size to be estimated (Richards and Short 2003). Further, sex ratio and age structure can be ascertained (Fischer and Lindenmayer 2000). Other monitoring techniques, such as spotlight surveys and the identification of animal sign, may also provide a guide to population numbers (Pople et al. 2001, Morris et al. 2004).

Impact of the released species on the environment

Translocated populations may have unwanted impacts on the release site environment, including the transmission of disease and decline of certain food species (Viggers et al. 1993, Copley 1994, Gittleman and Gompper 2001). Such impacts may be particularly acute when taxa are introduced to island environments, particularly when endemic species have not co-evolved with the translocated animal (Wilson and Stanley Price 1994). Further, where animals are released to augment an existing population, intra-specific competition and disturbance to social groups may be problematic (Kleiman 1989, Margan et al. 1998). Post-release monitoring may identify such issues, and permit early management intervention.

Evaluating effectiveness of release techniques

Conducting translocations as experiments provides the opportunity to test hypotheses regarding the effectiveness of various release techniques. For example, providing two separate founder groups with different levels of post-release assistance may provide information as to the optimal release strategy for particular species (Hardman and Moro 2006). Similarly, comparing the results of releases undertaken during different seasons may increase the chance of future translocation success (Carbyn et al. 1994). Such studies rely on post-release monitoring to produce quantifiable results (Armstrong et al. 1994).

Opportunity for population study

Translocations also provide opportunities for study that would otherwise require considerable, undesirable disturbance of wild populations (Sarrazin and Barbault 1996). As discussed in section 6.2, translocation programs typically create small populations, which are threatened by a range of extinction pressures. As a result, translocations provide an opportunity to study the effects of these pressures on populations of known individuals (Sarrazin and Barbault 1996, Serena and Williams 1994, Stanley Price 1991). Translocations also provide an opportunity to test hypotheses regarding other causes of species decline, such as predation by introduced species. For example, reintroductions permit the rapid collection of accurate data regarding the impact of predators on a

population of known demography (Pavey 2006). Further, simultaneous releases into two areas, one managed to remove the threatening process under investigation, and the other left untreated, may provide valuable information to assist broader species conservation (Armstrong et al. 1994). The effectiveness of management actions, such as predator control, may also be assessed (Short 1991).

Animal welfare, technical and management problems

Animal welfare issues concerning post-release monitoring and experimental translocations may arise. For example, the stressful effects of the translocation process may be exacerbated by trapping shortly after release. Trapping may also interfere with behaviour such as feeding patterns (Priddel and Wheeler 2004). In addition, the presence of radio collars may result in altered behaviour, loss of condition and minor injuries (Tuyttens et al. 2002, Priddel and Wheeler 2004). Radio collars have also been implicated in the death of translocated individuals, through the snagging of collars on vegetation, increased predation risk, and the entrapment of limbs between the collar and the neck (Morris et al. 2003, Richards and Short 2003). Animal welfare issues also require consideration when experimental releases are proposed. For example, in order to test the impact of a perceived threatening process, animals may be released into an area where the particular threat is present. Intentionally leaving threatening processes believed responsible for the species decline unmanaged at a release site may be viewed as unethical (Armstrong et al. 1994). Further, as the majority of translocations release threatened species, individuals may be too valuable to risk under such release scenarios (Serena and Williams 1994). Technical problems with post-release monitoring may also be encountered, including radio collar transmitter failure, the detachment of transmitters from individuals, dispersal of animals beyond the range of receiving equipment, and the use of ineffective trapping techniques (Morris et al. 2003, Lee et al. 1990).

In addition to technical problems and negative impacts on released animals, post-release monitoring may present other management difficulties. The IUCN recommends that monitoring be undertaken for a substantial period of time after release, to ascertain the success, or otherwise, of the program (IUCN 1987). Such long-term monitoring requires

a high level of management commitment and funding (Viggers et al. 1993). This requirement may limit post-release monitoring, particularly in remote areas (Morris et al. 2004, Delroy et al. 1986). For example, Australia's arid zone supports a low population density and few settlements, and the consequent lack of infrastructure can make satisfactory monitoring problematic.

2.8 Evaluation of translocation programs

2.8.1 Evaluation

The importance of establishing criteria with which to judge translocation success prior to release has been widely acknowledged (IUCN 1998, Kleiman et al. 1994, Morris 1999, Stanley Price 1991, Sarrazin and Barbault 1996, Saunders and Friend 1993). Without such criteria, translocation attempts cannot be effectively evaluated, and thus opportunities for advancing the science of reintroduction biology will be lost (Kleiman, et al. 1994, Miller et al. 1999). Further, ethical responsibilities for the welfare of released animals require the ability to effectively assess translocation success (Waples and Stagoll 1997). Identifying criteria for evaluating the success of a translocation program is also critical to the development of post-release management plans. For example, if pre-established indicators of translocation failure are apparent, management plans to ameliorate the situation can be promptly employed (Kleiman 1996, Kleiman et al. 1994). It has, however, been acknowledged that no established criteria for assessing translocation success have been developed, making objective judgment of program outcomes problematic (Beck et al. 1994, Fischer and Lindenmayer 2000, Kleiman 1996, Seddon 1999). A variety of factors, discussed below, may have contributed to this situation.

Creating a set of standard quantitative and temporal measurements of translocation success, applicable across taxa, may be impractical (Lundie-Jenkins 1998, Seddon 1999). Stanley Price (1991) states that criteria set to measure translocation success will be dictated by a species' biological and demographic characteristics, and the minimum viable population size required to ensure persistence. Therefore, such criteria will be species-specific. Further, critical to the establishment of success criteria will be a sound

understanding of species' biology (Kleiman et al. 1994, Sarrazin and Barbault 1996). Typically, however, a paucity of such knowledge exists for threatened species involved in translocation programs (Booth 1988, Southgate 1994). In addition, individual translocations may have different goals, and hence success will be measured in terms of program-specific objectives (Fischer and Lindenmayer 2000, Kleiman 1989, Kleiman et al. 1994, Waples and Stagoll 1997). Thus, what constitutes a failure in one program may be considered a success in another. However, success criteria have been applied to reviews of multi-species translocations. In their review of global translocations, Beck et al. (1994:273) deemed programs successful if '…the wild population subsequently reached at least 500 individuals which are free of provisioning or other human support, or where a formal genetic/demographic analysis…predicts that the population will be self-sustaining'. Short et al. (1992) suggested translocations were successful if populations survived for more than five years and appeared likely to persist.

It has been widely espoused that the primary aim of threatened species translocations is the establishment of (or in the case of an augmentation, assisting in the preservation of) self-sustaining wild populations (Fischer and Lindenmayer 2000, Griffith et al. 1989, Sarrazin and Barbault 1996, Seddon 1999, Kleiman, Stanley Price and Beck 1994, Stanley Price 1991, Waples and Stagoll 1997, Wolf et al. 1996). Whilst 'self-sustaining' assumes an absence of human intervention, considering the ubiquitous and generally negative human impact on natural systems, it may be unlikely that completely unmanaged populations could persist in the long term (Lundie-Jenkins 1988). Indeed, the IUCN (1998) concedes that long-term management may be required; however such intervention should be minimal. For example, as habitat destruction has been acknowledged as the most important overall cause of worldwide species decline (IUCN 1994), suitably large release sites may be unavailable for some taxa (Miller et al. 1999). If translocation is still recommended as the most appropriate method of species recovery, ongoing translocations of animals may be required to counter restriction on natural migration due to habitat fragmentation (Ginsberg 1994).

Assuming the establishment of a self-sustaining wild population is the primary aim of translocations; the critical indicator of success must therefore be the existence of such a population. However, it has been acknowledged that the long-term nature of translocation programs mitigates against prompt assessment of success (Smith and Hughes 2008, Wilson and Stanley Price 1994). Early 'success' may be followed by decline and ultimately failure (Morell 2008), and consequently it may be too early to judge the success of many recent translocation attempts (Ebenhard 1995, Wilson and Stanley Price 1994). If early assessments are attempted, incorrect conclusions may lead to the premature withdrawal of population management actions (Seddon 1999).

Clearly, a population must breed successfully to be viable, therefore analysis of success within the lifespan of the released species could be misleading (Wolf et al. 1996). Further, the inherently dynamic nature of faunal populations restricts the assessment of translocation success to a moment in time (Seddon 1999). Considering that population fluctuations may occur over considerable time periods, short-term declarations of the establishment of viable populations should be treated with caution (Seddon 1999). For example, sufficient time must elapse prior to evaluation to ensure the population can survive seasonal variation, and more serious cyclical perturbations, in environmental conditions (Lundie-Jenkins 1988, Pople et al. 2001, Stanley Price 1991). Further, other extinction threats, such as those posed by human resource use, may also fluctuate (Gorman 1999). In addition to natural population flux, translocation methodology may impact population size after the initial release. Translocations may involve separate releases conducted over several years (Griffith et al. 1989), therefore any assessment of self-sustainability would need to be conducted at a specified time after the final release. Further, high levels of mortality within founder groups may occur during initial translocations of threatened species. However, Miller et al. (1994) warned against judging the success of translocation programs on such early losses, as initial releases may be critical to evaluating factors such as release protocols, and the suitability of founder source populations. Indeed, Southgate (1994) has gone so far as to say that regardless of survivorship, translocations should only be deemed failures if they do not contribute to a greater understanding of threatened species conservation.

Considering the above, criteria may be applied to specific, critical periods after release in order to monitor translocation progress towards the long-term goal of establishing a viable population. For example, appropriate assessment may be made of initial release group survivorship, breeding success of the first wild-born generation, and population growth for a set period after wild breeding has occurred (Backhouse et al. 1994, Morris 1999, Richards and Short 2003, Seddon 1999). Other demographic characteristics, such as age and sex ratios, may also be monitored at specific times (Fischer and Lindenmeyer 2000). Further, targets may be set for ending management intervention, such as supplementary releases (Backhouse et al. 1994, Richards and Short 2003). Where extant populations of conspecifics exist, biological, ecological and demographic comparisons between released captive bred animals and their wild counterparts may assist in determining if translocated animals are successfully established in the wild (Miller et al. 1999, Moehrenschlager and Macdonald 2003, Soderquist and Serena 1994, Waples and Stagoll 1997). Predictive models based on data from wild populations can be used to evaluate translocation success by producing quantitative, temporal targets for released populations (Short and Turner 2000, Stanley Price 1991).

2.8.2 Global evaluations of translocation success

In their review of 145 global reintroductions of captive bred animals between 1900 and 1994, Beck et al. (1994) concluded that only 11% of the reintroduction projects undertaken were successful. After analyzing data pertaining to 87 conservation translocations conducted worldwide, Fischer and Lindenmayer (2000) reported a success and failure rate of 23% and 26% respectively. The outcome of the remaining 51% of translocations could not be determined at the time of publication. In a previous major study, Griffith et al. (1989) reported a success rate of 44% (n = 80) of threatened species translocations conducted in Australia, New Zealand, Canada and the United States (including Hawaii) between 1973 and 1986. In both of these reviews, translocations were considered successful if they resulted in the establishment of self-sustaining populations. When Wolf et al. (1996) re-visited the same translocation programs investigated by Griffith et al. (1989) in 1993, they identified no significant change in rate of success. As discussed above, the long-term nature of translocation programs mitigate against prompt

assessment of success. The inability of Fischer and Lindenmayer (2000) to determine the result of half of the translocations investigated makes comparisons between the success rates identified by these authors and Griffith et al. (1989) problematic.

Further to the low overall success rate of threatened species translocations (Mathews et al. 2005, Sigg et al. 2005, Teixeira et al. 2007), some authors have suggested that proponents of failed translocations may be unwilling to publish results, or respond to surveys seeking data regarding release programs (Fischer and Lindenmayer 2000, Reading et al. 1997). Surveys and published results may therefore lead to overly optimistic conclusions regarding rates of translocation success.

2.8.3 Factors contributing to translocation success

It has been widely acknowledged that species recovery logically depends upon the successful identification and removal of the threatening process(es) which lead to the original decline of the taxa (Lacy and Clarke 1990, Seebeck et al. 1990, Shaffer 1981, Terborgh and Winter 1980). The principal cause, or combination of causes, of species decline may be cryptic, and consequently impossible to ascertain with certainty (Serena and Williams 1994). However, threats which have been identified as contributing to global species decline include habitat fragmentation, disease, road kill, pollution, climate change, introduced predators and competitors, changed fire regimes and direct exploitation/persecution (Brown 1991, Burbidge and McKenzie 1989, Dickman 1992, IUCN 1998, Kleiman 1996, McCarty 2001, Miller et al. 1999, Pople et al. 2001, Seebeck et al. 1990, Shaffer 1981, Short et al. 1992, Stuart 1991, Viggers et al. 1993). Of these, the IUCN (1994) identifies loss of habitat as the primary cause. In the Australian context, however, it has been argued that predation by introduced species has been the most critical causal factor in small mammal decline (Johnson 2006), and this process can take place in what appears to be relatively pristine environments.

Ultimately, faunal decline and population extinction is usually driven by a combination of the abovementioned factors, which are compounded as the population becomes progressively smaller (Burbidge and McKenzie 1989, Clark et al. 1990, Gittleman and

Gommper 2001, Possingham 2001, Seebeck et al. 1990). For example, habitat degradation and fragmentation may result in a number of small disjunct populations of a particular species, which are then susceptible to introduced predation pressure, catastrophic events, and demographic/genetic extinction pressures (Clark et al. 1990, Johnson et al. 1989, Seebeck et al. 1990).

The correct identification, and effective control, of threatening processes is thus a critical primary step to translocation success (Caughley 1994, Fischer and Lindenmayer 2000, IUCN 1998, Johnson et al. 1989, Miller et al. 1999, Ostermann et al. 2001, Short and Turner 2000, Stanley Price and Soorae 2003). Without the control of such processes, translocation efforts will be ultimately futile, regardless of the release methodology employed (Richards and Short 2003). Indeed, failure of past translocation programs has been attributed to the lack of identification, or inability to control, the threatening process(es) that lead to the original decline of the species (Moritz 1999, Snyder 1996). Further, releases under these conditions may be deemed unethical (Lindenmayer 1994). Ultimately, an inability to effectively control threats may prohibit the use of translocation as a recovery tool for some threatened taxa (Ebenhard 1995).

Where the original cause of a species' decline is unclear, it has been suggested that conducting experimental translocations could assist in their identification (Armstrong et al. 1994, Christensen and Burrows 1994, Serena and Williams 1994, Short 1991, Soderquist 1994). However, problems in undertaking such experiments, including logistical difficulties involved in establishing controls, the rarity of species involved, and ethical considerations in releasing animals to potentially sub-optimal environments have also been acknowledged (Armstrong et al. 1994, Serena and Williams 1994). In addition, Soderquist (1994) has sought to qualify the results of such experiments, stating that results will only be directly applicable to current threatening processes, as environmental change may have occurred since the original decline of the species.

2.8.4 Suspected / known causes of translocation failure

Although it has been acknowledged that the causes of translocation failure have not always been clearly established (Pople et al. 2001, Ostermann et al. 2001), several factors have been identified as directly contributing to the failure of conservation translocations. The inability to control introduced predators in Australia has resulted in translocation failure in a wide variety of taxa (Christensen and Burrows, Pople et al. 2001, Short et al. 1992). Relatively low rates of predation by introduced carnivores may be sufficient to ultimately thwart translocation attempts (Pople et al. 2001). Indeed, it has been stated that the successful translocation of Australian terrestrial mammal species up to 5.5 kilograms in weight is dependent upon the control of introduced predators (Brown 1991, Johnson et al. 1989). Consequently, in areas where introduced predators cannot be eradicated, wild release of susceptible species may not be feasible (Christensen and Burrows 1994). However, predator/prey models suggest that in some cases, a reduction in the number of introduced predators, rather than complete eradication, may enable translocations to succeed (Sinclair et al. 1998). Further, Johnson (2006) suggests that a reduction in introduced prey species (namely rabbits), an increase in native predator numbers (dingoes) and the management of vegetation to benefit native mammals may permit successful translocations even without direct efforts to control introduced predators.

In addition to the impact of introduced carnivores, an inability to control predation by native species has also contributed to translocation failure (Wolf et al. 1996). Further, the lack of effective control of poaching has compromised translocation success in some countries (Gorman 1999, Morell 2008). Stress has also been identified as a factor contributing to translocation failure, and Teixeira et al. (2007) believe its effects may often be underestimated. As described in section 2.6.6 above, the process of translocation involves numerous steps which may cause acute/chronic stress in animals. Further, inappropriate release methodology may compromise translocation success. When too few individuals are released, significant demographic fluctuations caused by chance events may lead to translocation failure (Moritz 1999). Indeed, the release of too

few animals has been identified as an important causal agent in the decline or failure of a small number of translocations (Wolf et al. 1996).

Selection of unsuitable release candidates may also result in translocation failure. In some translocations, inappropriate behaviour, such as an inability to successfully obtain food, evade predators, migrate and interact appropriately with conspecifics, has prevented the establishment of viable populations (Snyder et al. 1996, Wolf et al. 1996). In addition, environmental catastrophes, such as bushfires and severe weather events, were identified as significant contributors to the failure of a small number of translocation attempts investigated by Wolf et al. (1996). Further, the dispersal of animals beyond the release site into sub-optimal habitat, and the impact of disease, have been identified as directly contributing to the failure of a small number of translocations (Dobson and Lyles 2000, International Wolf Centre 2004, US Fish and Wildlife Service n.d.).

In the following chapter, 109 Australian macropod translocation programs are analysed and discussed. Included is an attempt to identify the factors critical to successful translocations, and an investigation into those which were suspected/known to have caused programs to fail. Comparisons with global translocation assessments are also made.

2.9 References

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Chapter 3. Review and analysis of Australian macropod translocations 1969-2006

3.1 Introduction

Since European colonisation, Australia has experienced a mammal extinction rate unparalleled on any other continent (Brown 1991, Burbidge and McKenzie 1989). Six species of the Superfamily Macropodoidea have become extinct (Burbidge 1995), and a further four are extinct on the mainland; their ranges having contracted to offshore islands (Strahan 1995). Many others have had their continental distributions reduced alarmingly (Strahan 1995), with twelve mainland taxa currently listed as threatened under the *Environment Protection and Biodiversity Act 1999* (Department of Environment, Water, Heritage and the Arts n.d.).

The intentional movement of mammals, including macropods, has been undertaken in Australia since the early 1900s (Copley 1994). Releases conducted as early as the 1920s and 1930s included a number of translocations aimed at species conservation (Copley 1994). Considering the current suite of extinction threats facing Australian fauna, animal translocation has been identified as an increasingly important species recovery tool (Brown 1991, Serena 1994). Indeed, the IUCN action plan for the conservation of Australasian marsupials and monotremes (Kennedy 1992) recommends the use of translocation to aid the recovery of eight out of ten threatened macropod taxa. Similarly, translocation is advocated for four of the five macropod species listed as threatened under the *Environment Protection and Biodiversity Conservation Act 1999* for which recovery plans have been completed (Department of Environment, Water, Heritage and the Arts 2008).

The only multi-species review of Australia-wide macropod translocations preceding this study was published by Short et al. (1992), in which the authors provided overview and analysis of 25 translocations. These programs involved 12 taxa released between 1905 and 1989, the majority of which were undertaken with the aim of species conservation. The paper also included translocation management recommendations. In addition to that nationwide review, Copley (1994) provided a detailed examination of mammal translocations in South Australia, including 17 macropod translocations conducted between 1907 and 1992.

Many additional macropod translocations have been attempted since these studies were published. Further, the past 15 years have seen progress in a range of technologies that may assist the translocation process. These include the development of PVA software and advances in the analysis and interpretation of genetic data. My study aims to identify such changes in translocation methodology, and to identify links with program success/failure. It provides data regarding 109 Australian macropod translocations conducted between 1969 and 2006. These translocations include those undertaken for species conservation described by Short et al. (1992) and Copley (1994) over this period. These programs are revisited, and in some cases additional information obtained permitting further analysis. Translocations conducted prior to 1969 were omitted due to a lack of available data. Data and analysis are provided in topical sections, each of which examines a different suite of translocation variables. Finally, the success/failure of Australian macropod translocations is investigated.

3.2 Methods

3.2.1 Background to the study

In September 2005, 24 *Lagochestes hirsutus* (mala) were translocated from a predator-proof enclosure at Watarrka National Park to a newly built enclosure at Uluṟu - Kata Tjuṯa National Park. This review and analysis of Australian macropod translocations was undertaken to provide a context for the Uluṟu mala release within the history of such conservation work in Australia. Further, the study sought to assess the state of knowledge, and level of success, of translocations undertaken prior to 1 January 2007.

Through the investigation of translocation tools and methodology, and analysis of program results, the chapter aims to provide a greater understanding of the requirements for successful translocation of macropod species.

3.2.2 Sources of data

In order to identify Australian macropod translocations, I searched a variety of sources, including: 40 peer-reviewed journals; books; both government and non-government conservation agency websites; translocation proposals; unpublished reports; protected area management plans; species management/recovery plans; unpublished research theses; conference proceedings; magazines; media releases; newsletters; and tourist information brochures. Data regarding macropod releases found in scientific and popular literature were collated, and questionnaires sent to translocation proponents seeking additional information where required. In addition, questionnaires were sent to translocation proponents regarding known translocations which had not been published in any form. Macropod translocations conducted for reasons other than conservation, such as to provide a food source or tourist attraction, were not considered.

Similar to Beck et al (1994), when gathering data I considered a translocation as a distinct program moving a particular species to a particular location. Thus, ongoing releases of the same species to the same area over time are considered a single translocation program.

Data on Australian macropod translocations are presented and discussed within the context of global trends in species translocation. Results (section 3.3) and discussion (section 3.4) are grouped under similar broad headings to those used in Chapter 2, with some additions:

- location of translocations by state/territory
- species chosen for translocation
- use of captive bred animals for translocation
- release area

- use of Population Viability Analysis (PVA)
- release methodology and founder group composition
- use of post-release monitoring
- translocation success
- suspected/known causes of translocation failure
- methodological and environmental factors critical to Australian macropod translocation success

3.2.3 Location of translocations by State/Territory

Translocations were listed as per the state/territory in which the release occurred, with the Australian Capital Territory included in New South Wales. I determined the number of potential macropod species available for translocation per state using the following:

- the number of species that are no longer present in a particular state, but are still extant in other states
- the number of species that no longer occur on the mainland territory of a particular state, but are extant on that state's offshore island(s)
- the number of species that have experienced range decline since colonisation

3.2.4 Pre-release training

Considering 29% of Australian macropod translocations investigated released animals from captive breeding colonies, data was sought regarding whether or not pre-release training had been attempted. It was assumed that pre-release training may be undertaken by managers in each of the three situations detailed in Table 3.1 below. Data regarding releases to predator-proof enclosures 100 hectares or greater were not sought, as managers are largely able to control threatening processes, and problems such as inefficient foraging behaviour can be mitigated. Pre-release training is therefore unlikely to be attempted as part of these translocations.

Table 3.1 Situations where managers would be expected to consider pre-release training as part of the translocation process

Source	Release site	
Captive breeding colony	Mainland/island	Threatening processes are expected to be absent from captive breeding colonies, however released animals are likely to face a variety of threats in an mainland/island release site
Predator-proof enclosure	Mainland/island	Although some threatening processes may be present in a predator-proof enclosure, it would be expected that a broader suite of threats may occur in a mainland/island release site
Island	Mainland	Although some threatening processes may be present in the island environment, it would be expected that a broader suite of threats may occur in a mainland release site

3.2.5 Release area

In this review I have used and expanded Cunningham's (1996:350) definition of 'translocation': '…any assisted animal movement, including introductions, reintroductions...' to also include augmentations (the release of animals into extant populations). I have adopted the IUCN (1998:6) definition of 'reintroduction': 'an attempt to establish a species in an area which was once part of its historical range, but from which it has become extirpated or become extinct'. I also use the IUCN (1998:6) definition of 'conservation/benign introductions': '…an attempt to establish a species, for the purpose of conservation, outside its recorded distribution but within an appropriate habitat and eco-geographical area'. I refer to such actions simply as 'introductions'. The term 'threatened' is used as an umbrella term for all faunal management classifications such as 'vulnerable' and 'endangered'.

This study investigated three characteristics of release areas selected for Australian macropod translocations, specifically: i) whether the release was inside (a reintroduction) or outside (an introduction) the species' former range; ii) whether the release site was

located on mainland Australian, an offshore island, or within a predator-proof enclosure; and iii) whether or not the release site was actively managed.

The ability to address the threatening processes leading to the local decline/extinction of the species will to a large extent dictate the type of release site chosen for translocation attempts. By placing the location of Australian macropod release sites into three categories, namely 1) mainland, 2) island, 3) predator-proof enclosures, the relative popularity of each could be ascertained.

For this study, only 'free range' predator-proof enclosures of 100 hectares or greater were considered translocation sites. Although a somewhat arbitrary figure, it was reasoned that for small macropods (the candidates in nearly all Australian macropod translocations) at low densities, a minimum of 100 hectares may provide the opportunity for animals to display relatively 'natural' behaviour, for example foraging and mate selection. Conversely, enclosures smaller than 100 hectares were commonly intensively managed, including controlling the interaction between animals.

Information regarding the use of pre- or post-release site management for releases to predator-proof enclosures was not sought, due to the inherent differences in management regimes between these and other release sites. As enclosures produce a 'semi-artificial' environment, certain forms of management critical to other releases, such as ongoing predator baiting at the release site, are unlikely to be required. Further, other forms of post-release management, such as the ongoing provision of supplementary food, may be necessary. Therefore the inclusion of translocations to predator-proof enclosures in the analysis of the overall data set could obfuscate analysis and findings. However, fire management for ecological or fuel reduction purposes was included in the analysis, as predator-proof enclosures do not negate the need for management of this type.

Several translocations investigated were to peninsulas which had been fenced across the isthmus in an effort to manage introduced species. It could be argued that such areas are in fact similar to predator-proof enclosures, and therefore should not be included when

considering the use of pre/post-release site management. However, as these areas were bounded on all sides but one by natural barriers, which proved at times to be ineffective against introduced predators, they have been included as 'mainland' sites in the following data presentation, analysis and discussion.

Within the set of Australian macropod translocations investigated, two were releases onto islands within mainland water bodies. Although explicit statements regarding why the release site was chosen could only be obtained for one of the translocations, it could be assumed that common to both programs was a desire to release animals to a predator-free area. Unfortunately one, and possibly both, of these programs failed due to predation, suggesting that predators were still able to reach these islands. As there were only two such releases, and their outcomes were likely to have been determined by the same factors as those to mainland release sites, they have been included as 'mainland' sites in the data presentation and discussion.

3.2.6 Use of Population Viability Analysis

Data was sought regarding whether or not Australian macropod translocations used PVA as part of their program planning and management. As the first published papers regarding PVA appeared in the late 1980s, the use of PVA was only investigated for translocations undertaken after 1988.

3.2.7 Initial release group size

For clarity and consistency regarding the size of initial translocation program release cohorts, I consider releases occurring during a single calendar month as one release. For 51 translocations, either the release month, or year, could not be identified. These translocations have been omitted, as I could not ascertain how many releases occurred over the particular 12 month period. However, those releasing 10 animals or less have been added to the table, as such small founder numbers are unlikely to have been released in two or more events.

3.2.8 Total number of animals released

For some programs, data regarding subsequent releases after known translocations was unavailable, or requests for information denied. Translocations known to have released animals before 2000, and for which papers pertaining to these programs published three or more years later did not mention subsequent releases, were judged to have not augmented the known founder numbers.

3.2.9 Number of release events

A release event is defined as the release of animals on one occasion, or several releases occurring during a single calendar month. For some translocations, data regarding the total number of releases was unavailable, or requests for information denied. Translocations known to have released animals before 2000, and for which papers pertaining to these programs published three or more years later did not mention subsequent releases, were judged to have ceased release activities.

3.2.10 Soft and hard releases - post-release assistance

When considering the use of post-release assistance (termed a ‘soft release’ protocol), only programs translocating animals to the mainland or island environments were considered. Releases to predator-proof enclosures greater than 100 hectares were not included in my study as assistance, usually in the form of supplementary food and water, is often provided for enclosed populations.

3.2.11 Post-release monitoring

My study investigated the use of radio tracking over time as a post-release monitoring tool. As radio tracking evolved as a wildlife research tool in the 1960s (Mech and Barber 2002), I believe that it could be reasonably expected that Australian macropod translocations undertaken from 1970 onwards would have had access to the technique. Considering the 109 Australian macropod translocations investigated in this study, 107 were releases conducted from 1970 onward.

As part of this study, information was sought as to the rate of publication of Australian macropod translocation program results. When undertaking preliminary research into Australian macropod translocations, I obtained information from a variety of sources, which were subsequently placed in two categories:

scientific paper	Independent peer-reviewed journal articles dedicated solely to the presentation of data pertaining to a specific translocation program, and independent peer-reviewed journal articles providing basic details of translocation programs which are not the specific focus of the paper
other	Government and non-government web pages, magazines, media releases, newsletters, books, tourist information brochures, translocation proposals, unpublished reports, protected area management plans, species management/recovery plans, unpublished research theses, conference proceedings

As previously discussed, translocations are long-term programs. Consequently, proponents may be unlikely to publish program details until several years after release. If Short et al's (1992) assessment criteria is adopted, five years of population persistence must elapse (and the population deemed to be likely to continue to persist) before a program can be judged a success. Thus, as the final literature search for scientific papers regarding Australian macropod translocations was conducted in October 2008, only programs which released animals before October 2003 might be reasonably expected to have published data.

3.2.12 Translocation success

Two different success criteria were applied to the Australian macropod translocations studied. First, as the presence of a population at the release site is the fundamental measurement of translocation success, programs were deemed successful if a population was present on 1-1-2007. This simple criterion permitted all translocations, whether historic or recent, to be analysed as a single data set. However, considering the major shortcoming in this measurement, namely that sufficient time must elapse after release to assess whether the population will establish and persist, findings must be considered accordingly. As Short et al. (1992) have produced the most detailed appraisal of

Australia-wide translocations to date, their criteria of success were also applied, where possible, to this study's data set.

3.2.13 Methodological and environmental factors critical to Australian macropod translocation success

Translocations are complex, multi-faceted undertakings, and simplistic assessments as to the cause of success/failure should therefore be avoided (Griffith et al 1989). As theory suggests that translocation outcomes are likely to be determined by a combination of variables, classification tree analyses were performed in an effort to identify factors common to successful Australian macropod translocations. Using this analytical tool also permitted the inclusion of both numerical (for example average rainfall at release site) and categorical (for example source population type) variable responses within the analysis, and could accept the missing response values within the data set (De'ath and Fabricius 2000) which resulted from the lack of available information for some translocation programs.

The following methodological and environmental variables were included in the analyses:

- species translocated
- state/territory where the translocation occurred
- whether the release area was managed by a private or government agency
- whether or not the species fell into the 'critical weight range' theorised by Burbidge and McKenzie (1989) as an indicator of acute susceptibility to threatening processes
- average yearly rainfall at release site
- source population (captive, wild, predator-proof enclosure or combination of these)
- release site (mainland, island, predator-proof enclosure)
- use of acclimatization pens
- use of post-release assistance

- initial release group size
- total release group size
- gender ratio
- use of pre-release site management
- use of post-release site management
- use of post-release monitoring

Two analyses were conducted, the first including all the translocations for which success (population extant 1-1-07) or failure (population extinct 1-1-07) was known. The second included only those programs eligible for assessment against Short et al.'s (1992) success criteria.

3.3 Results

3.3.1 Location of Translocation by State/Territory

I identified a total of 109 Australian macropod translocations conducted between 1969 and 2006, of which 56% were undertaken in Western Australia. The next most prolific state was South Australia, which completed 17% of total translocations. The remaining states/territories carried out less than 10 translocation programs each (figure 3.1). Of the total set of translocations studied, the date of release was known for all but two programs. These data suggest there has been a sharp increase in the number of translocations in the last two decades (Figure 3.1). It is important to note that for some translocations, animals were moved interstate and thus more than one state/territory conservation agency was involved. However, the data only pertains to the state/territory in which the animals were released.

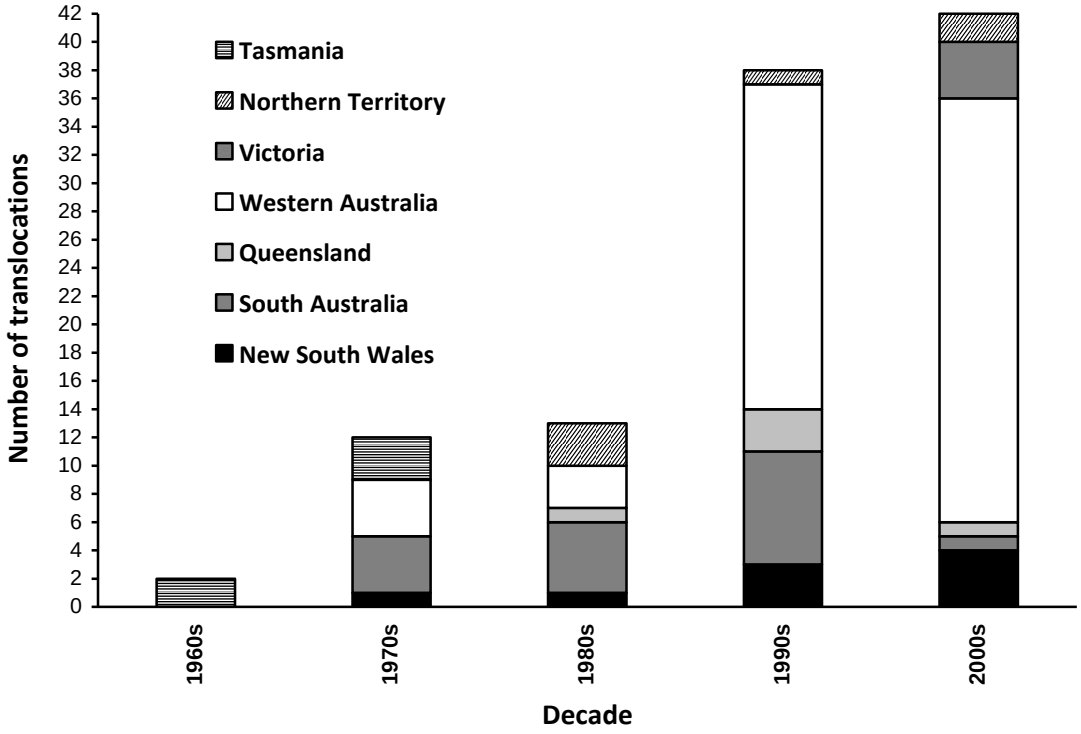

Figure 3.1 Number of Australian macropod translocations by state or territory over the past 5 decades

Translocations were undertaken by either state/territory government conservation departments, or non-government organizations, and in some cases involved collaborations between the two (Table 3.2). A total of 27 translocations across Australia (25%) were to non-government managed release sites. Although the total number of programs was small, such translocations made up a considerable proportion of releases in Victoria (100%), Queensland (40%), and New South Wales (67%). The highest number of releases to non-government managed land were in Western Australia (n = 12, 20%).

Table 3.2 Releases to government and non-government managed release sites

State	Government-managed release site	Non-government managed release site	Total
QLD	3	2	5
NSW (including ACT)	3	6	9
VIC	0	4	4
SA	15	3	18
WA	48	12	60
TAS	5	0	5
NT	6	0	6

Western Australia and South Australia have the highest number of potential species available for translocation, whereas Tasmania and the Northern Territory have the least (Table 3.3). Correlation analysis was performed between the number of translocations undertaken per state/territory and the number of potential species available for translocation per state/territory. The result showed a degree of positive correlation, however this was not statistically significant ($r = 0.6$, $p = 0.15$).

As no species have become extinct in Tasmania, or had their previous ranges substantially reduced, there are no species that require translocation on mainland Tasmania (Table 3.3). The only Tasmanian translocations undertaken during the study period were to offshore islands in the late 1960s / early 1970s, and although these were conducted with the future conservation of the species in mind, the taxa in question have not declined significantly on the Tasmanian mainland.

As identified in Chapter 2, threatened species translocations are expensive undertakings. Therefore, it may be theorised that states/territories with the largest economies may be more likely to conduct translocations. Considering the Gross State Product (a measure of the value added to the Australian economy by economic production in a particular state/territory) per capita for each state and territory (Table 3.4), Western Australia and the Northern Territory have the largest economies.

Table 3.3 Number of potential species available for translocation per state

	Number of species present on Australian/ Tasmanian mainland 2008	Number of species that have gone nationally extinct since colonisation	Number of species previously occurring on the mainland but present only on offshore islands 2008	Number of species that have gone extinct within the State since colonisation but still extant elsewhere	Number of species which have experienced mainland range decline since colonisation	Number of species potentially available for translocation
Queensland	31	1	0	1	7	8
New South Wales (including ACT)	15	1	0	6	3	9
Victoria	9	2	0	5	2	7
South Australia	8	5	1	7	4	12
Western Australia	17	3	3	0	9	12
Tasmania	5	0	0	0	0	0
Northern Territory	10	2	0	3	2	5

Van Dyck and Strahan (2008)
*translocated populations are not included in above table

Table 3.4 Gross State Product per state/territory

	Population 2007	**Gross State Product 2006-7 ($m)****	**Gross State Product per capita**
Queensland	4,228,300	187,339	44,305
New South Wales (including ACT)	7, 267,800	342,310	47,100
Victoria	5,246,100	242,595	46,243
South Australia	1,591,900	65,676	41,256
Western Australia	2,130,800	127,775	59,966
Tasmania	495,800	19,239	41,842
Northern Territory	217,600	13,405	61,604

*Australian Bureau of Statistics (2008)
**Australian Bureau of Statistics (2007)

Correlation analysis was performed between the Gross State Product of each state/territory and the number of translocations undertaken. The result showed little correlation between the two variables ($r = 0.4$, $p = 0.37$).

3.3.2 Species chosen for translocation

20 different macropod species were translocated for conservation objectives in Australia over the period 1969-2006 (Table 3.5). Clearly, the most commonly translocated species was *B. penicillata*, releases of which accounted for 40% of the total translocations investigated (n = 109; Table 3.5). The majority of *B. penicillata* releases were in Western Australia (n = 30, 68%), followed by South Australia (n =11, 25%), and New South Wales (n=3, 7%). *Macropus eugenii* was the second most frequently translocated macropod. Of the other species, seven were released on only one occasion during the study period (Table 3.5).

Table 3.5 Australian macropod species translocated 1969-2006

Group and Species	Number of translocations	% of total translocations
Bettongs and Potoroos		
Bettongia penicillata woylie	44	40
Bettongia lesueur burrowing bettong	8	7
Bettongia gaimardi Tasmanian bettong	1	1
Potorous tridactylus long-nosed potoroo	2	2
Potorous gilbertii Gilbert's potoroo	1	1
Aepyprymnus rufescens rufous bettong	1	1
Kangaroos and Wallabies		
Macropus eugenii tammar wallaby	14	13
Macropus giganteus eastern grey kangaroo	1	1
Macropus parma parma wallaby	2	2
Macropus rufogriseus red-necked wallaby	1	1
Rock Wallabies		
Petrogale lateralis black-footed rock-wallaby	8	7
Petrogale xanthopus yellow-footed rock-wallaby	3	3
Petrogale rothschildi Rothschild's rock-wallaby	1	1
Petrogale persephone Proserpine rock-wallaby	1	1
Petrogale penicillata brush-tailed rock-wallaby	1	1
Hare-wallabies		
Lagorchestes hirsutus mala	9	8
Lagostrophus fasciatus banded hare-wallaby	3	3
Nailtail Wallabies		
Onychogalea fraenata bridled nailtail wallaby	4	4
Quokka		
Setonix brachyurus Quokka	2	2
Pademelons		
Thylogale billardierii Tasmanian pademelon	2	2

3.3.3 Use of captive bred animals for translocation

Half of all Australian macropod releases sourced animals from the wild, which is considerably more than the next most common source of founders (captive bred animals 28%; Table 3.6).

Table 3.6 Source of animals – Australian macropod translocations

Source	**All translocations**	**Reintroductions only**	**Introductions only**	**Augmentation only**
Wild	54 (50%)	46 (52%)	7 (35%)	1 (100%)
Captive bred	32 (29%)	24 (27%)	8 (40%)	0
Predator-proof enclosure > 100ha	7 (6%)	5 (6%)	2 (10%)	0
Combinations of two sources	10 (9%)	9 (10%)	1 (5%)	0
Unknown	6 (6%)	4 (5%)	2 (10%)	0
Total	109	88	20	1

The data appear to suggest a different preference for the source of animals for releases to the mainland than for translocations to islands or predator-proof enclosures (Table 3.7). For reintroductions, introductions and the single augmentation to the mainland (n = 66), 65% released animals sourced from wild populations. Translocations to islands or predator-proof enclosures (n = 43) used wild animals in only 26% of programs (Table 3.7).

Table 3.7 Source of animals according to release site – Australian macropod translocations

Source	Reintroductions to the mainland	Reintroductions to islands	Reintroductions to predator-proof enclosures >100ha	Introduction to the mainland
Wild	41 (64%)	1	4 (22%)	1
Captive bred	11 (17%)	4	9 (50%)	0
Predator-proof enclosure > 100ha	4 (6%)	0	1 (6%)	0
Combination of two sources	4 (6%)	1	4 (22%)	0
Unknown	4 (6%)	0	0	0
Total	64	6	18	1

Source	Introduction to the islands	Introductions to predator-proof enclosures >100ha	Augmentation to the mainland
Wild	6 (38%)	0	1
Captive bred	5 (31%)	3	0
Predator-proof enclosure > 100ha	2 (13%)	0	0
Combination of two sources	1 (6%)	0	0
Unknown	2 (13%)	0	0
Total	16	3	1

Of the 109 translocations studied, sufficient data to identify whether pre-release training would have been considered was available for 89 programs. Using the criteria in Table 3.1 above, 70 releases would not have been expected to consider pre-release training. Of the remaining 19 programs, only two used any form of pre-release training. Both of these releases involved the same species, and were releases from captive breeding colonies to the wild. The training was 'passive' in nature, and involved encouraging foraging behaviour by adjusting food provision techniques and introducing food plants found within the release site.

3.3.4 Release area

For the 109 Australian macropod translocations investigated, 88 were reintroductions, 20 were introductions and one was an augmentation of an existing population. Data regarding whether or not pre-release site management was used were available for 51 programs (releases to predator-proof enclosures were not considered). Of these translocations, 46 used some form of pre-release site management (Table 3.8). When considering all translocations, the most popular form of pre-release site management was attempts to control introduced predators (85%). The clear trend towards the use of pre-release site management in macropod translocation programs is even more pronounced when considering only reintroductions to the mainland. For this type of release, all but one program used some form of pre-release site management (Table 3.8). However, a different pattern emerges when considering introductions to offshore islands. Although only a small data set of known use or otherwise of pre-release site management could be obtained (n = 10), only two of these programs employed pre-release site management (Table 3.8).

Data regarding the use of post-release site management were available for 46 Australian macropod translocations. Of these programs, 38 used some form of site management after release (Table 3.9). Similar to the trend observed for pre-release site management, the most frequently used management after release was the control of introduced predators. Introduced competitor control was the next most frequently used form of management after release, undertaken by 16% of programs (Table 3.9). The popularity of introduced predator control work continues when only reintroductions to the mainland are considered, where all programs attempted to control these species (Table 3.9). Less than half of translocations to islands used any form of post-release site management.

Table 3.8 Types of pre-release site management, and the type/number of release sites employing particular techniques – Australian macropod translocations

Types of pre-release site management	All translocations	Reintroductions to the mainland	Reintroductions to islands	Introductions to the mainland	Introductions to islands	Augmentation to the mainland
Introduced predator control*	35 (85%)	32 (91%)	2 (50%)	0	1 (50%)	0
Introduced predator monitoring	1 (3%)	1 (3%)	0	0	0	0
Introduced competitor control**	15 (37%)	10 (29%)	4 (100%)	0	1 (50%)	0
Native predator control	5 (13%)	5 (15%)	0	0	0	0
Fencing particular release site boundaries	7 (18%)	7 (21%)	0	0	0	0
Introduced plant control	2 (5%)	1 (3%)	0	0	0	0
Soil erosion control	1 (3%)	1 (3%)	0	0	0	0
Creation of firebreaks***	4 (10%)	4 (11%)	0	0	0	0
Burning to create suitable habitat	2 (5%)	2 (5%)	0	0	0	0
Removal of domestic stock	7 (18%)	5 (15%)	2 (50%)	0	0	0
Rehabilitation of mine site	1 (3%)	1 (3%)	0	0	0	0
No pre-release site management employed	5	1	2	0	2	0
Total number of programs for which data available	51 of 109	36 of 64	6 of 6	0 of 1	10 of 16	0 of 1

* includes shooting, bating and trapping
** includes shooting, baiting, trapping and mustering
*** includes burning and clearing

Table 3.9 Types of post-release site management, and the type/number of release sites employing particular techniques – Australian macropod translocations

Types of pre-release site management	All translocations	Reintroductions to the mainland	Reintroductions to islands	Introductions to the mainland	Introductions to islands	Augmentation to the mainland
Introduce predator control*	32 (84%)	32 (100%)	0	0	0	0
Introduced/native predator monitoring	2 (5%)	3 (9%)	0	0	0	0
Native predator control**	4 (11%)	4 (13%)	0	0	0	0
Introduced competitor control***	6 (16%)	5 (16%)	0	0	1 (25%)	0
Introduced competitor control outside release site	1 (3%)	0	0	0	0	0
Introduced plant control	1 (3%)	0	1 (50%)	0	0	0
Creation of firebreaks, burning to create suitable habitat****	6 (13%)	4 (13%)	1 (50%)	0	1 (25%)	0
Removal of domestic stock	2 (5%)	1 (3%)	0	0	1 (25%)	0
Culling of translocated species	1 (3%)	0	0	0	1 (25%)	0
Grazing of domestic stock to improve habitat	1 (3%)	0	0	0	0	0
Increasing road signage to minimize road fatalities	1 (3%)	0	0	0	0	0
No post-release site management employed	8	0	3	0	5	0
Total number of programs for which data available	46 of 109	32 of 64	5 of 6	0 of 1	9 of 16	0 of 1

* includes shooting, bating and trapping
** includes baiting and shooting
*** includes baiting and warren destruction
**** includes burning and clearing

3.3.5 Use of Population Viability Analysis

Of the 109 translocations studied, 83 were known to have commenced after 1988 (the release date for two translocations was unavailable). Of these, data regarding whether or not PVA was used was available for 35 programs. PVA was completed for three of these translocations, one each in 1993, 1994 and 2004.

3.3.6 Release methodology and founder group composition

Considerably more Australian macropod translocation programs released between six and 10 animals initially than any other release size category (Figure 3.2). Overall, the majority of programs (64%) release between one and 15 animals (Figure 3.2).

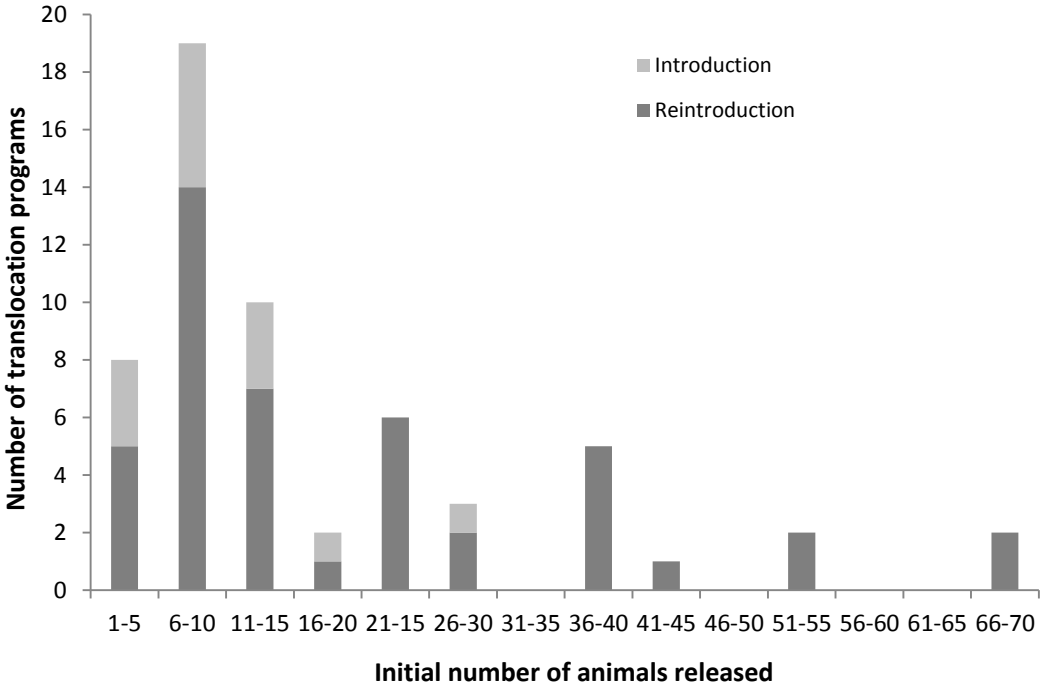

Figure 3.2 Size of initial release cohort - Australian macropod translocations

When considering both overall translocations and only reintroductions, the most frequently released total number of animals was between 36 and 40 individuals (Figure 3.3). However, in the case of introductions, releases of between 11 and 15 animals were the most common.

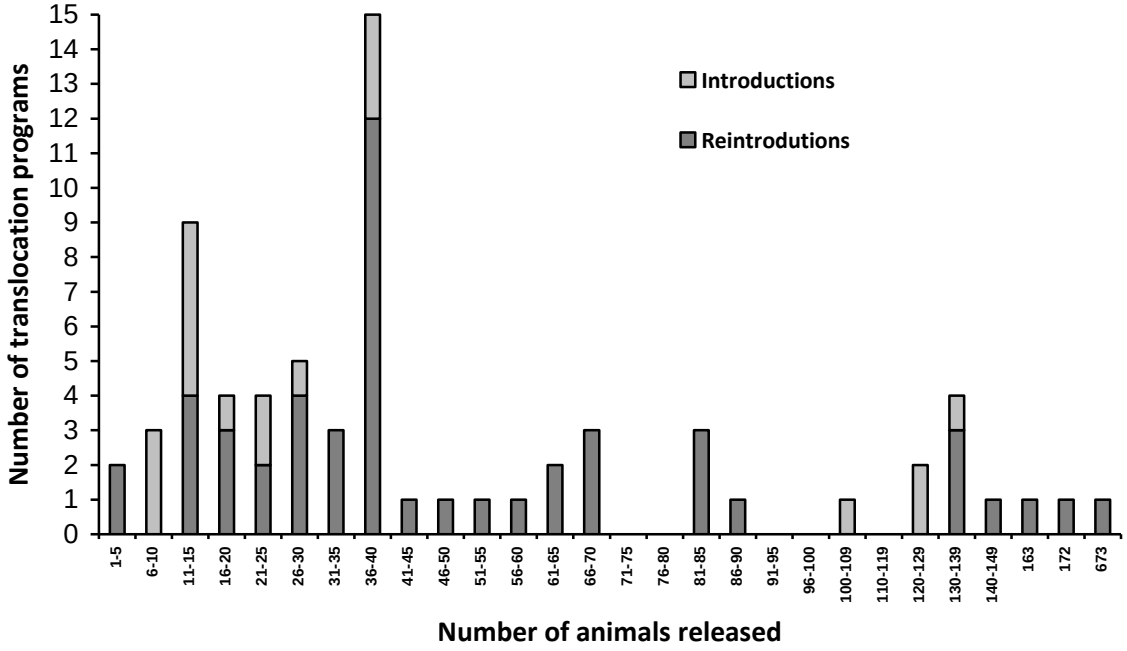

Figure 3.3 Total number of animals released – Australian macropod translocations

Data concerning the gender ratio of the initial release group was available for 46 programs. Of these, 40 programs released both males and females initially, four released only males and two only females.

Data regarding the gender composition of total released animals was available for 53 Australian macropod translocation programs. Of these, 17 (32%) released more males than females, 33 (62%) released more females than males, and 3 (6%) released an equal gender ratio. No programs translocated a single-sex release cohort. Table 3.10 provides data as to the ratio of male to female animals released.

Information was sought as to the age of animals released in Australian macropod translocation programs. Of all the translocation variables investigated, data regarding the age of released animals was the most difficult to obtain. Of the 109 translocations investigated, data regarding the age of released animals was available for 45 programs. Of these 45 programs, 19 stated that only adults were released, 20 that both adults and

Table 3.10 Ratio of males to females released (n = 53) – Australian macropod translocations

Ratio M:F	Number of translocations
0.00 to 0.24	1
0.25 to 0.49	7
0.50 to 0.74	12
0.75 to 0.99	13
1.00	3
1.01 to 1.24	6
1.25 to 1.49	7
1.50 to 1.74	2
1.75 to 1.99	1
2	1

sub-adults were released, and 16 reported females carrying pouch young. No translocations reported releasing only sub-adults.

Only two Australian macropod translocations (both of *Petrogale xanthopus*) were found to have considered social grouping important within the release group structure. As part of these translocations, animals were released in social groups which had been previously established within captive breeding colonies.

Data concerning the number of release events undertaken during the release phase was available for 49 translocations. More translocation programs released all animals in a single release event than any other number of release events (Figure 3.4). Releasing animals on two occasions was the next most popular category.

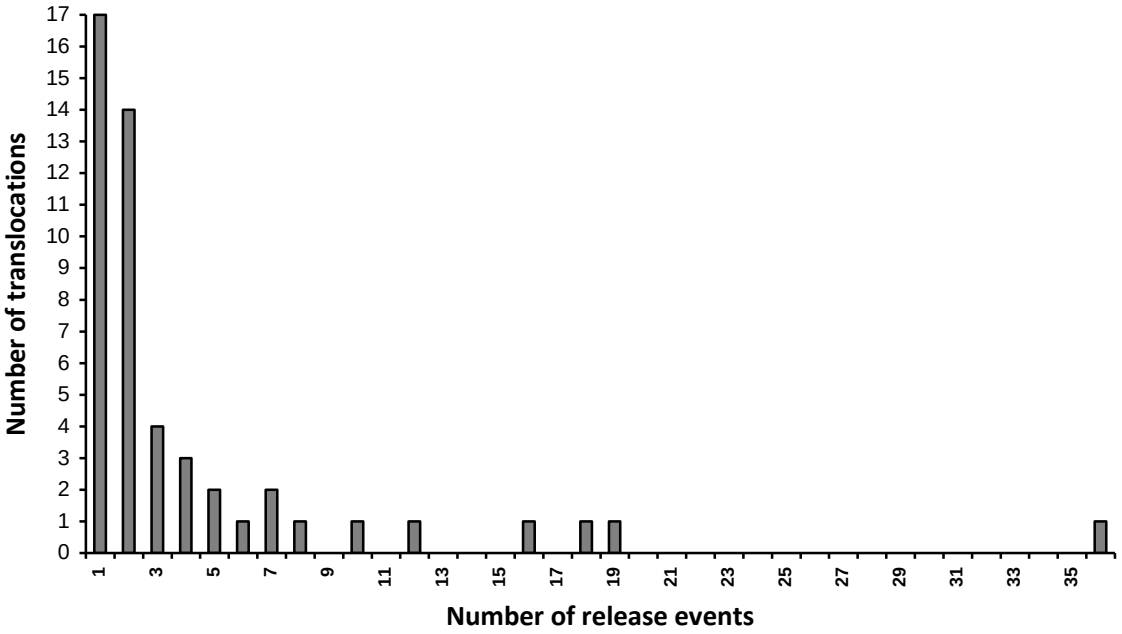

Figure 3.4 Number of release events – Australian macropod translocations

Information regarding the month of release could be obtained for 33 translocation programs, totaling 159 release events. September and December were the most popular months for Australian macropod translocations (Figure 3.5). The pattern of monthly releases is spread very evenly across the other months with the exceptions of January and February (Figure 3.5).

Of the 109 Australian macropod translocations investigated, data pertaining to the use of acclimatization pens was obtained for 44 programs (translocations to predator-proof enclosures of 100 hectares or greater were not considered). Of these, 16 housed animals in acclimatization pens, whilst 28 did not. Of the 16 translocation programs which initially released animals into acclimatization pens, 14 were reintroductions and two were introductions. The most frequent length of time spent by animals in acclimatization pens was 7-14 days (Table 3.11). The popularity of using acclimatization pens in Australian macropod translocations has generally increased over chronological time (Figure 3.6). Those programs which did not use acclimatization pens for all releases are listed as per the first release date that pens were used.

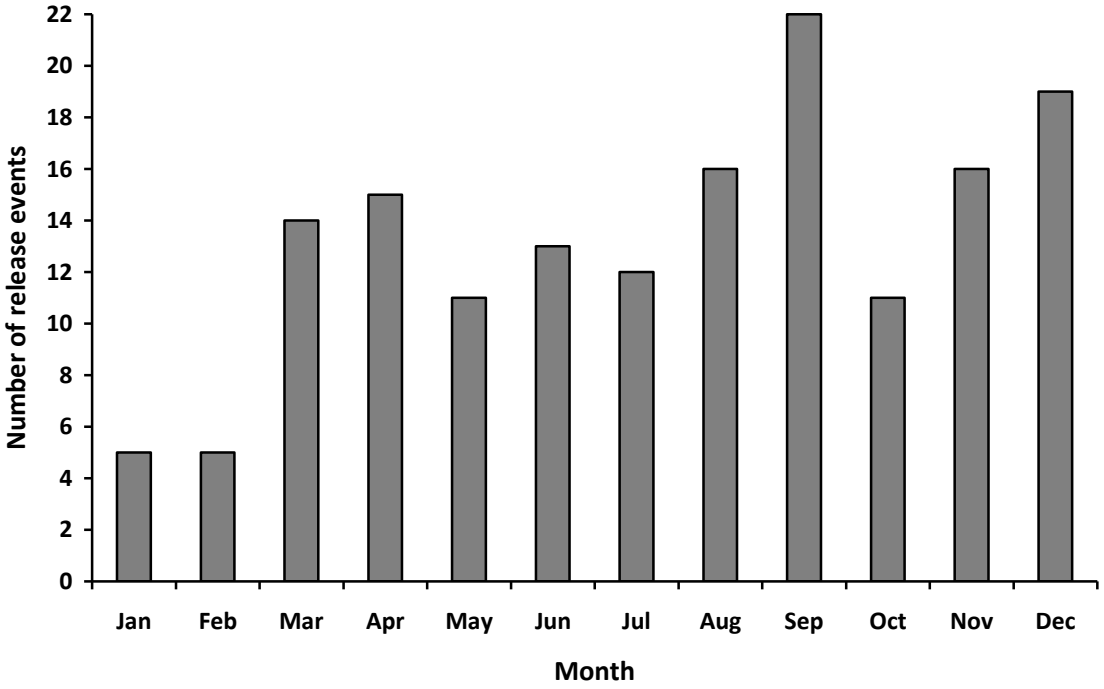

Figure 3.5 Number of release events by month – Australian macropod translocations

Table 3.11 Length of time spent in acclimatization pens

Length of time spent in acclimatization pen	Number of translocation programs
1 day	1
2 days	1
7-14 days	7
28 days	2
42 days	1
3 months	1
unknown	3

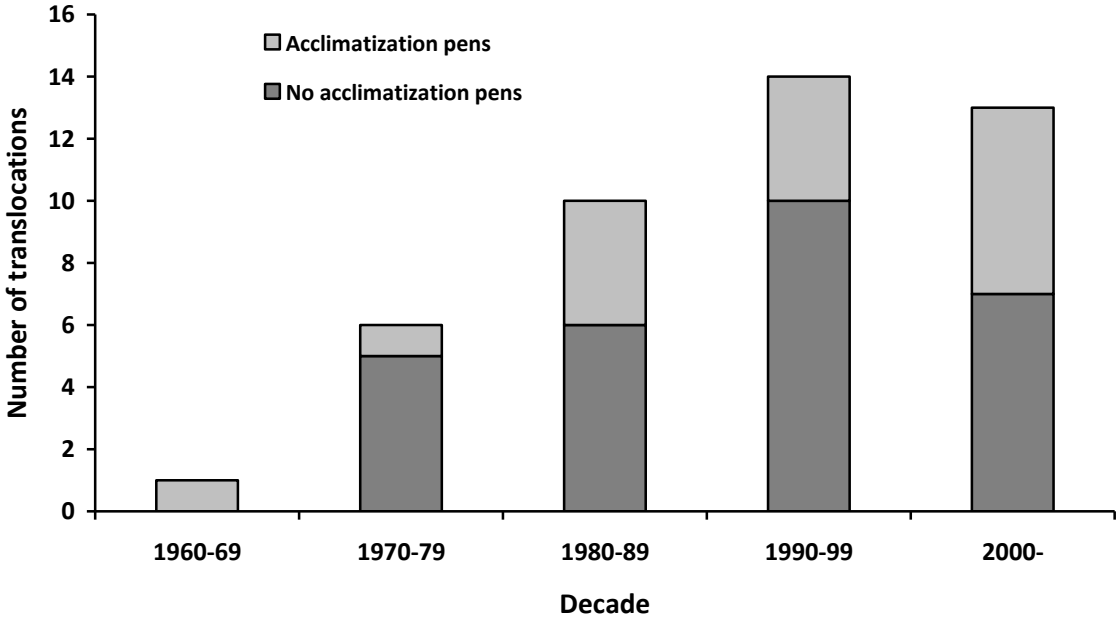

Figure 3.6 Use of acclimatization pens over time – Australian macropod translocations

Four translocations adjusted their use of acclimatization pens over the course of the project. Of these, one program abandoned the use of pens after the first release, two translocations used the pens for some releases and not others, and the fourth used the technique for the final of four releases. Of the 88 translocations which released animals to the mainland or island environments, 18 (13 reintroductions and five introductions) provided some form of post-release assistance, 24 did not, and information for the remaining 46 programs could not be obtained. The most frequently used form of post-release assistance was the provision of food and water (Table 3.12).

Table 3.12 Types of post-release assistance provided

Type of assistance	Number of translocations
Food	2
Water	1
Food and water	12
Artificial shelter	1
Food, water, artificial shelter	1
Unspecified	1

A change in the level of assistance provided after release was identified for four translocations. For two of these programs, a hard release protocol was followed initially, however food and water respectively were provided for animals in subsequent releases. Another program initially provided assistance to animals, however a hard release method was adopted for additional releases. In the final case, food, water and artificial shelter were supplied in conjunction with the first release, but water was not provided for subsequent releases.

A decrease in the popularity of soft release protocols occurred during the 1990s, followed by an increase from 2000 onwards (Figure 3.7). Those programs which did not use post-release support for all releases are listed as per the first release date that such support was provided.

Translocation managers which released animals to islands provided post-release support more often (proportionally) than did proponents releasing individuals to the mainland (Figure 3.8).

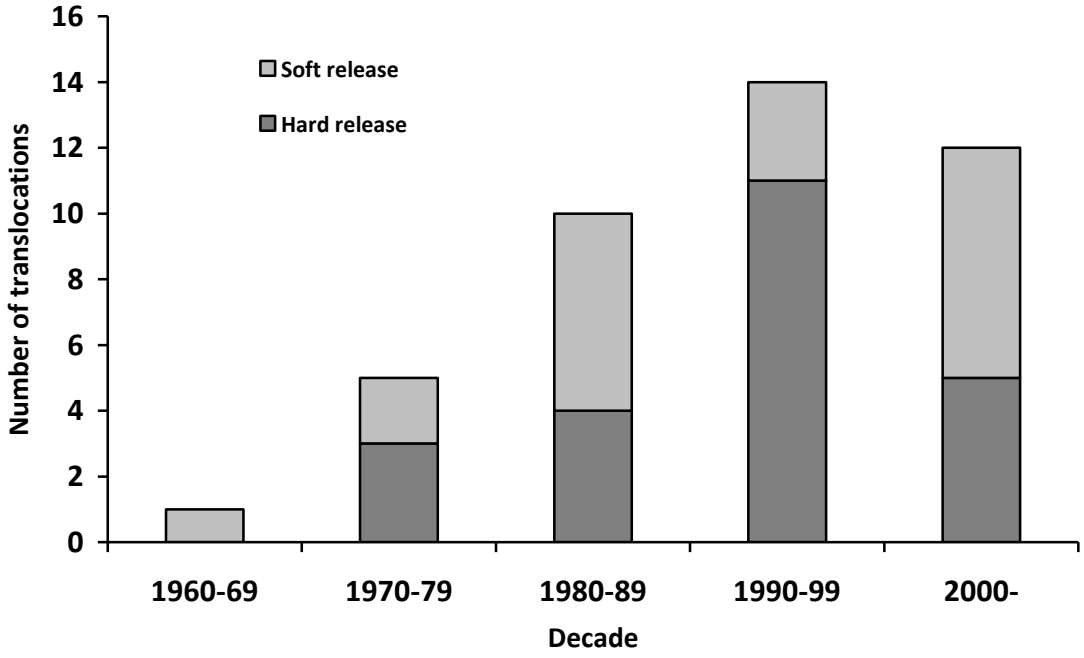

Figure 3.7 Use of soft/hard release protocols over time - Australian macropod translocations

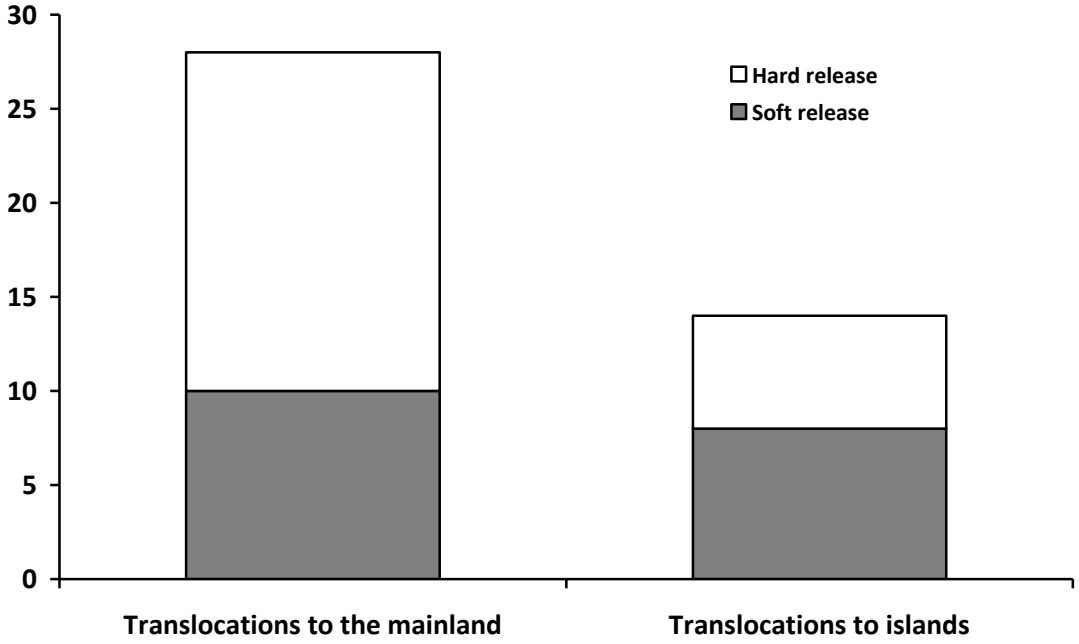

Figure 3.8 Release protocol and release site environment - Australian macropod translocations

3.3.7 Post release monitoring

Of the 109 Australian macropod translocations investigated, data regarding whether or not post-release monitoring had been undertaken was obtained for 88 programs. All of these translocations completed some form of monitoring after release.

The following post-release monitoring methods were used:

- trapping
- radio tracking
- spotlighting
- search for sign
- autopsy
- video surveillance
- scat analysis
- remote photography

Of the 88 translocations which used post-release monitoring, details regarding the technique(s) employed were available for 55 programs. Most programs used two or more methods. Overall, eight forms of post-release monitoring were reported to have been used, with trapping being the most popular (75% of translocations; Table 3.13), followed by radio tracking (68%). Of the remaining techniques, spotlighting (23%) and search for sign (18%) were the only other forms of post-release monitoring used by more than three programs (Table 3.13).

When only data pertaining to reintroductions is considered, the popularity of trapping and radio tracking is again clear (Table 3.13). Further, trapping is the most frequently used technique when regarding only the small data set relating to introductions (n = 10), however radio tracking was employed in only one program. Only two Australian macropod translocations used video surveillance and remote photography (Table 3.13).

Table 3.13 Frequency of use of post-release monitoring techniques

	Overall translocations (n =56)	**Reintroductions (n=46)**	**Introductions (n=10)**
Trapping	42 (75%)	33 (72%)	10 (100%)
Radio tracking	38 (68%)	37 (80%)	1 (10%)
Spotlighting	13 (23%)	13 (28%)	0
Search for sign	10 (18%)	9 (20%)	1 (10%)
Video surveillance	1 (2%)	1 (2%)	0
Remote photography	1 (2%)	0	1 (10%)
Scat analysis	1 (2%)	1 (2%)	0
Autopsy	3 (5%)	3 (7%)	0

It appears that radio tracking has been more frequently used to monitor Australian macropod translocations in recent times (table 3.14).

Table 3.14 Use of radio tracking over time

Date of first release	**Number of translocations**	**Number of translocations for which data regarding the type of post-release monitoring was available**	**Number of translocations using radio tracking**
1970-74	8	3	0
1975-79	4	3	1 (33%)
1980-84	8	6	1 (17%)
1985-89	5	4	4 (100%)
1990-94	11	9	6 (67%)
1995-99	27	11	11 (100%)
2000-2004	40	15	13 (87%)
2005	2	2	2 (100%)

Considering the publication of program results for the 109 Australian macropod translocations researched, data was available in scientific journals for 36 programs (33%). Of these, eight (7% of total translocations) were papers solely dedicated to the detailed presentation of data pertaining to a specific translocation program. Data regarding the remaining 73 programs (67%) was found via 'other' sources as defined in 'methods' above.

A total of 97 translocation programs were undertaken before October 2003, allowing sufficient time for Short et al.'s (1992) success criteria to be applied. Of these, 34 (35%) published scientific papers, eight (8%) of which were dedicated to a particular translocation. Therefore, even if a translocation proponent is not expected to publish a paper within five years of release, this does not affect the overall rate of publishing calculated for all 109 translocations.

3.3.8 Translocation success

For the 109 translocations studied, data regarding population presence/absence was available for 72 programs. Of these, 44 (61%) translocation programs had a population present on 1-1-07. Of the 44 translocation programs which had a population extant at 1-1-07, 29 were considered successful against Short et al.'s (1992) benchmark, whilst 15 programs could not be assessed due to lack of data or insufficient elapsed time since release.

3.3.9 Suspected / known causes of translocation failure

Of the 109 translocations studied, data regarding the presence/absence of a population at the release site at 1-1-07 was obtained for 72 programs. Of these, 28 translocations were unable to establish viable populations, with the suspected/known causes of failure available for 24 programs (Figure 3.9). Predation by introduced species was listed as the sole, or a contributing, cause of program failure by 17 (71%) of these translocations. For those translocations where a factor other than predation by introduced species was believed/known to have been responsible for program failure, cat/foxes were not present at the release site.

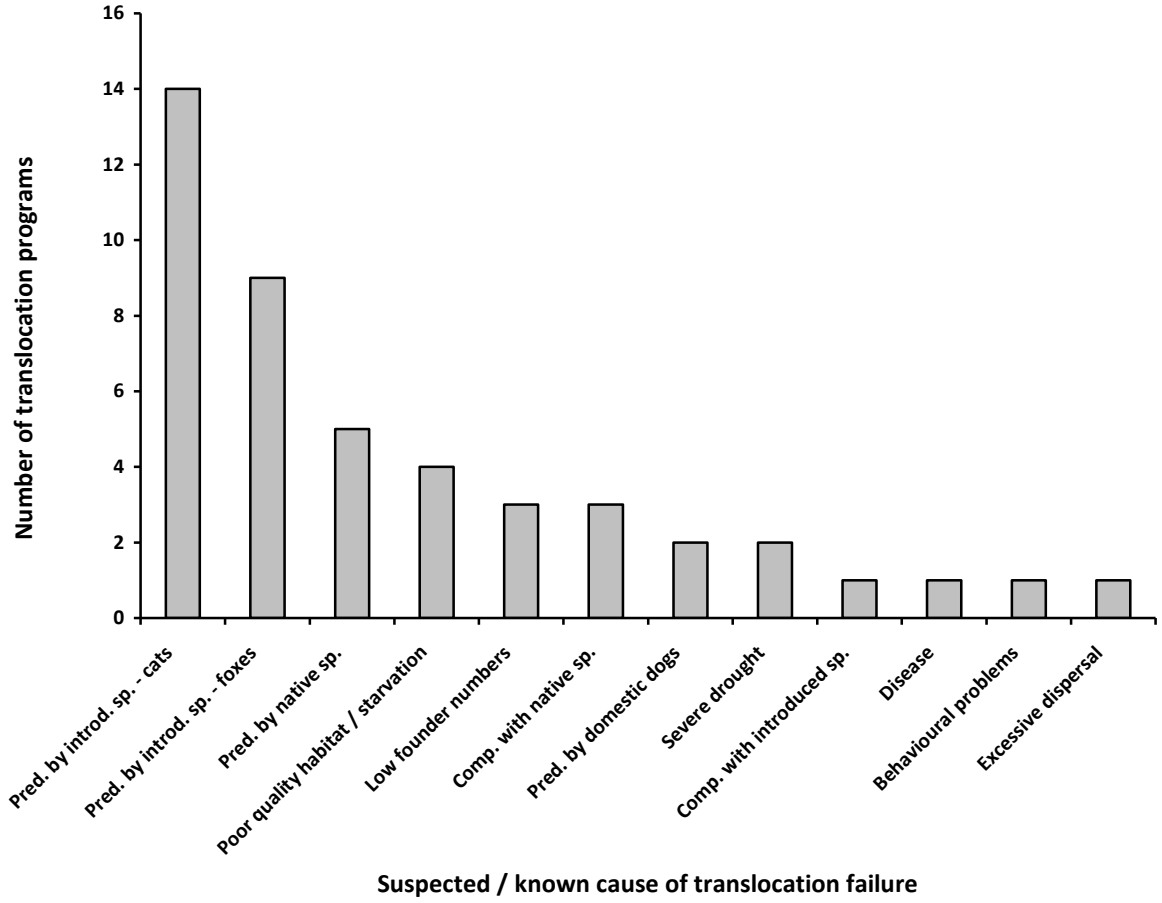

Figure 3.9 Suspected / known causes of translocation failure - Australian macropod translocations. Programs which listed 'introduced predators' as known/suspected causes were added to both cat and fox columns. Some programs listed more than one suspected/known cause(s)

Considering the 24 translocations for which the suspected/known causes of the failure could be identified, 23 of these were reintroductions. When considering only reintroductions to the mainland (n = 15), predation by introduced species was again the most dominant suspected/known (sole and contributing) cause of translocation failure (80%; Table 3.15). Only two failed reintroductions to islands were identified, one suspected/known to have failed due to predation by pythons and competition with brown bandicoots, and the other from predation by domestic dogs (Table 3.15). A single failed introduction was also identified, with cat predation known/suspected to have thwarted the translocation attempt. Considering only the six failed reintroductions to predator-proof enclosures greater than 100 hectares, predation by introduced species once again appears

to be the greatest contributor to program failure (Table 3.15). Native competitors and poor quality habitat/starvation were also identified as significant known/suspected causes of reintroduction failure.

Table 3.15 Suspected/known causes of reintroduction failure. Suspected/known cause(s) of failure is displayed in the left column, and the number of translocations that listed that particular cause in the right columns. Programs which listed 'predation by introduced species' as known/suspected causes were added to both 'cat' and 'fox' columns. Some programs listed more than one suspected/known cause

Suspected/known cause of translocation failure	**Number of translocations that included this factor as a known/suspected cause**		
	Reintroduction to the mainland (n=15)	**Reintroduction to islands** (n=2)	**Reintroduction to predator-proof enclosure ≥100ha** (n=6)
Predation by introduced species – fox	4 (27%)	0	5 (83%)
Predation by introduced species – cat	7 (47%)	0	5 (83%)
Domestic dog predation	1 (7%)	1 (50%)	0
Low founder numbers	3 (20%)	0	0
Severe drought	2 (13%)	0	0
Native predation	4 (27%)	1 (50%)	0
Native competition	0	1 (50%)	2 (33%)
Competition with introduced species	0	0	1 (17%)
Poor quality habitat/starvation	1 (7%)	0	3 (50%)
Disease	0	0	1 (17%)
Behavioural problems	1 (7%)	0	0
Dispersal	1 (7%)	0	0

Data regarding the primary threat(s) to translocation success identified by managers before release could be obtained for 12 reintroductions to the mainland (Table 3.16) and five releases to predator-proof enclosures (Table 3.17). Of these, 14 correctly identified

prior to release the threat (introduced predators) which would ultimately cause the translocation to fail (Table 3.17).

Table 3.16 Reintroductions to the mainland - primary identified threat(s) to translocation success and suspected/known cause(s) of translocation failure. Each row represents an individual translocation program

Primary identified threat(s) to translocation success	**Suspected/known cause(s) of translocation failure**
Fox predation	Fox predation
Fox predation	Domestic dog predation
Dingo predation	Dingo and cat predation, starvation, dispersal
Fox predation, rabbit competition	Cat predation
Predation by introduced species, poor quality habitat	Cat predation
Fox and cat predation, poor quality habitat	Cat predation
Predation by introduced species	Wedge-tailed eagle and fox predation, behavioural problems
Poor quality habitat, predation by introduced species, issues with radio collars	Cat predation
Poor quality habitat, predation by introduced species, issues with radio collars	Cat predation
Predation by introduced species	Predation by introduced species
Predation	Fox predation
Unknown	Low founder numbers
Unknown	Low founder numbers, native predation, drought
Unknown	Low founder numbers, native predation, drought
Predation by introduced species	Cat

Table 3.17 Reintroductions to predator-proof enclosures 100 hectares or greater - primary identified threat(s) to translocation success and suspected/known cause(s) of translocation failure. Each row represents an individual translocation program

Primary identified threat(s) to translocation success	**Suspected/known cause(s) of translocation failure**
Predation by introduced species, rabbit competition	Other macropod and rabbit competition, predation by introduced species including cat, poor quality habitat
unknown	Fox and cat predation, poor quality habitat
Predation	Predation by introduced species
Fox and cat predation	Fox and possibly cat predation
Predation by introduced species	Poor quality habitat, disease, predation by introduced species including fox
Fox and cat predation	macropod competition

When considering individual taxa, it appears that particular species, in certain parts of Australia, are succumbing to the same threats over many years of translocation attempts. Two efforts to translocate *B. penicillata* in the 1980s failed due to an introduced predator (fox), and similarly attempts made between 1998 and 2001 also failed due to this threat (foxes and cats; Table 3.18).

Table 3.18 Translocations of *Bettongia penicillata* - known/suspected cause(s) of translocation failure

Year(s) of release(s)	**Translocations of *Bettongia penicillata***
1979	Domestic dog predation
1980-82	Fox predation
1981-87	Native predation/competition
1982	Fox predation
1996	Low founder numbers
1997-98	Low founder numbers, severe drought, native predation
1998-99	Fox and cat predation
1999-01	Cat predation
2001	Cat predation

When considering *L. hirsutus*, all four translocation programs undertaken between 1984 and 2001 failed due to introduced predator impact (Table 3.19). Endeavors to reintroduce *B. lesueur* between 1992 and 2005 met with a similar fate (Table 3.20).

Table 3.19 Translocations of *Lagochestes hirsutus* - known/suspected cause(s) of translocation failure

Year(s) of release(s)	**Translocations of *L. hirsutus***
1984-85	Cat predation, dingo predation, starvation, dispersal
1989-1995	Predation by introduced species
1990-92	Cat predation
2001	Cat predation

Table 3.20 Translocations of *Bettongia lesueur* - known/suspected cause of translocation failure

Year(s) of release(s)	**Translocations of *B. lesueur***
1992	Cat predation
1995-2000	Poor quality habitat, disease, fox predation
2004-5	Native predation, fox predation, behavioural problems

3.3.10 Methodological and environmental factors critical to Australian macropod translocation success

Classification tree analysis performed on the data set comprising all programs for which a population was extant on January 1 2007 initially split the translocations by species (Figure 3.10). The 'branch' to the right included 11 species which were translocated a total of 17 times. Of these species, *Petrogale lateralis* was the only taxon to report a failed translocation (two successful translocations from three attempts). A lower success rate was achieved for those species included in the left 'branch' of the tree (Figure 3.10). Of all the releases of these eight species (n = 55), only 51% were successful. The analysis further split these 55 releases by their use of acclimatization pens. Only 21% of those programs which used acclimatization pens succeeded, compared to 67% of those which did not use pens (Figure 3.10). The releases which did not use pens were

subsequently divided by the mean annual rainfall at the release site. All releases to regions receiving less than 232 millimetres of rain still had a population extant on 1-1-07, whereas only 59% of translocations to areas receiving higher rainfall were successful (Figure 3.10). Considering only those programs releasing animals to areas of greater than 232 mm annual rainfall, releases to Western Australia and the Northern Territory had a higher success rate (75%) than those to either New South Wales or South Australia,

When only those programs eligible for assessment against Short et al.'s (1992) criteria were analysed, the first split was also by species (Figure 3.11). The right 'branch' comprised 10 species which were each successfully translocated on one occasion. The remaining nine species, involved in 47 releases, recorded a 40% program success rate (Figure 3.11). As in the first classification tree analysis, the data was further split by the use of acclimatization pens. Of the programs which used acclimatization pens, only 15% were successful, whereas those which didn't use pens had a 50% success rate (Figure 3.11). Considering the releases that didn't use pens, a further branching by species occurred. Releases of *Bettongia leseuer*, *Lagochestes hirsutus* and *Petrogale lateralis* had a higher success rate (67%) than releases of *Bettongia penicillata*, *Macropus eugenii*, *Onychogalea fraenata* and *Setonix brachyurus* (44%; Figure 3.11). Of the more successful species, two of the releases were to predator-proof enclosures, two onto islands, and the remaining programs to mainland WA where introduced predator control has been very successful. Of the species which experienced lower success rates, 50% of the successful releases of these species were to introduced predator-free islands or predator-proof enclosures, and 67% of the mainland releases were to Western Australia. These species further split into source of animals, with programs releasing animals from captive breeding facilities proportionally more successful (71%) than those using animals from other sources (33%; Figure 3.11).

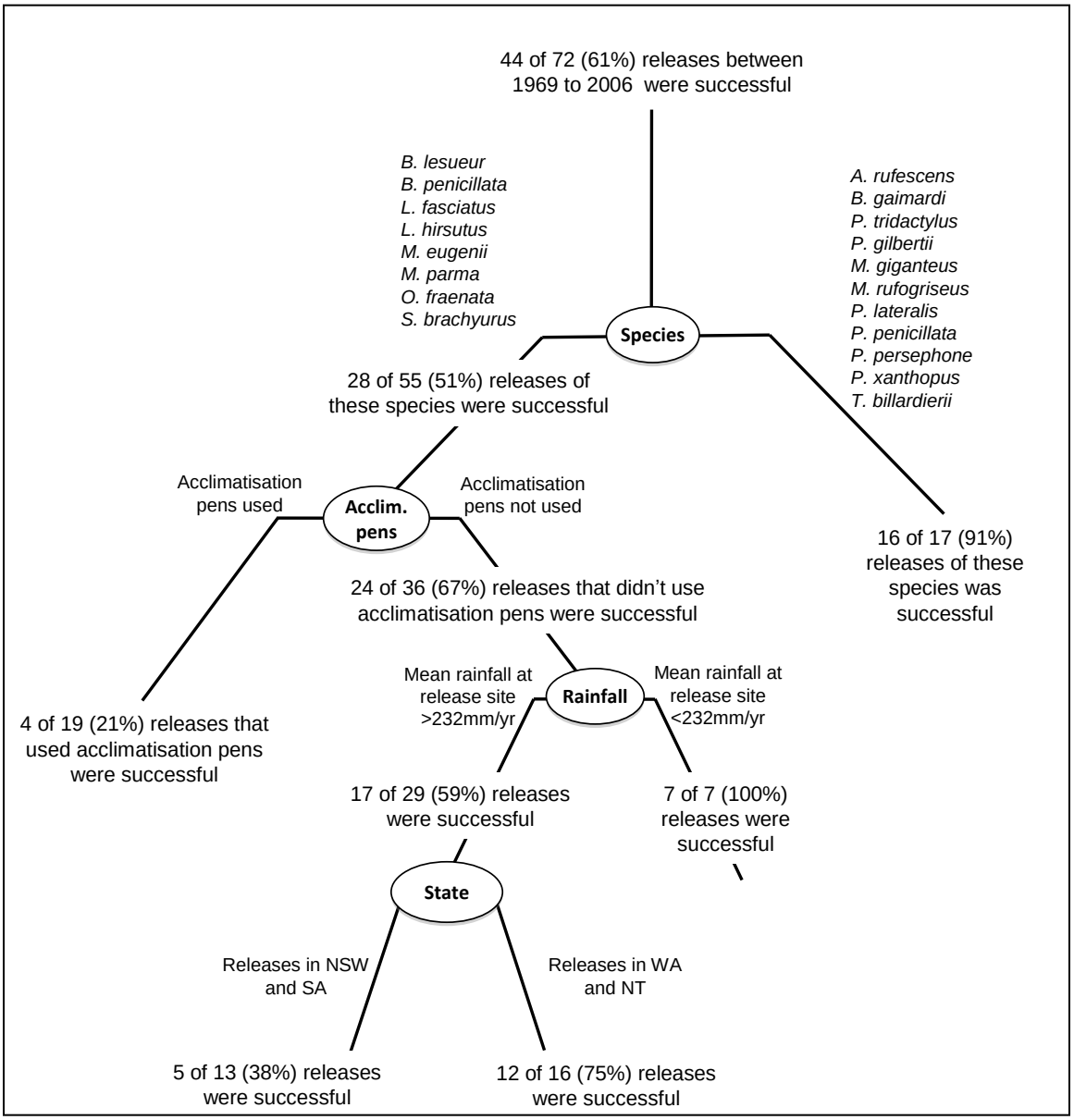

Figure 3.10 Results of classification tree analysis - all Australian macropod translocations programs for which a population was extant 1-1-07

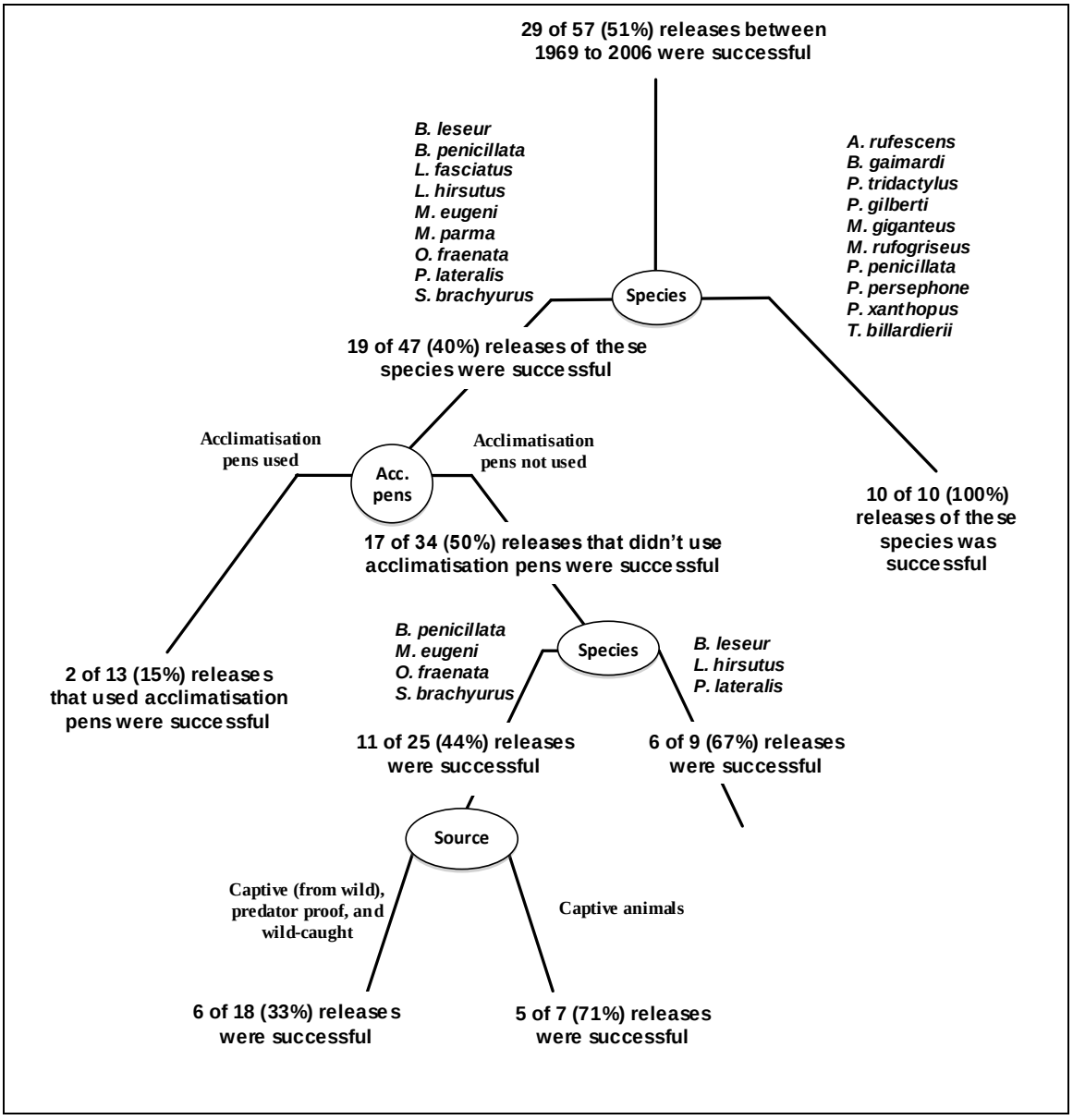

Figure 3.11 Results of classification tree analysis – all Australian macropod translocations programs eligible for assessment against Short et al.'s (1992) criteria

3.4 Discussion

3.4.1 Location of Translocations by State/Territory

Australian States and Territories differ markedly in size, climate, human history and population density, and macropod diversity among other measures. Therefore, it is interesting and potentially informative to assess the variation in use of translocations for macropod conservation across states/territories and over time. Although a thorough

search for macropod translocation programs was conducted for this study, the data presented is not assumed to be comprehensive. However, I believe that overall trends can be confidently identified.

In addition to biological considerations, inter-related political, social and economic factors will also affect the conservation management of threatened species (Lande 1988). Inconsistency in the number of translocations undertaken by each state/territory may therefore be due to a number of different factors.

As data analysis showed a degree of positive correlation between number of potential species and number of translocations undertaken, this may in part explain why Western Australia and South Australia undertook considerably more macropod translocations than the other states/territories. In addition to the history of macropod decline within each state/territory, the natural environment can have a bearing on the number of translocations undertaken. Within Western Australia, sodium fluoroacetate (a toxin to which canids and felids are particularly susceptible) is found naturally in some floral taxa. As a result, poison baits made with a synthetic version of this toxin can be used to control introduced predators without risking non-target species (with the exception of dingoes). Western Australian threatened species managers therefore have an advantage in controlling introduced predators in areas of otherwise suitable habitat for macropod translocations. The majority of the 46 releases undertaken in Western Australia between 1996 and 2005 were part of the 'Western Shield' project, a State Government initiative aiming to recreate faunal assemblages after successful introduced predator control with sodium fluoroacetate (WA DEC 2008). The popularity of translocations in Western Australia is therefore, in part, a result of favourable environmental factors (Morris 2000).

Analysis showed little correlation between the Gross State product and the number of translocations undertaken. Therefore, although states/territories with strong economies may have the resources required to undertake translocations, it does not guarantee that such species recovery work will be attempted.

Within democratic states, the desires of citizens will theoretically dictate the use of public resources. Thus, community and political will are necessary to ensure such resources are committed to threatened species programs. As the majority of natural resource management falls under state/territory jurisdiction within Australia, the community must exert sufficient pressure upon state/territory governments to ensure funding is allocated to threatened species conservation. Further, as the majority of Australian macropod translocation programs release animals to government-managed land, there clearly needs to be the community and political support for the reservation of crown land for conservation ends. Inequalities in the number of threatened macropod translocations undertaken by individual Australian States/Territories may be in part due to differing community attitudes towards biodiversity conservation. However, attempting to measure such differences is beyond the scope of this study.

It is clear from the data that non-government conservation organizations are heavily involved, arguably disproportionately, in the translocation of macropods in Australia. It must be considered, however, that state/territory government conservation agencies have been involved in the majority of these releases, providing both regulatory oversight and practical assistance.

3.4.2 Species chosen for translocation

Identifying the range of macropod species translocated in Australia between 1969 and 2005 permitted this study to investigate taxonomic translocation priorities, conservation status, and any change in species' classification over time. The following discussion looks at two translocated Australian macropod species, *B. penicillata* and *B. gaimardi*. Although closely related, one of these species was the most commonly translocated, whilst the other was released only once. Several explanations as to why translocation effort differed so markedly between the two taxa are proposed.

In Chapter 2, 12 biological/social factors were identified as potentially influencing the decision to select a particular species for translocation. The most likely of these factors

to have resulted in *B. penicillata* being the most commonly released species are discussed below.

The level of threat to a particular species will theoretically have considerable influence on the level of recovery action undertaken. Therefore, since enormous effort has clearly been exhausted on *B. penicillata*, it is reasonable to assume that this species was facing significant threat when translocations were undertaken. The earliest known conservation releases of *B. penicillata* in Western Australia occurred in 1977 and 1980-2. During this time, the Western Australia *Wildlife Conservation Act 1950* listed the species as 'fauna that is rare, or otherwise in need of special protection' (Department of the Environment, Water, Heritage and the Arts 2008). The first releases of *B. penicillata* in South Australia were between 1979 and 1983. The *National Parks and Wildlife Act 1972* listed the species as 'rare' during this period. *Bettongia penicillata* was translocated to New South Wales for the first time in 1998, where the species was extinct. In addition to Australian State listings, in 1982 the IUCN listed *B. penicillata* as 'endangered' (IUCN 2008). Considering the above points, it can be argued that the level of threat to *B. penicillata* was a factor in determining its choice as a translocation candidate species.

However, this premise is obfuscated by the fact that *B. penicillata's* conservation status was downgraded during the study period. In 1996, the species was delisted from the Western Australian *Wildlife Conservation Act 1950* as a result of the success of conservation efforts (including translocations) (Department of Environment and Conservation Western Australia n.d.). Similarly, the IUCN downgraded the species to 'Lower Risk' (IUCN 2008). Although *B. penicillata* was therefore no longer considered as threatened as prior to the commencement of recovery work, and at less risk than other species, translocations continued to occur in Western Australia.

Mawson (2003:116) gives 5 reasons for continuing to translocate *B. penicillata* after its delisting:

- the species was still absent from large areas of its former range

- *B. penicillata* is a useful 'indicator' species for adequacy of introduced pest control, i.e. if it fails to establish it is unlikely that other, more susceptible species will persist
- translocation programs involving the species provide opportunities for advancing staff animal handling skills
- *B. penicillata* translocation and monitoring provides the opportunity for positive media and community involvement in conservation projects; and
- translocations of *B. penicillata* have been part of larger faunal assembly reconstruction programs, where re-establishing a suite of species, rather than a single taxon, is the overall objective.

Although substantial gains had been made in improving the likelihood of survival of *B. penicillata*, since 2001 the species has been experiencing a second, dramatic decline on mainland Western Australia and South Australia (Woylie Conservation Research Project 2008). The cause(s) of this decline are as yet unknown, however 'multiple interactive factors are expected responsible, with disease considered the most likely primary and ultimate agent of decline' (Woylie Conservation Research Project 2008:5). Consequently, translocations may again be an important tool for future *B. penicillata* 're-recovery'.

It has been suggested that *B. penicillata* is a keystone species, with respect to its contribution to the construction of soil mosaics (via its digging activity), and its dispersal of seeds and fungal spores (Carter 2004, Murphy 2005). This may have influenced the decision to translocate the species. Further, consideration of the probability of successful translocation is likely to have influenced the choice of *B. penicillata* for release. The initial success of translocations, and the continuing recovery of remnant populations as a result of introduced predator baiting, is likely to have favoured the ongoing selection of this species.

Hardiness and high fecundity are attractive physiological traits for translocation candidate species (Christensen and Burrows 1994, Richards and Short 2003). As *B. penicillata*

displays both of these characteristics (Mawson 2003), decisions to select the species for translocating may have been influenced by its physiology. In addition, the fact that suitable source populations existed may also have influenced the choice of *B. penicillata* for translocation. Increases in the size of remnant *B. penicillata* populations following the introduction of baiting for exotic predators permitted the removal of animals for translocation. Further, populations which were successfully re-established had in turn become source populations for further releases (Mawson 2003, Australian Wildlife Conservancy 2002).

As introduced predators are considered the primary threat to macropod translocation success (Brown 1991, Christensen and Burrows 1994, Johnson et al. 1989, Priddel and Wheeler 2004, Short et al 1992), the existence of predator-controlled release sites within suitable habitat in Western Australia, and appropriate habitat free of predators in South Australia (seven of 11 *B. penicillata* releases were to offshore islands / predator-proof enclosures), are likely to have favoured the choice of *B. penicillata* for translocation. In addition, releases of *B. penicillata*, even when the conservation status of the species had been downgraded, may have continued in part due to *B. penicillata* being a useful 'surrogate' species for other, less hardy taxa (Mawson 2003).

Only a single translocation of *B. gaimardi* was attempted over the study period. In order to identify why a second species of *Bettongia* was translocated on only one occasion, in stark contrast to the number of *B. penicillata* releases, the most likely factors resulting in this discrepancy are discussed below.

The single successful translocation of *B. gaimardi* was attempted in Tasmania in 1971. At this time, loss of habitat on the Tasmanian mainland raised concern about the future of a suite of species, including *B. gaimardi*. Conservation status was thus the motivation for this release. Converse to fears, however, the species remained common in Tasmania, and this is likely to have lead to a cessation of *B. gaimardi* translocations. However, although the species is considered common in Tasmania (and therefore not requiring conservation work including translocation), it is extinct on the mainland (Rose and Johnson 2008).

The remainder of this discussion will focus on why the species was not translocated to its former range in south-east Australia.

It has been recognised that due to their diet and habit, *B. gaimardi* are likely to be important dispersal agents of hypogeous fungal spores (Rose and Johnson 2008). As symbiotic relationships exist between fungal and tree species in the forests where *B. gaimardi* are found, the species could be considered an important environmental component. Even if published research has not explicitly identified *B. gaimardi* as a keystone species, it could be argued that enough is known of their potential to affect ecosystem health that they would be considered an important species for translocation on these grounds. It is therefore likely to have been a factor other than 'keystone status' that has precluded *B. gaimardi* from translocation programs.

The demise of *B. gaimardi* on the south-east Australian mainland is believed to have been caused by the red fox *Vulpes vulpes* (Rose and Johnson 2008). As this species is still common, along with feral cats, an inability to control these two predators may severely compromise translocation success. This may be one reason why *B. gaimardi* releases to the mainland have not occurred. Fecundity is similar between *B. gaimardi* and *B. penicillata* (Mawson 2003, Rose and Johnson 2008), and thus the species would appear suitable for translocation with regards to its ability to rapidly increase population size. Therefore, factors other than reproductive physiology are likely to have deemed *B. gaimardi* unsuitable for translocation.

Rose and Johnson (2008) identify 2 sub-species of *B. gaimardi*; *B. gaimardi gaimardi* (mainland sub-species), and *B. gaimardi cuniculus* (Tasmanian sub-species). As the mainland sub-species is extinct (Rose and Johnson 2008), it is obviously not available for translocation. However, it is unlikely that this would have stopped translocation attempts using *B. gaimardi cuniculus*, as sub-species of *B. penicillata* have been translocated into the former range of *B. penicillata penicillata* within South Australia and New South Wales (Copley 1994, Priddel and Wheeler 2004). Considering the common status of the species in Tasmania, the absence of a suitable source population(s) appears not to have

been the cause of the lack of *B. gaimardi* translocations during the study period. Conversely, it is highly likely that a lack of suitable habitat was a key factor in not considering *B. gaimardi* a suitable translocation candidate species. South-eastern Australia has undergone dramatic environmental change since European colonisation. In addition to contributing to the initial decline of the species, the vast tracts of land cleared for cropping and grazing have no doubt restricted the amount, size and quality of habitat now available as prospective *B. gaimardi* release sites. In addition, these areas are further degraded by the presence of introduced predators.

It appears that the essential difference explaining why two different species of the same genus are represented so unequally in translocations during the study period is a lack of suitable habitat exacerbated by the presence of introduced predators. Even if forest areas large enough to support viable *B. gaimardi* populations exist on the mainland, baiting for introduced predators on the scale undertaken in Western Australia may not be considered an option for eastern States conservation managers, due to concerns about non-target bait uptake by native species lacking natural tolerance to sodium fluoroacetate.

3.4.3 Use of captive bred animals for translocation

Two major reviews of worldwide animal translocations have been undertaken. Griffith et al. (1989) surveyed translocations of birds and mammals undertaken between 1973 and 1986 in Australia, Canada, New Zealand and the United States (including Hawaii). Fischer and Lindenmayer (2000) studied 116 reintroductions conducted between 1979 and 1998 as part of their review of published papers regarding animal relocations. Griffith et al.'s (1989) study of 197 translocations revealed 83% of programs released wild-caught animals. Conversely, Fischer and Lindenmayer (2000) found that releases of captive bred animals were favoured, with 45% of programs releasing animals from captivity and 39% from the wild. A lack of consistency is therefore evident between reviews of animal translocations when the variable of founder source is investigated. Australian macropod releases, some of which were included in both of these reviews, follow the pattern identified by Griffith et al. (1989) of favouring wild animals in translocation programs.

Not all translocation managers have the choice of using either wild or captive bred animals, or both. For some species, wild populations may no longer exist, necessitating the use of captive bred animals in attempts to create additional populations. Conversely, sufficient numbers and/or satisfactory demographic characteristics of captive bred animals may not be available.

The data shows several factors which may explain the lower reliance on wild animals for translocations to islands or predator-proof enclosures. First, translocation managers may be less concerned about the use of wild animals as some/all of the threatening processes which lead to the initial decline of the species may be absent. Issues such as potential behavioural problems associated with captive raised animals may therefore be considered less critical. Second, for some species, wild populations no longer exist, or may be unable to sustain the removal of animals for translocation. However, captive colonies may exist and be able to provide sufficient individuals for release. This may be the case for those species which are particularly susceptible to the effects of introduced predators. Such taxa are also likely only to be translocated to islands or predator-proof enclosures. In some instances, this inability to use wild animals may change as a result of wild population recovery and the successful establishment of additional wild populations. Last, islands or predator-proof enclosures may be the release site of choice for rehabilitated animals or those translocated from public wildlife parks.

My study has shown that pre-release training is not considered an important part of Australian macropod translocations. This may be due to predation by introduced carnivores being identified as the critical threatening process facing macropod translocations. It has been estimated that cats and foxes have been in Australia since the early 1800s (Catling and Coman 2008) and 1860s (Denny 2008) respectively. Since their arrival, threatened macropods have not developed satisfactory behaviours to avoid predation, even though declines may have occurred over a substantial length of time. Therefore, it could be argued that relatively short periods of training prior to release will not result in behaviours that did not develop during over 100 years of wild interactions between predators and prey (Lapidge 2001). Pre-release training may not, therefore, be

considered a worthwhile investment of resources available to Australian macropod translocations.

3.4.4 Release area

The arrival of Europeans dramatically changed the nature of the Australian natural environment. These changes must be acknowledged, and in almost all cases, addressed when undertaking threatened species translocation. The data show the majority of macropod translocation programs used some form of pre-release site management. Although information could not be obtained regarding the use of pre-release site management for 35 translocations, the majority of these releases were undertaken in Western Australia where a massive introduced predator control program is conducted. This program is likely to be critical to the success of the translocations. Thus, it is unlikely that if information was made available for these particular programs, it would differ markedly from the data set obtained. Clearly, pre-release site management is considered an important aspect of macropod translocations in Australia.

The preference for macropod reintroductions in Australia clearly indicates that threatened species managers believed, at least at the time that translocations were undertaken, that the species' former range could support their re-establishment. Reintroductions may also have been preferred where little knowledge existed of rare species' habitat requirements. In such circumstances, conducting releases within areas of known former range increases the likelihood of suitable habitat occurring at the release site. Introductions may have been less popular due to the potential for significant environmental impacts from such translocations, including habitat alteration/destruction and predation/competition with fauna naturally occurring at the release site (IUCN 1987). Further, the broad commitment not to introduce species outside their former range is likely to have been an important contributing factor (IUCN 1998, Serena and Williams 1994).

The fact that only one of the 109 translocations was an augmentation likely reflects a pattern in Australian macropod decline and management. Many Australian macropod species declined critically in number and range after European colonisation, until only a

single, or few, populations remained (e.g. *Lagorchestes hirsutus*, *Onychogalea fraenata*). When animals have been available for translocation from captive breeding programs, management focus has often been to establish additional populations, rather than supplementing existing ones, thereby insuring against the catastrophic (or gradual) loss of the extant population. Dependent upon the long-term success of existing programs, perhaps a greater proportion of future macropod translocations will be augmentations.

The noted reluctance to introduce animals outside their former range may also have dictated that 19 of 20 introductions occurred to restricted release sites; that is offshore islands and predator-proof enclosures. As discussed above, by restricting the dispersal of translocated populations, either within the boundaries of an island or fenced enclosure, potential negative impacts caused by the introduced species will be contained. Consideration must, however, be given to the carrying capacity of the release area, and the presence of other threatened species.

Offshore islands are primarily chosen as macropod release sites in order to avoid the threatening processes responsible for species decline on the mainland. The fact that relatively few introductions to offshore islands used pre-release site management could thus be explained by a lack of critical threatening processes at the release site. Conversely, for the small number of reintroductions to offshore islands (n = 6), 67% employed pre-release site management; likely indicating the need to remove the threats which lead to the original extinction of the species.

There is a growing consensus that the root cause of macropod decline in Australia is predation by introduced taxa. Further, such species caused the failure of early macropod translocation attempts. Consequently, the popularity of pre-release introduced predator control work, particularly for releases to the mainland, is not surprising. The removal of introduced competitors was also common (37%), reflecting the perceived importance of this threat to translocation success. Interestingly, very few translocation programs (four) mentioned fire in relation to pre-release site management. As fire is an important environmental influence and habitat management tool in Australia, it could perhaps have

been expected to feature more frequently in pre-release site management. However, protected area management agencies are likely to include the use/control of fire within their jurisdictions generally, and thus may not have undertaken work specifically in relation to translocations.

Considering the vast majority of translocations (for which data was available) used some form of post-release site management, it is clear that such action is considered an important part of efforts to establish populations of threatened Australian macropods. The fact that the use of post-release site management is even more pronounced when considering only reintroductions to the mainland could arguably be expected. As reintroductions by definition are releases to former range, the species must have previously been driven to extinction in the area by particular threatening processes. As the control of such threats is likely to be an ongoing concern (such as the management of introduced predators), it is expected that continued site management would be necessary.

As threatening processes may be more effectively controlled within 'closed' release sites (islands), it may be expected that some site management activities may not be required on an ongoing basis. Others, however, such as fire management (used by one program) and weed eradication (used by the other) may require continual assessment and action. As offshore island release sites are often chosen because the threatening processes which contributed to the extinction of the species in other areas are absent, it could be expected that ongoing post-release amelioration work, such as predator control, would not be required. This may explain the lack of post-release site management evident in the data.

The types of post-release site management employed as part of introductions to offshore islands may indicate issues unforeseen prior to release. Such issues may arise as the species has not historically existed in the release environment, and therefore it may be difficult to predict its impact on the island ecosystem. Further, the island's ability to sustain the species may be unknown. An example exists in the macropod introductions to islands investigated, where a program initiated culling post-release to ensure the translocated species did not overgraze the release area.

3.4.5 Population Viability Analysis

The data suggests PVA is not considered an important tool in Australian macropod translocations. Further, there appears to have been no increase in the use of the technique over time. Even though more translocations were undertaken in the latter half of the period 1989-2007, a contemporaneous increase in the use of PVA did not occur. Although data was only available for less than half of the translocations undertaken during this period (42%), there is no reason to believe that translocations for which data could not be found, or requests for data were denied, would have a higher use of PVA than programs for which published data was available or requests for data honoured.

3.4.6 Release methodology and founder group composition

The relatively small release group numbers observed in Australian macropod translocations may be directly related to the absence of critical threats in the release area. As identified by Short et al (1992), without suitable release habitat (i.e. where introduced predators have been successfully controlled), macropod translocations may fail even if large numbers of animals are released. Therefore, relatively small founder numbers may be all that are required in the absence of these threats. As previously discussed, introductions are often to habitat where the primary threat(s) which lead to the initial decline of the species is/are absent. Translocation managers may be confident in releasing smaller founder groups than those conducting reintroductions, on the premise that most, or all, the animals will survive.

The number of releases most frequently employed over the study period was one, which may reflect confidence in the effectiveness of predator control measures undertaken at the release site. Islands are generally chosen as release sites for Australian macropods because threatening processes which lead to the decline of the species are often absent. It is not surprising therefore, that nine of the 17 translocation programs which released all animals at once did so into an island environment. Similarly, translocation managers may have confidently released all animals in a single event to predator-proof enclosures greater than 100 hectares. Of the six releases to the mainland which released all animals at once, three were Western Australian translocations to areas where predators had been

successfully controlled with baiting programs. Therefore, it would appear that the trend towards translocating all animals in a single event is linked with confidence regarding the suitability of the release site habitat.

Although more translocations chose to release all animals at once than any other number of release events, a significant number of Australian macropod translocation programs (29%) released animals on two occasions. Of these 14 programs, initial release cohort numbers were smaller than, or equal to, the number of animals in the secondary release group in eight cases. It may be speculated that the initial releases were smaller in order to assess the suitability of habitat before risking further animals. For the remaining six translocations, the first release group was larger than the second. As all of the secondary (and all but one of the primary) releases sourced animals from captive breeding colonies, animal availability may have dictated the pattern of releases.

Three Australian macropod translocation programs did not follow the pattern of only conducting one or two releases. These programs sourced animals from captive breeding colonies (including one which released hand-raised individuals), and therefore it appears that staggered availability of animals determined the high number of release events.

Any discussion regarding the timing of Australian translocations which groups all releases into one data set must be careful in interpreting results due to climatic differences across the continent. In addition, although reasonably predictable by annual and longer cyclical (southern oscillation) patterns, the Australian climate is dynamic and variable. The following discussion is thus broad and general. The lack of translocations undertaken in tropical regions (3 of 109), however, allows more confidence in the observations. In the southern regions of Australia, September can be considered climatically benign, and thus a suitable time to translocate animals. However, considering that heat is arguably one of the most significant stressors of animals during translocations, it is surprising that so many releases occurred in December. Perhaps such stress is considered acceptable if other agents of stress are minimised (e.g. short

translocation distance, freedom from disease, predator-free release environment), or seasonal conditions are favourable (high rainfall, below-average temperatures).

Short et al's (1992) review of Australian macropod translocations identified a trend towards releasing more females than males. This current study of 53 translocations for which data was available (including the single release investigated by Short et al. for which gender ratio data could be obtained) showed a similar pattern. As 37 of the releases occurred from 1992 onwards, it appears that the trend towards female-dominated release groups has continued over time. A desire for rapid population growth may explain the apparent preference for female dominated release groups. Alternatively, as Short et al. (1992) identified a higher mortality rate within translocated female macropods, releasing more females than males may have been an attempt to address this problem. On the occasions when male dominated groups were released, the size of the bias towards males was smaller than when female dominated groups were translocated. Therefore, male bias in release groups may have been due to availability rather than a conscious effort to release more males, whereas it appears that releasing female-biased founder groups may have been intentional.

As discussed in Chapter 2, it has been identified that for some species releasing only females initially may have advantages in avoiding excessive male dispersal in subsequent releases. However, the fact that only two of 46 Australian macropod translocation programs for which data were available released exclusively females initially suggests that such benefits are not acknowledged for Australian macropod species.

It appears that no clear preference exists for releasing a particular age mix of Australian macropods. The difficulty in obtaining data regarding animal age on release may be an indication that it was not an important consideration for translocation managers. This may be due to the fact that many species become sexually mature at a relatively young age, particularly females which may produce young when less than 12 months old (Van Dyck and Strahan 2008). Therefore, translocated sub-adults will have the potential to contribute to population growth relatively soon after release. The risk that these animals

will not survive the short period of time after release until they reach sexual maturity may be regarded as acceptable. The fact that 16 translocations released females with pouch young suggests that some translocation managers are confident that individuals are suitably robust to endure the stresses of both translocation and carrying offspring. However, several programs stated that only females with small young were selected, presumably to reduce stress on the mother and the likelihood that the young may be ejected from the pouch.

The fact that only 36% of Australian macropod translocations undertaken between 1969 and 2006 (for which data was available) released animals initially into acclimatization pens suggests that the majority of managers did not believe this step was necessary. However, more recently, the use of pens has proven more popular, as nearly half of the releases carried out this decade have used them. The use of acclimatization pens also appears to be significantly more popular with reintroduction programs than introductions. This trend may be linked to the release site environment, as introductions are often conducted where primary threats to the translocated species are absent. As the extra expense incurred by using this technique (materials, construction, staff time) is likely to be considerable, managers should be certain of the benefits of pens prior to using them in a translocation program. Consequently, experimental releases to test the effectiveness of pens for particular species will be required. Hardman and Moro (2006) undertook such an experiment with both *Lagostrophus fasciatus* and *Lagorchestes hirsutus*. Their research indicated that there was no advantage in using acclimatization pens for these species. Translocation proponents should heed the results of this study before assuming that acclimatization pens will provide benefits to translocated animals.

It appears that managers initially choosing to use acclimatization pens are reassessing their use over the course of the programs. Although the instances were few, this resulted in both the abandonment, and adoption, of the technique over subsequent releases. Abandonment may have been due to an assessment that no benefit was derived from the pens, or that animal welfare was in fact compromised. For example, managers abandoned the use of pens for *B. lesueur* early in a translocation attempt when collision

between the fence and animals occurred (Christensen and Burrows 1994). Similarly, the data shows translocation managers appear to be adjusting the level of assistance provided to released animals in response to experience gained. Greater efficiency in release protocols is likely to be driving such reassessment, as post-release support is unlikely to be detrimental to translocated animals.

However, as post-release assistance other than that provided by acclimatization pens is being provided in contemporary translocations, it is clearly still considered beneficial by many translocation proponents. The fact that translocation managers releasing animals to islands provided support more often (proportionally) than did proponents releasing individuals to the mainland may be explained by concerns regarding the suitability of release site habitat, particularly when introductions were undertaken.

3.4.7 Post release monitoring

The data show that post-release monitoring is considered an important part of Australian macropod translocations. Although data regarding the use of post-release monitoring was available for only 81% of translocations investigated (n = 88), I have no reason to believe that the remaining programs would differ markedly from this trend. It appears that post-release monitoring has been considered an important aspect of Australian macropod translocations over the entire study period 1969-2005, with no pattern of change in the use of monitoring over time.

The popularity of both trapping and radio-tracking as post-release monitoring methods is likely due to the broad range of information that can be obtained by these techniques (Chapter 2). Spotlighting and search for sign (the only other forms of monitoring used by more than three programs) are labour intensive, and although relatively inaccurate when compared to trapping and radio tracking, may provide valuable data for trap-shy species. The fact that only two Australian macropod translocations used video surveillance and remote photography is not surprising. Such technology is relatively new, as is its application to wildlife research. As remote surveillance technology advances, its use in translocation monitoring is likely to increase.

Short et al. (1992) were critical of the limited use of radio tracking in the macropod translocations they studied. Within their paper, six translocations are described in detail, and tables provide an overview of a further 19. Only one translocation was explicitly identified as having used radio tracking post-release; one of the six releases described in depth. However, a further four of the translocations listed in table form by Short et al. also used the technique. As it could be reasonably expected that radio tracking technology would have been available for Australian macropod translocations undertaken from 1970 onwards, 17 of the 25 translocations listed by Short et al (1992) could have used radio tracking to monitor animals post-release, and five of these did (29%). It appears from my data that radio tracking has been more frequently used in recent times. Although for many translocations undertaken from 1995 onwards data regarding the type of post-release monitoring used could not be obtained, I have no reason to suspect these translocations would have used radio tracking to a lesser degree than those for which data was available.

Many authors have been critical of the low rate of publication of translocation program results (Mathews et al 2005, Fischer and Lindenmayer 2000, Stanley Price 1991, Short et al. 1992, Saunders and Friend 1993). I was able to find published, peer reviewed papers for only a third of the translocations identified. My study therefore supports the findings of previous reviews that identified low rates of translocation data publication. Although a published document available worldwide is the ideal, basic information regarding the release and translocation proponent responsible can be very valuable. For example, mention of a particular Australian macropod translocation discovered during an internet search permitted contact to be made with the proponent. A questionnaire requesting information was sent, and within 3 days a reply was received which included the completed questionnaire and an unpublished PhD thesis. Thus, with very little effort, a highly detailed account of the translocation program was obtained. Therefore, although the material was unpublished, it was still very accessible. On other occasions, requests for unpublished internal agency documents providing translocation program data were honoured promptly.

Although the above examples show that translocation information, although unpublished, can at times be easily accessed, I also encountered situations where the opposite occurred. Mentions of translocations discovered during research lead to dead-ends; either resulting from data being unavailable (never recorded adequately, or records had been lost/destroyed), or requests for further information were denied.

3.4.8 Suspected / known causes of translocation failure

From the data presented regarding all translocations, it appears that more than a single cause was known/suspected in the failure of the majority of unsuccessful programs. This supports Griffith et al's (1989) suggestion that translocation success is based upon a number of interrelated factors. Predation by native species was listed as a suspected/known cause of the failure of five translocation programs, and thus should not be underestimated in some release environments. Native competitors and poor quality habitat/starvation were identified as significant known/suspected causes of failure in programs releasing animals into predator-proof enclosures. This finding highlights the need to consider the carrying capacity of spatially restricted release sites. Predation by introduced species is increasingly regarded as the principle cause of mammal decline on mainland Australia (Johnson 2006, Burbidge 2008), and control is problematic. Therefore, it is not surprising that cat/fox predation was identified as the single most dominant known/suspected cause of program failure. The fact that so few unsuccessful island translocations could be found is likely due to their selection as translocation sites precisely because threatening processes found on the mainland are often absent/controllable.

From the data, it is clear that the primary threat to Australian macropod translocation success over the study period was predation by introduced species, namely foxes and cats. Further, translocation managers were aware of this critical threat prior to release. This is the same conclusion drawn by Short et al. (1992) in their review of Australian macropod translocations. Short et al.'s data were included in this review, however data for a further 15 failed translocations were also obtained. The critical relationship between introduced predator control and macropod translocation success in Australia has

also been noted in attempts to re-establish other Australian faunal groups (Serena 1994). Although this trend is mirrored in New Zealand threatened species management (Serena 1994), it is not a world-wide phenomenon. Fischer and Lindenmayer's (2000) review of worldwide translocations identified that predation was not identified as a cause of program failure in the United States of America. This is perhaps not unexpected, as a suite of native predators exist in North America, with which other threatened indigenous taxa have evolved.

The data identified particular species which are succumbing to the same threats over many years of translocation attempts. It may be assumed that pre/post- release site management to control introduced predators was ineffective in these examples, and that techniques to ameliorate the threat of introduced predation have, at least in some regions of Australia, not improved over time. However, this does not mean that releases cannot be successful. *Lagorchestes hirsutus* was successfully translocated four times, *Bettongia lesueur* on five occasions, and *Bettongia penicillata* 12 times over the period covered by the study. However, all but three of these translocations were to predator-free islands, predator-proof enclosures, or the south-west of Western Australia where broad-scale introduced predator control has been very successful. The other three programs were to fenced isthmuses managed with intensive predator control. I have found no evidence that the pattern of translocation success identified for these three species does not apply to other threatened Australian macropods. The data implies that without effective introduced predator control, translocation of macropod species will fail. In these situations, other variables, such as the release methodology employed, is unlikely to have any effect on translocation outcome.

3.4.9 Methodological and environmental factors critical to Australian macropod translocation success

Classification tree analysis was completed in an effort to identify methodological and environmental factors common to successful Australian macropod translocations. If such factors could be identified, it may provide useful information for maximising the success of future translocation attempts.

When considering the first analysis, which examined all programs for which success/failure was known, particular species were clearly more successfully released than others. It is important to note, however, that 7 of these 11 species were translocated only once, two were released once and one species on three occasions. As some of the less successful species were released more than five times, the difference in sample size must be considered. However, the data suggest that either some species are physiologically and behaviourally more suitable for release, or that specific release methodologies have enabled their successful translocation. Considering that all but one of the species in the taxa recording a lower success rate were translocated successfully at least once, it appears more likely that the release environment, or release protocol, is responsible for the dichotomy between the groups.

Of the 16 different environmental and methodological factors analysed for each translocation, only the initial release size, and the use of post-release site management, were common to all the releases of those species recording a higher translocation success rate. Theory and past review suggest that releasing larger numbers of animals, up to a point, will increase the chance of translocation success (Griffith et al. 1989, MacArthur and Wilson 1967). Therefore it would be unlikely that the higher rate of program success reported for one group of species would result from releasing cohorts within this study's smallest size range category (1-30). Further, as all programs (for which data were available) also used post-release site management this factor cannot have dictated a greater rate of success. The most likely reason for the difference in success rates is that 16 of 17 translocations of the species within the higher success group were to areas free of introduced predators (the status of introduced predators at the failed release site was unknown).

The finding that acclimatisation pens were associated with a poor success rate suggests that the pens may have negatively impacted certain translocation attempts. However, I could only find one published account of the abandonment of pens due to concerns for animal welfare (Christensen and Burrows 1994). Further, the only Australian macropod release which experimentally tested the use of acclimatization pens found they conveyed

no disadvantage, or advantage. To link translocation failure with pen use is thus likely to be erroneous.

As cats are believed to be a greater threat to native wildlife in arid zones than in wetter areas (Johnson 2006), it is perhaps surprising that within the species group recording a lower translocation success, animals released to drier areas (mean annual rainfall less than 232mm per year) had a higher success rate than those to wetter regions. However, this success rate may be explained by the fact that three of the seven releases to lower rainfall areas were to predator-proof enclosures, two to islands, and the remainder to areas where introduced predators had been heavily targeted.

At the 'base' of the tree, a final branching split releases to higher rainfall areas into translocations to New South Wales and South Australia, and into Western Australia and the Northern Territory. The higher success rate for translocations to the latter states is likely due to the broad-scale and effective use of 1080 baits for introduced predator control in WA, and that both NT releases were to predator-proof enclosures.

When classification tree analysis was applied to those projects which could be assessed against Short et al.'s (1992) criteria, the initial splitting into species and use of acclimatisation pens also occurred. However, I believe that the subsequent split, again by species, had less to do with the characteristics of the species themselves than introduced predator presence at the release site. The more successful species were released to predator free areas or to where predators had been effectively controlled. Further, the majority of successful releases of those species which experienced lower success rates occurred to similar sites where introduced predators were absent/controlled. Thus, although factors other than introduced predators can cause translocation failure in these species, releases to sites free of cats/foxes greatly enhances the chance of program success.

3.5 Conclusion

When planning translocations, managers must make decisions regarding all stages of the project, from identifying the candidate species and release location to the most appropriate release protocols and post-release management techniques. When classification tree analysis was conducted on the macropod translocation data, no significant pattern emerged regarding particular release protocols that, if followed, will ensure program success. This has important management implications.

First, as translocations require significant resource investment, the identification of the most efficient way to release a given species is of paramount importance in order to maximise funds available for threatened species conservation. Experimental translocations designed to test release protocols, such as those undertaken by Hardman and Moro (2006), are required before particular strategies are accepted as necessary for program success.

Second, control of introduced predators appears to be significantly more important than the adoption of a particular set of translocation methodologies to Australian macropod translocations success. Introduced predators caused, or contributed to, the failure of 71% of unsuccessful programs investigated by this study. Further, the outcome of the majority of translocations deemed successful is likely to have been very different if cats/foxes had been present at the release sites, regardless of the release protocols employed.

Monitoring of both the last wild population of *L. hirsutus*, and subsequent reintroduction attempts, showed that the presence of even very low numbers of cats/foxes resulted in population extinction (Gibson et al 1994, Langford 1999). My study supports Christensen and Burrows' (1994) proposal that, for individual species, experimental research is required to establish the acceptable level of introduced predator presence at translocation release sites that will permit population persistence. For many species, no safe level of cat/fox presence may exist.

Although other factors may compromise translocation outcome in the absence of exotic predators, the presence of the primary introduced predator threat(s) (cat, fox or both) at

unacceptable levels at the release site will result in unsuccessful translocation. Unless this fact is appreciated, particularly that for some species predator presence must terminate any plans for translocation, threatened species managers may continue to undertake macropod translocations which are doomed to failure.

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Chapter 4. Demography of the Uluṟu – Kata Tjuṯa National Park mala population from establishment to five years post-release: 2005-2010

4.1 Introduction

The primary aim of threatened species introductions and reintroductions is to create new, viable populations that will persist into the foreseeable future (Fischer and Lindenmayer 2000, Griffith et al. 1989, IUCN 1998, Kleiman et al. 1994, Pople et al. 2001, Sarrazin and Barbault 1996, Seddon 1999a, Stanley Price 1991, Waples and Stagoll 1997, Wolf et al. 1996). However, the history of global threatened species translocations shows a low overall rate of success (Beck et al. 1994, Fisher and Lindenmayer 2000, Griffith et al. 1989). Small populations, such as those created by introductions and reintroductions, are particularly susceptible to extinction pressures which fall broadly into three interrelated categories: genetic, demographic and environmental (Frankham 1998, Lundie-Jenkins 1998, Shaffer 1987, Seebeck, et al. 1990, Soderquist 1993, Stanley Price 1991).
In most cases, a combination of processes will drive the decline or extinction of a population (Gittleman and Gompper 2001, Seebeck et al. 1990). These factors are addressed in detail in Chapter 2.

The need for comprehensive post-release monitoring is widely recognized as essential to maximize the chance of translocation success (IUCN 1998, Kleiman 1989, Groombridge 1992, Seddon 1999a, Stanley Price 1991). Not only is it impossible to determine the success of a translocation without monitoring, the establishment of a viable population may depend upon intervention after the initial release (Seddon 1999a, Seddon 1999b, Seddon et al. 2003). Considering the intrinsically low population growth potential of some species, the threat of climate change, and that genetic and environmental extinction pressures act over the long term, monitoring must also be long term in nature (Hayward

et al. 2010, Morell 2008, Pople et al. 2001). In spite of this, many translocation programs are subject to little or no post-release monitoring (Fischer and Lindenmayer 2000, Short et al. 1992). This fact compromises both individual programs and the potential to increase the understanding of translocation biology worldwide (Stanley Price 1991).

Although post-release monitoring can be used to gain a variety of information about the fate of translocated animals (for example dispersal of founders, behaviour of captive bred animals in the wild, animal health, cause of death, impact of founder cohort on the release site environment; refer Chapter 2), this chapter is primarily concerned with population demography. Demographic monitoring of populations can either involve 'direct' or 'indirect' techniques. The post-release capture of animals can identify new individuals born after release, thereby allowing population growth and size to be estimated (Richards and Short 2003). Known to be alive (KTBA) estimates can also be made (Short and Turner 2000). Further, sex ratio and age structure can be ascertained (Fischer and Lindenmayer 2000). Other 'indirect' monitoring techniques, such as transect counts and the identification of animal sign, may also provide a guide to population numbers (Morris et al. 2004, Pople et al. 2001, Seddon et al. 2003). Although radio tracking of released animals will not provide information regarding population growth, it can assist with KTBA estimates. In addition to providing contemporary data regarding population demographics, post-release monitoring also provides the opportunity to create population viability models (Ruiz-Miranda 2010).

Case studies provide clear evidence of the need for long-term post-release monitoring of translocated populations. As a consequence of over hunting and collection of animals for the wildlife trade, the Arabian oryx (*Oryx leucoryx*), became extinct in the wild in 1972 (National Wildlife Centre 2010, Zafar-ul Islam 2010, Morell 2008). An international captive breeding program permitted reintroductions of the species to commence in 1982 (Ostrowski et al. 1998, Marshall and Spalton 2000, Seddon et al. 2003), and initial post-release population monitoring consisted of absolute counts of individuals (Seddon et al. 2003). Drought conditions coincided with the early years of the program, however significant population increases were recorded for six years following the return of

favourable conditions (Marshall and Spalton 2000). As population increases made absolute counts impractical, estimates were made using the mark-resighting technique (Seddon et al. 2003). By 1995, the population had expanded to an estimated 280 oryx (Gorman 1999), however monitoring revealed poaching and hunting of animals had recommenced (Gorman 1999). Despite this, the wild population reached a peak of around 450, until human harvesting induced a crash which reduced numbers to 138 by 1998 (Gorman 1999, Morell 2008). Project managers deemed the wild population to be unviable and 40 oryx were captured and returned to breeding facilities (Gorman 1999). If post-release population monitoring had ceased following the recording of high population growth from 1987-1993 (average 31%, Marshall and Spalton 2000), oryx would likely have been poached to extinction. Consequently, the genetic value of the 40 rescued individuals would have been lost, and program managers could have been accused of not fulfilling their ethical responsibility to care for the released animals (Waples and Stagoll 1997).

The brush-tailed bettong (*Bettongia penicillata*) has been the most commonly translocated macropod species in Australia, with the majority of releases occurring in Western Australia (Chapter 3). The initial success of these WA translocations, combined with the recovery of remnant populations (primarily through massive, broad scale fox baiting campaigns), led to the removal of the species from both Commonwealth and State threatened species lists (Department of Environment and Conservation n.d.). Ongoing monitoring of population trends was integral to brush-tailed bettong translocations, and included trapping and monitoring of sign (Chapter 3). Despite the initial strong recovery of the species, brush-tailed bettong numbers commenced a rapid decline in 2001 (Wayne 2009). Consequently, the species has been returned to WA, Commonwealth and International Union for the Conservation of Nature threatened species lists (Commonwealth of Australia 2009, Wayne 2009). The Woylie Conservation Research Project was launched in 2006 bringing a range of stakeholders together with the WA Department of Environment and Conservation to investigate the possible causes of bettong decline (Department of Environment and Conservation n.d.). Large scale bettong monitoring, including trapping, is central to this research (Department of Environment

and Conservation n.d.). Health assessments of brush-tailed bettongs trapped near Manjimup WA revealed a high percentage of animals were carrying two particular parasites which may, in combination with other stressors, be contributing to the population crash (Towie 2011). Considering that declines have been recorded in translocated populations established up to 30 years ago (Chapter 3), the history of brush-tailed bettong recovery efforts provides a clear example of the need to monitor population demographics over the long term.

Of 109 Australian macropod translocations conducted between 1969 and 2006 (Chapter 3), data regarding whether or not post-release monitoring had been undertaken was available for 88 programs. All of these translocations completed some form of monitoring after release. Although data regarding the use of monitoring was available for only 81% of translocations investigated, I have no reason to believe that the remaining programs would differ markedly from this trend. Therefore, it appears that post-release monitoring has been considered an important aspect of historic Australian macropod translocations.

In order to gauge the success of the reintroduction of mala to Uluṟu – Kata Tjuṯa National Park (UKTNP), regular trapping was conducted over a period of four years and nine months after the reintroduction of the 24 founders in September 2005. It was considered that an increasing population of robust, healthy, breeding individuals would indicate that the reintroduction has been successful, at least in the short term. In addition to evaluating program success, data from the trapping study also permitted the investigation of population demographics. Once quantified, these characteristics could be compared with the Watarrka National Park mala population, which is also housed within a predator-proof enclosure. These population studies are potentially important to the identification of optimum strategies for future mala translocation and management.

4.2 Methods

4.2.1 Translocation of mala to Uluṟu

Animals translocated to UKTNP were sourced from Watarrka National Park, where a 120 hectare predator-proof enclosure housed the largest population of mainland mala. (The Watarrka National Park population was established from mala translocated in 2000-2001 from a decommissioned enclosure at Lake Surprise in the Tanami Desert.) On the night of September 28 2005, two trapping sessions were conducted by staff from the then Parks and Wildlife Service of the Northern Territory and Environment Australia. 'Bromilow' traps (a treadle operated trap which encloses the animal in a suspended string bag, Figure 4.1, Kinnear et al. 1988) were baited with fresh lucerne and placed in areas of known high mala density within the enclosure. Two clearances of the traps were conducted during the course of the night. Captured mala were weighed, sexed, assessed for breeding and overall condition, and fitted with a small numbered ear tag (Kent Scientific Equipment UK). Those mala considered suitable for translocation were placed in hessian bags, which in turn were placed into commercially manufactured plastic pet transport containers. As female mala are prone to ejecting large pouch young when under stress, only females with small or no pouch young were selected. A total of 24 mala were chosen for translocation; 15 males and 9 females. Although a more even sex ratio was preferred, insufficient suitable females were caught on the night to create an 'ideal' release cohort. The gender inequality present in the founder group was tolerated for two reasons. First, a second release of mala from Watarrka was part of the translocation strategy, and therefore any initial imbalance could be rectified at a later date. Second, the release provided an opportunity to observe how a population of mala develops after being established with a male bias. At 0300 on 29 September 2005, the mala were loaded into three vehicles and transported to UKTNP, arriving at approximately 0630. The majority of animals were released into the 170 hectare enclosure near the supplementary feed stations before 0700 hrs. Several mala were released near the enclosure gate for the benefit of attending stakeholders and media.

4.2.2 Surveys of Uluru population

A total of eight trapping surveys were conducted at Uluru between September 2005 and June 2010. Initially, trapping was conducted every six months to permit close monitoring of the newly established population. Later surveys were spaced at longer time intervals. Early surveys used 'Bromilow' traps (Figure 4.1), and later the 'Thomas trap', a purpose-built small macropod trap using a treadle mechanism to confine animals within a suspended shade cloth bag, was employed (Figure 4.2).

Figure 4.1 Bromilow trap

Figure 4.2 Thomas trap

Traps were placed near the supplementary feed stations and within areas regenerating from management burns. A number of different burnt areas were used for trapping during early surveys, until standardised trap locations were adopted in 2008. Thus, although the same overall closed population was surveyed during each period (the 170 hectare predator-proof enclosure), different areas within the enclosure were trapped prior to 2008. Traps were baited before sunset using various baits including fresh lucerne, chopped apples and chopped carrots. Difficulties in obtaining fresh lucerne lead to exclusive use of apples and carrots in later surveys. Two hours after sunset, the traps were cleared and animals processed. Each mala was removed from the trap and placed in a hessian bag to reduce animal stress and facilitate ease of handling. The animals were weighed, sexed, breeding condition established, ear tag number recorded and a subjective assessment of body condition made. Mala were also inspected for injury or signs of illness. Mala born within the enclosure were marked with small, numbered ear tags (Kent Scientific Equipment, UK). Females with large pouch young were released

without processing in order to minimise the risk of young being ejected from the pouch. After all traps had been cleared, researchers left the area for a period of not less than two hours. The same procedure was followed for a second trap clearing per night, and a third was attempted if low initial captures (and thus processing time) left sufficient hours of darkness to permit a further trap clearance. Between April 2006 and October 2007, eight trapping nights were conducted per survey. In April 2008 free feeding prior to trapping was attempted by UKTNP management in an effort to increase trapping efficiency. This desire to hone trapping method thus resulted in inconsistencies with earlier survey protocols. However, because the primary purpose of the trapping program was to allow UKTNP management to conduct an effective, regular population survey, it was necessary that my project adopt these changes. For four nights prior to the start of the census, the traps were put in place, baited, and wired open. Mala could then enter the trap, consume the bait, and leave freely. A substantial first-night increase in capture rate resulted, which suggested that the technique was successful in familiarising mala with the traps, and increasing their confidence to enter. As a result, only four trapping nights were subsequently undertaken, as the vast majority of individuals were captured over the first two nights of each survey.

4.2.3 Data analysis: Uluru

The CAPTURE program within the MARK analysis package (White 2008) was used to determine the most appropriate model for calculating population estimates from Uluru mala trapping data. Four models were selected for consideration: *Mo* (equal catchability model); *Mb* (trap response model); *Mh* (heterogeneity model); and *Mbh* (heterogeneity and trap response model; Otis et al. 1978, Pollock 2010). The *Mo* and *Mb* models were discounted as they both assume that every animal in the population is equally likely to be trapped. Mala exhibit behaviours which make this assumption improbable, such as varying home range sizes (and therefore likelihood of encountering a trap), and aggressive behaviour towards conspecifics at preferred feeding sites (Otis et al. 1978, Chapters 5 and 7). Further, many species show differing levels of shyness / boldness between individuals (Wolf et al. 2008, Bremner-Harrison et al. 2004). If this phenomenon also occurs within mala populations, assumptions of equal trapability will

be questionable. Using the Uluru mala trapping data, both *Mh* and *Mbh* models were compared within the CAPTURE program to ascertain the most appropriate model to adopt. When data from all six trapping periods were considered individually, the suitability of both models was ranked similarly by the program. However, when initial population estimates were calculated using both *Mh* (Jackknife estimator) and *Mbh* (Pollock and Otto estimator) models within the CAPTURE program, *Mbh* results for two trapping periods returned a 95% confidence interval of zero. Pollock (pers. comm.) has acknowledged the propensity for the *Mbh* model to return such results. When Known To Be Alive (KTBA) calculations (see below) were compared to these two population estimates, they were both revealed to underestimate the population size (one substantially). I therefore decided to use the *Mh* model (Jackknife estimator) to calculate Uluru mala population estimates (Jackknife estimator was chosen to replicate the methodology used to estimate population size at Watarrka National Park). Data regarding the individual mala trapped, and any successive recaptures, for each survey were used to estimate population size, with the exception of March-April 2007 when no mala were recaptured.

The number of mala KTBA after each trapping survey up to and including April 2009 was also determined. This was achieved by comparing data in reverse chronological order, thereby ascertaining that particular animals were alive during past trapping events even though they may not have been caught. For example, an individual released in 2005 may not have been trapped again until 2009, however this animal must have been alive, and passed undetected, during all previous survey events. As the accuracy of the KTBA figure calculated for a particular survey time is dependent upon the number of subsequent surveys completed (Short and Turner 2000), the 2010 figure was excluded as it was likely to be an underestimate. The effective population size of the Uluru founder population was estimated from the trapping history of translocated animals. Simple linear regression analysis was used to determine if a relationship existed between population growth rate (calculated from mark-recapture estimates) and annual rainfall, or between the percentage of females with pouch young and annual rainfall. Rainfall data from Yulara Airport was used, as this is the closest Bureau of Meteorology weather station to the study area

(located approximately 21km to the north; Australian Bureau of Meteorology 2010). The Wilcoxon Rank Sum test (also known as the Mann Whitney U test) was used to determine if a significant difference existed between the number of male and female mala trapped during the survey period. Simple linear regression was used to determine if a relationship existed between the percentage of females with pouch young and population size. Pearson correlation was used to test if an association existed between the percentage of females with pouch young and sex ratio. All statistical analyses were completed using the statistical package PHstat (Levine 2008), with the exception of correlation analyses which were performed with Microsoft Excel 2010 (Microsoft Corporation 2010).

4.2.4 Survey of Watarrka National Park mala population

The Watarrka National Park mala population was surveyed seven times between 2000 and 2009 (Northern Territory Department of Natural Resources, Environment, the Arts and Sport unpublished data). 'Bromilow' traps were used to catch mala during early surveys before the 'Thomas' trap described above was adopted. Trapping effort was focused around supplementary feed stations, and fresh lucerne used to bait traps. Traps were checked up to four times a night, with a minimum of two hours elapsing between each check. After removal from the trap, captured mala were transferred to hessian bags for processing. Animal body mass, sex and breeding condition were recorded, and a subjective rating of overall condition assigned to each individual. For animals which had not previously been caught, a small, numbered ear tag (Kent Scientific Equipment, UK) was applied. Females with large pouch young were released immediately without processing to minimise the chance of pouch young ejection. The *Mh* (heterogeneity) model and jackknife estimator were used within Program CAPTURE (MARK analysis package; White 2008) to calculate estimates of population size (Chris Pavey pers. comm.).

4.2.5 Data analyses: Watarrka National Park

After the 2004 census, several management issues arose at Watarrka National Park which caused a temporary reduction in the mala population. Therefore, as the development of

the population was artificially affected, I have used only data collected at Watarrka between 2000 and 2004 for my comparisons with the Uluru population. I performed Pearson correlation analyses on the Watarrka National Park data to determine if a relationship existed between the percentage of females with pouch young and population size, and the percentage of females with pouch young and sex ratio. I used the Wilcoxon Rank Sum test to investigate whether a significant difference existed between the number of males and females trapped, and between the Uluru and Watarrka populations with regard to the percentage of females caught with pouch young. Correlation analyses were performed with Microsoft Excel 2010 (Microsoft Corporation 2010), and Wilcoxon Rank Sum tests with PHstat (Levine 2008).

4.3 Results

A total of eight trapping surveys of the Uluru mala population were undertaken between 2006 and 2010 (Table 4.1). Total captures per survey ranged from 14 to 128 (average 56, s.d. 42.62), and total individuals trapped from 13 to 51 (average 32, s.d. 16.75). The lowest percentage survey trap success was 8.0%, and the highest 32.7% (average 18, s.d. 8.49; Table 4.1).

Table 4.1 Overview of eight trapping surveys of the Uluru mala population 2006-2010

Date of survey	Total captures	Total individuals	Number of traps per night	Number of trap nights	% trap success
Mar-Apr 2006	17	13	22	176	9.7%
Oct 2006	34	26	22	176	19.3%
Mar-Apr 2007	14	14	22	176	8.0%
Oct 2007	17	15	22	176	9.7%
Apr 2008	128	51	49	392	32.7%
Jan 2009	62	35	49	392	15.8%
Apr 2009	93	47	49	392	23.7%
Jun 2010	85	51	49	392	21.7%

4.3.1 Population estimates

An overall increase in population size occurred between September 2005 and June 2010 (Figure 4.3). Over this period, the population increased from the original 24 founders to an estimated 116 individuals. However, population estimates show that the growth of the Uluru mala population was not constant (Figure 4.3).

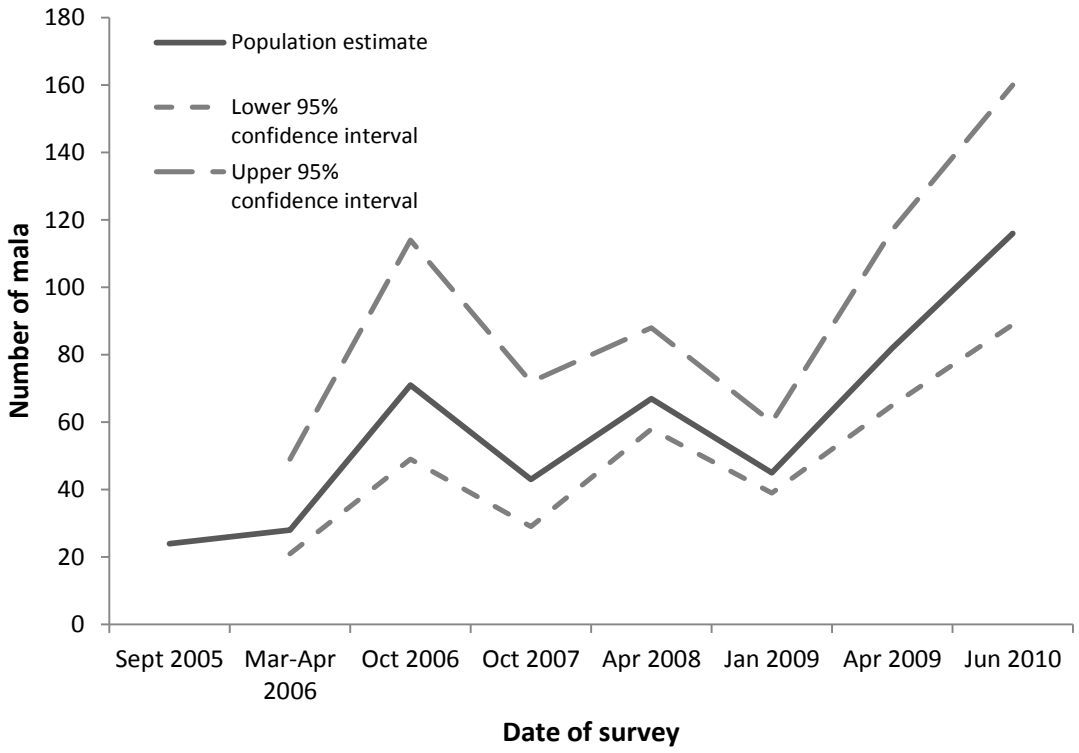

Figure 4.3 Uluru mala population estimate, including upper and lower 95% confidence intervals, for each of the surveys (except March-April 2007) calculated with program CAPTURE

Total animals KTBA for each survey up to and including April 2009 also showed an increase up until October 2007 when 59 animals were known to be alive (Figure 4.4). Subsequent to this survey the KTBA estimate has twice fallen over successive surveys. The April 2009 estimate of 58 is similar to the October 2007 estimate of 59 (Figure 4.4). Discrepancies are clear when mark-recapture estimates are plotted against the number of mala KTBA (Figure 4.5).

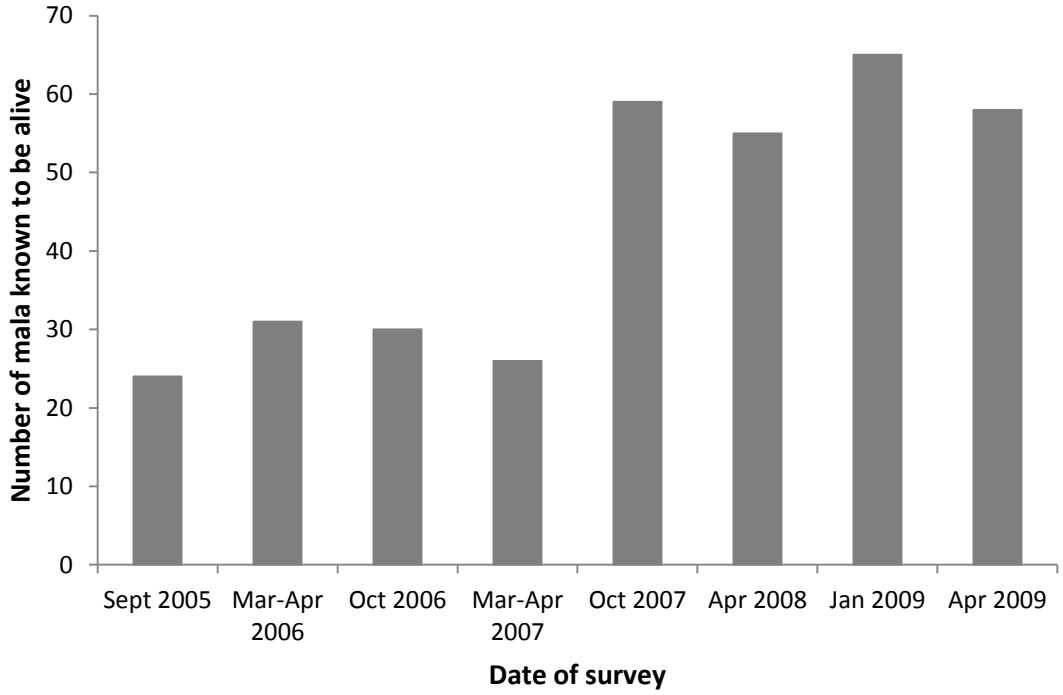

Figure 4.4 Total Uluru mala KTBA for each survey up to and including April 2009

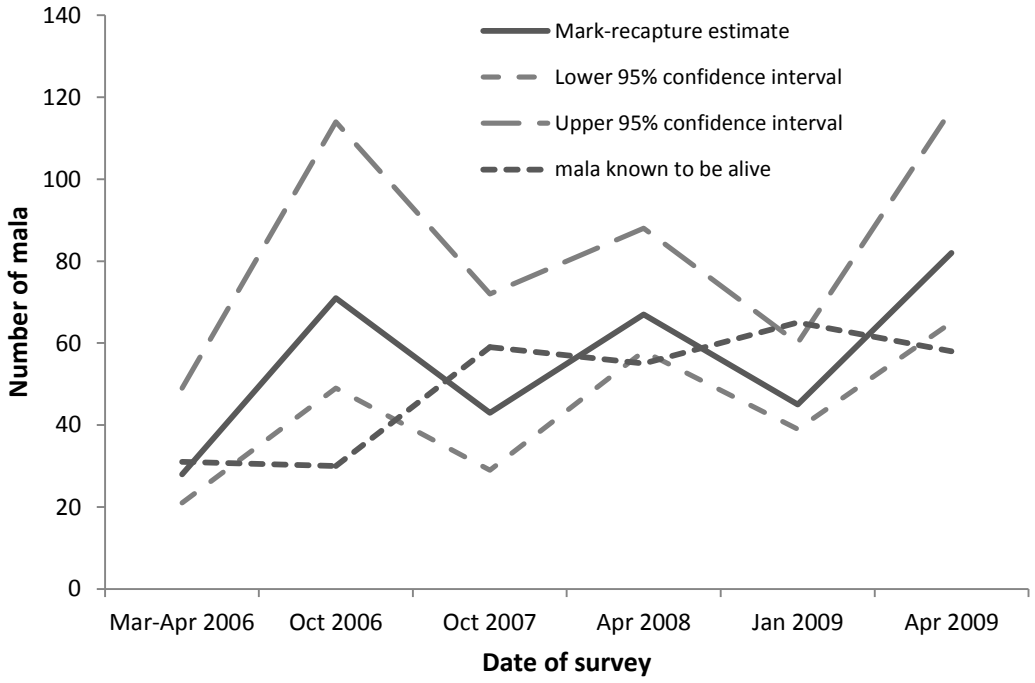

Figure 4.5 Uluru mala population mark – recapture estimates and number of mala KTBA at each survey

4.3.2 Impact of rainfall on population growth

When population growth rate and annual rainfall over the survey period were compared at each survey using simple linear regression (figure 4.6), no relationship between the two variables was detected ($r^2 = 0.04$, $P > 0.05$).

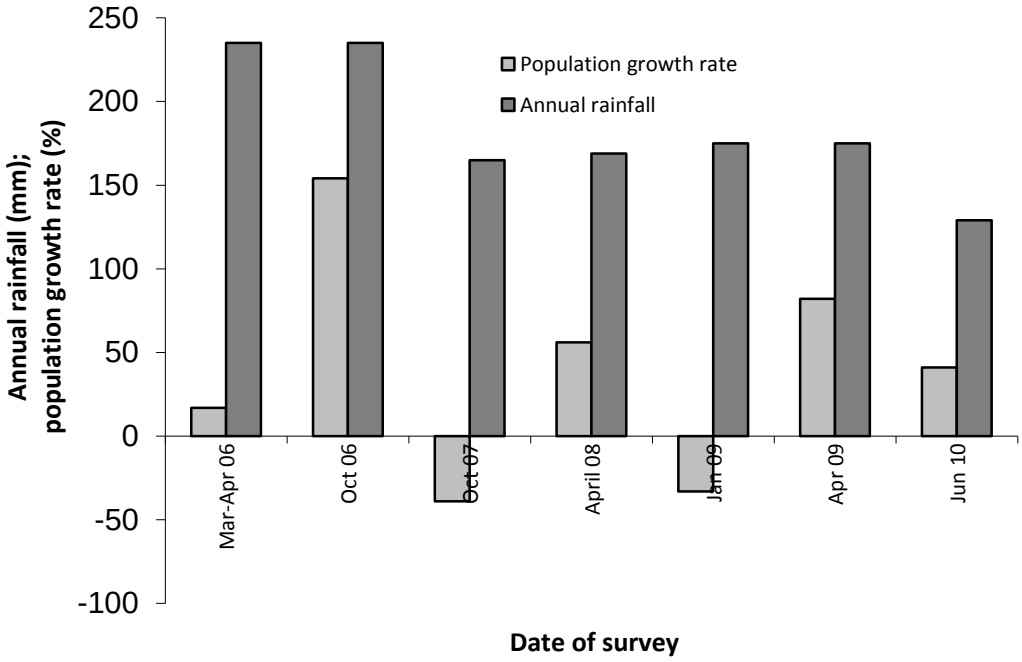

Figure 4.6 Uluru annual rainfall and mala population growth rate for each survey

4.3.3. Fate of Founders

Over 90% of founders were KTBA 12 months after release, and 75% after 24 months (Figure 4.7, Table 4.2). A decline in the number of founders KTBA was observed over the course of the study period (Figure 4.7, Table 4.2), with 24% recorded in 2010. Two of the original founders were not caught after their initial release. Of the 13 mala recaptured six months after release, 9 (69%) had sustained their release mass or gained mass (Table 4.2). One female (335) carrying a large offspring was released without being weighed to lessen the chance of pouch young ejection. Seventeen of the founders (71%) had sustained or gained mass between release and their final capture date (Table 4.2). Mala 348 lost weight with each successive trapping, and was not captured again

after April 2008. Six of the nine females reintroduced to Uluru are known to have produced pouch young after release (Table 4.2). Of the females that are known to have produced pouch young, three mala had three one offspring, two animals had two offspring, and one female produced seven pouch young.

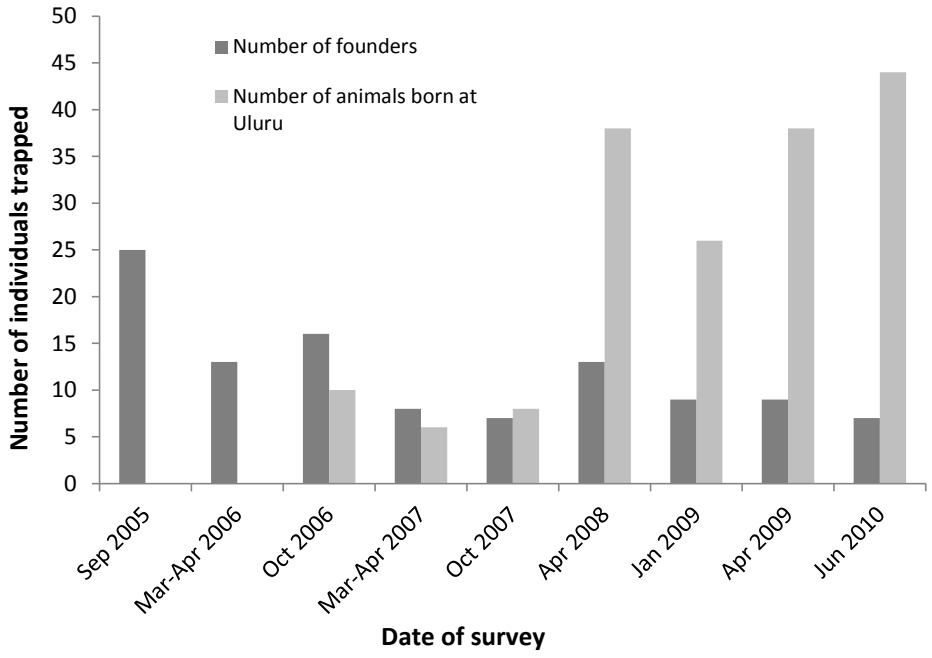

Figure 4.7 Number of Uluru mala population founders, and mala born within the Uluru enclosure, trapped during each survey

Table 4.2 Capture history of Uluru mala population founders. Mass of animal at time of capture shown in kilograms, and presence of pouch young indicated by letters 'py'. Dash indicates animal was not trapped during the survey period

Mala	Sex	Sep 2005	Mar-Apr 2006	Oct 06	Mar-Apr 2007	Oct 2007	Apr 2008	Jan 2009	Apr 2009	Jun 2010	Change in mass between release and final date of capture
452	F	1.50	-	1.50 py	1.35	-	1.60	-	-	-	+ 0.10 (7%)
341	F	1.50 py	-	-	1.40 py	-	1.60	1.80	1.75	1.70	+ 0.20 (13%)
530	M	1.40	1.35	1.45	-	-	-	-	-	-	+ 0.05 (4%)
785	F	1.35 py	1.35	-	-	1.50	-	-	-	-	+ 0.15 (11%)
779	M	1.30	1.40	1.40	-	-	1.50	-	-	-	+ 0.20 (15%)
794	M	1.30	1.45	-	-	-	-	-	-	-	+ 0.15 (12%)
347	M	1.10	1.60	1.30	1.60	1.35	1.05	-	-	-	- 0.05 (5%)
335	F	1.40	? py	1.60 py	-	1.60 py	1.50 py	1.65 py	1.55 py	1.80 py	+ 0.40 (29%)
545	M	1.50	1.60	1.65	-	-	-	-	-	-	+ 0.15 (10%)
323	M	1.40	1.40	1.35	-	-	1.30	1.55	1.40	-	0
795	F	1.30	-	1.20	1.75 py	-	-	-	1.65 py	1.55	+ 0.25 (19%)
777	F	1.25	-	-	1.15	1.15		-	-	-	- 0.10 (8%)
560	M	1.25	-	1.35	1.25	-	1.40	1.45	1.40	1.35	+ 0.10 (8%)
792	M	1.30	-	-	-	-	-	1.55	1.60	-	+ 0.30 (23%)
581	M	1.45	1.40	1.55	-	-	1.6	1.8	-	-	+ 0.40 (28%)
322	F	1.40	-	1.85	1.75	-	1.90	2.0	-	-	+ 0.60 (43%)
778	M	1.15	1.25	1.30	-	-	1.45	-	1.45	1.45	+ 0.30 (26%)
583	F	1.65 py	1.65 py	-	1.70	1.80	1.60 py	-	-	1.45	- 0.20 (12%)
384	M	1.40	1.35	1.50	-	1.55	-	1.70	1.55	-	+ 0.15 (11%)
781	M	1.25	-	-	-	-	-	-	-	-	-
791	M	1.30	1.45	1.50	-	1.50	1.50	1.50	1.55	1.55	+ 0.25 (19%)
348	M	1.35	-	1.10	-	-	0.90	-	-	-	- 0.45 (33%)
369	F	1.20	-	-	-	-	-	-	-	-	-
334	M	1.35	-	1.25	-	-	-	-	-	-	- 0.10 (7%)

4.3.4 Sex ratio

With the exception of March-April 2007, consistently higher numbers of male mala were captured during each survey than females (Figure 4.8). However, this difference between the absolute number of male and female captures over the course of the study period was not significant (Two-tailed Wilcoxon Rank Sum test, Z = 1.208, p = 0.227).

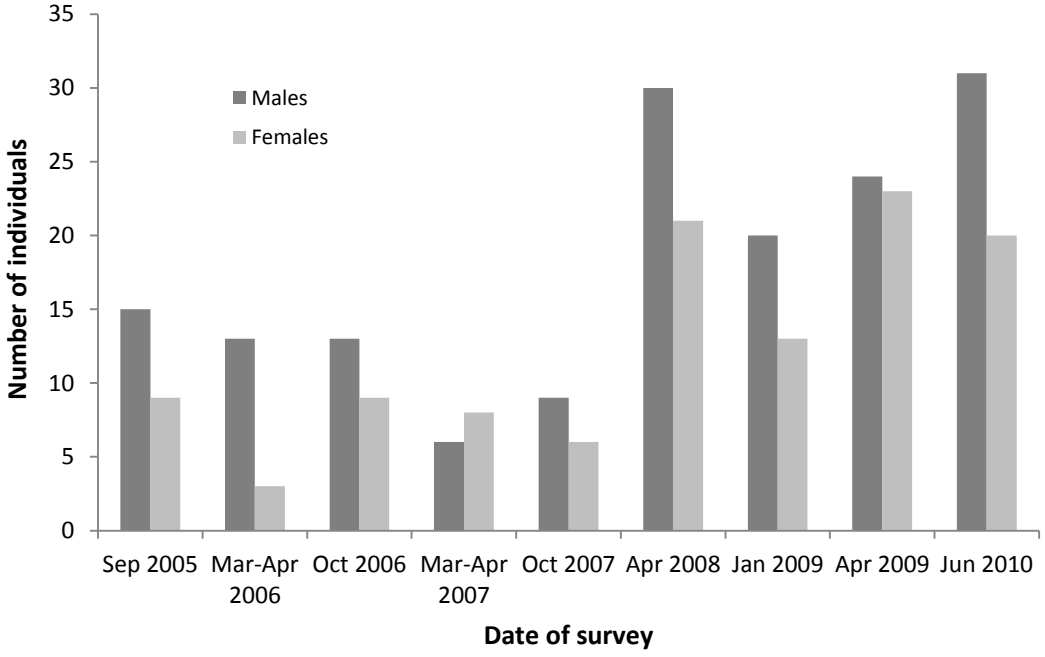

Figure 4.8 Number of individual male and female mala trapped at Uluru during each survey

4.3.5 Reproduction

Over the eight surveys, between 38% and 100% (average 66%, s.d. 20.7) of females caught at Uluru had pouch young (Figure 4.9). The result of regression analysis of survey results found no statistically significant relationship between the percentage of females with pouch young and population size ($r^2 = 0.320$, F = 2.348, d.f.= 1,5, p = 0.186).

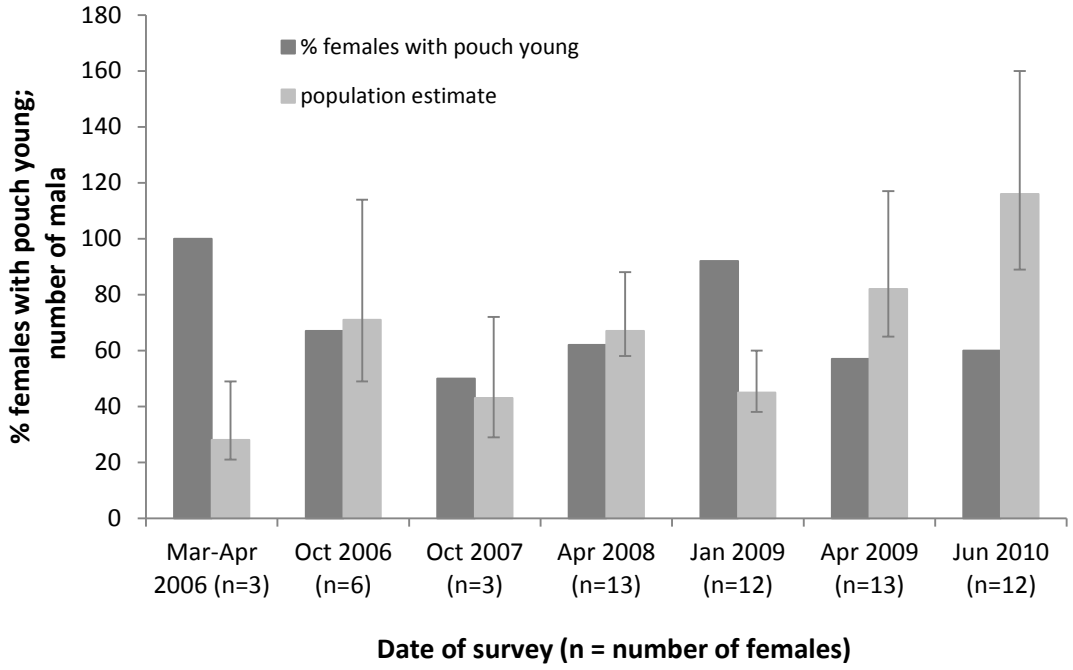

Figure 4.9 Percentage of Uluru female mala trapped with pouch young and mark-recapture population estimates during each survey. Error bars show lower and upper 95% confidence intervals for population estimates

Comparison of the percentage of females trapped with pouch young, and a three- and six-month lag in rainfall recorded prior to the survey (Figure 4.10) using regression analysis showed that the amount of rainfall received in the three months leading up to a survey was a significant explanatory variable ($r^2 = 0.685$) for the percentage of females carrying pouch young ($F = 13.032$; d.f. = 1,6; $p = 0.011$). There was a weak relationship between the percentage of females with pouch young and rainfall received in the previous 6 months leading up to the survey ($r^2 = 0.48$) that was at the margin of significance ($F = 5.58$; d.f. = 1,6; $p = 0.056$).

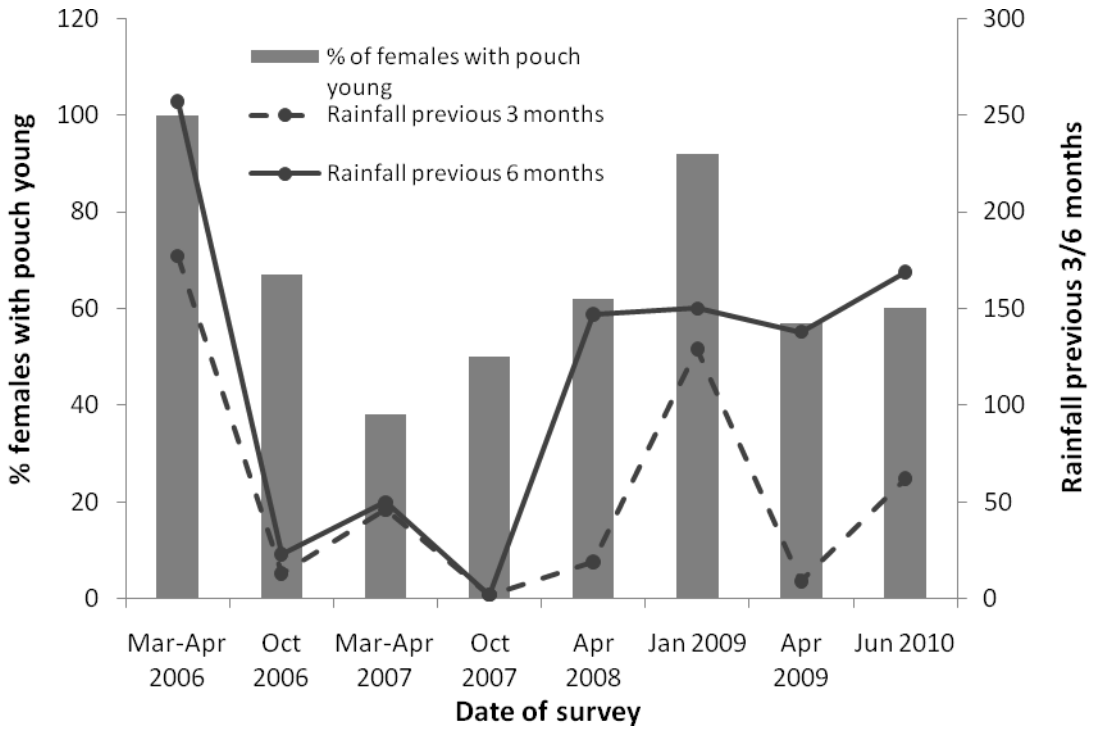

Figure 4.10 Percentage of Uluṟu females trapped with pouch young, and rainfall for the the 3/6 months before date of survey

There was a significant relationship between the percentage of females with pouch young and sex ratio ($r = 0.77$, $t = 2.96$, d.f. = 6, $n = 8$). As male bias in the sex ratio increased, the percentage of females with offspring increased (Table 4.3).

Table 4.3 Percentage of Uluṟu female mala with pouch young, and male to female ratio for each survey date. Sample size shown in brackets

Date of survey	**Mar-Apr 06**	**Oct 06**	**Mar-Apr 07**	**Oct 07**	**Apr 08**	**Jan 09**	**Apr 09**	**Jun 10**
% fem. pouch young	100 (n=3)	67 (n=6)	38 (n=8)	50 (n=3)	62 (n=13)	92 (n=12)	57 (n=13)	60 (n=12)
Male to female ratio	4.33	1.44	0.75	1.5	1.43	1.54	1.04	1.55

4.3.6 Watarrka National Park data

The relatively large founder population released at Watarrka National Park (n = 81) recorded an increase of 220% in the first three years after release (Figure 4.11).

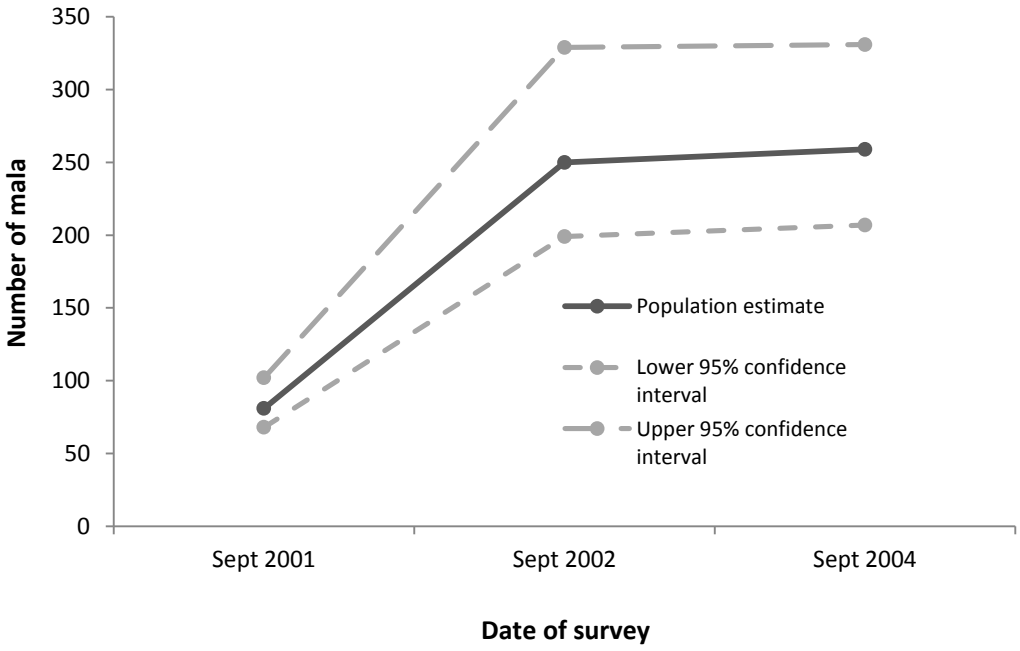

Figure 4.11 Watarrka mala population estimate for the first three years post-release, including upper and lower 95% confidence intervals calculated with program CAPTURE

On average, 56% (s.d. 9.45) of mala caught during trapping surveys at Watarrka National Park between 2001 and 2004 had pouch young. When compared with Uluru trapping results between 2006 and 2010, there was no significant difference between the percentage of females with pouch young in either population (Two-tailed Wilcoxon Rank Sum test, Z = -0.594, p = 0.552).

Although variation was observed between the percentage of females with pouch young at different population sizes at Watarrka (Figure 4.12), no statistically significant correlation was found between these two variables (r = -0.34, t = -0.36, d.f. = 1, n = 3).

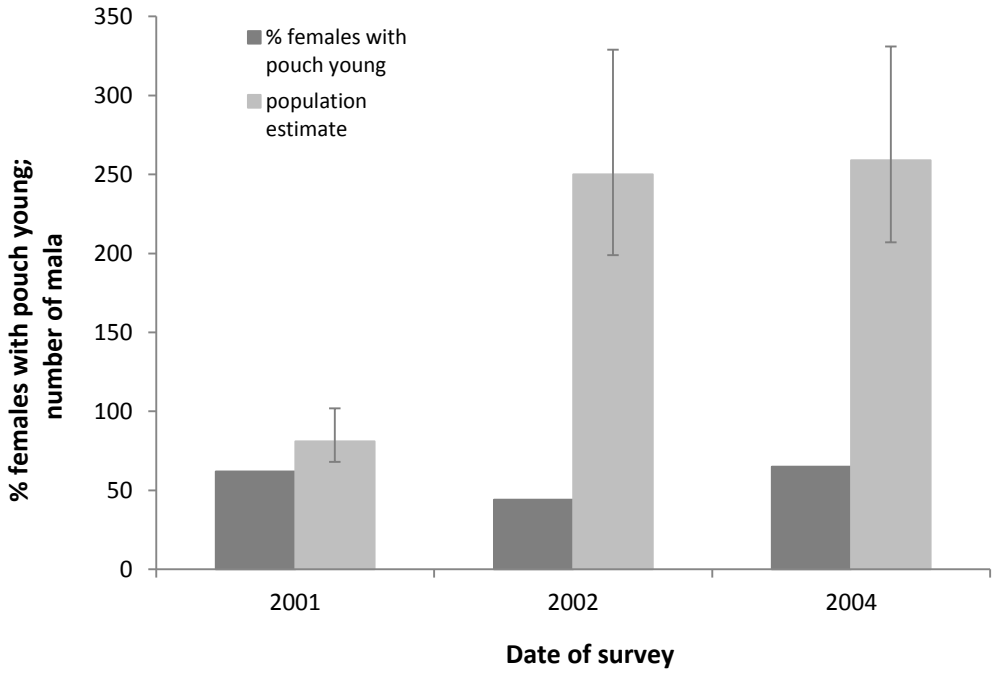

Figure 4.12 Percentage of female mala trapped with pouch young and mark-recapture population estimate for each survey date, Watarrka National Park. Error bars show lower and upper 95% confidence intervals for population estimates

Watarrka survey data showed a variation in the percentage of females with pouch young at different gender ratios (Table 4.4). However, correlation analysis showed no statistically significant relationship between these two variables ($r = 0.841$, $t = 1.256$, d.f. $= 2$, $n = 4$), and no significant difference between the number of males and females trapped (two-tailed Wilcoxon Rank Sum Test, $Z = 0.577$, $p = 0.564$).

Table 4.4 Percentage of Watarrka female mala with pouch young and male to female ratio for each survey date

Date of survey	**2001**	**2002**	**2003**	**2004**
% fem. pouch young	62	44	53	65
Male to female ratio	1.10	0.87	1.17	1.26

4.4. Discussion

4.4.1 Population growth

An important caveat when interpreting the results of this demographic study is that the impact on population variables of several changes in methodology during the course of the sampling is unknown. Specifically, the trap type used changed from the 'Bromilow' to the 'Thomas' trap design, and free feeding of traps became part of the sampling protocol part-way through the study. Further, for the comparison of population data between Uluru and Watarrka, trap type changed at the same time at both locations however free feeding was not used at Watarrka.

The increase in mala numbers at Uluru from 24 founders in September 2005 to an estimated 116 animals in June 2010 indicates the reintroduction has achieved short term success. This conclusion is supported by the high survivorship of the founders (Table 4.2), and the health and breeding condition of the animals trapped during each survey period. I argue that the provision of supplementary food and water, a lack of mammalian predators, and (an assumed) availability of sufficient space to provide for individual needs (refer Chapter 5), provided consistently good breeding conditions for mala. A linear or exponential growth pattern may therefore be expected, however each survey period did not show an increase in mala numbers. In fact, population decreases were recorded over three trapping periods (Figure 4.3), which could indicate that the Uluru mala population did indeed fluctuate during the study.

A decline in the mala population would require not only zero recruitment over the previous months since the preceding trapping study, but also deaths within the population (no emigration could have occurred due to the enclosed nature of the population). However, as the number of trapped females with pouch young ranged from 38 - 100% over the course of the study period (Figure 4.8), it would be surprising if a lack of recruitment could explain the population decreases observed in the mark-recapture estimates. Further, no sign of disease was found during veterinary inspections of mala during trapping. In addition, although no targeted search was undertaken, during the greater than 750 hours I spent in the mala enclosure undertaking fieldwork and

management activities during the study period less than ten carcasses were found. As discussed above, supplementary food and water were provided over the course of the study period, thus eliminating drought as a cause of the apparent decline. Although old age may have resulted in the deaths of a small number of the founder animals, this would not have been sufficient to cause the apparent decreases. In addition, there has been no bushfire within the enclosure; the only phenomenon capable of killing mala swiftly on a relatively large scale. Further evidence of an underestimation of population size, rather than an indication of true decreases, can be found when mark-recapture calculations are compared with KTBA numbers (Figure 4.5). The number of mala KTBA across the study period provides an actual, minimum population number. Contrary to the mark-recapture estimate, the KTBA number shows an increase between October 2006 and October 2007. Further, the KTBA minimum population shows an increase in mala over the period April 2008 to January 2009, compared to a 22 animal decrease from mark-recapture estimates. Although small decreases were observed from KTBA estimates in the latter survey period, these are likely to have resulted from a combination of chance, the release of females prior to processing (to avoid the ejection of pouch young), and the occasional escapee during trapping/handling. Further, the tapering of the KTBA minimum population size towards the latter part of the survey period may be explained by a saturation of mala density around the supplementary food and water points. That is, a similar number of animals were captured during each survey. This explanation will be tested in the future when surveys are planned to include a greater proportion of the overall enclosure area. I believe, therefore, that my data indicate that the mala population within the entire 170 hectare enclosure increased throughout the entire study period.

4.4.2 Effective population size of release cohort and Uluru mala population gender ratio

Trapping data permit the crude estimation of the effective population size of the Uluru founder population. Three females within the release cohort were carrying pouch young when translocated. If these offspring survived and bred, and other founders also bred successfully, the maximum effective population size would have been 27. However, two mala were not re-caught after release, and may have died before they could breed.

Further, another two females were not recorded with pouch young during trapping surveys. If these four animals did not contribute to further generations, and the three pouch young carried by females on release did not survive, the effective population size may have been as low as 20 (assuming all males bred successfully). As no genetic study was attempted, the number of male founders contributing to successive generation is unknown, and thus the effective population size may have been quite small indeed. The possible consequences of the small Uluru founder population on long-term genetic health are discussed in section 4.4.6 below.

As no significant difference was found between the number of male and female mala captured over the study period, it appears that the skewed gender ratio present in the founder population has moved towards equilibrium. An unequal gender ratio is likely to exacerbate the genetic bottleneck present at the foundation of a new translocated population (Dufty et. al 1994). Considering that not all female mala released may have bred, this may well have been the case at Uluru. However, mala returned to gender equilibrium relatively quickly, thus avoiding perpetuating this potential problem over the long-term. The data also suggests that there is no difference in the level of 'boldness' between males and females with regard to their likelihood of entering traps.

4.4.3 Pouch young and rainfall

Rainfall induced food resource increases have been identified as a major determinant of reproductive rates in both large and small arid/semi-arid Australian macropods (Bayliss 1985, Morgan and Peglar 2010, Robinson et al. 1994, Short et al. 1997, Tenhumberg et al. 2004) including mala (Lundie-Jenkins 1998, Short et al. 1997). Presumably, if a spatially restricted population is experiencing sub-optimal growth due to limited food resources, the provision of palatable supplementary food will produce an increase in population size due to higher reproductive and survival rates (Predavec 2000, Treby et al. 2007). Supplementary food and water are provided within the mala enclosure at Uluru in an effort to provide conditions conducive to rapid population growth regardless of natural climatic flux. If this management action is effective, one would expect to find no causal relationship between rainfall and the percentage of females with pouch young. However,

a statistically significant relationship was found between pouch young present and rainfall in the three months prior to the survey date. Although not tested statistically, Lundie-Jenkins (1998:94) similarly suggested that a relationship may exist between major rainfall events and spikes in recruitment to the reintroduced Tanami Desert population, which also received supplementary feed.

There are only two published experimental studies of the effect of supplementary food provision on Australian arid zone mammal population dynamics. In his study of two Australian arid zone rodents, Predavec (2000) found only a weak, short-term positive numerical response to supplementary food provision in *Pseudomys hermannsburgensis*, and no response in numbers of *Notomys alexis*. Predavec (2000) offered two possible explanations for this result. First, considering that both study species are nomadic, the size of the treatment area may have been too small to account for animal movements. Second, the supplementary food may not have contained all the necessary nutritional requirements to promote population growth (Predavec 2000). Further, the presence of water in addition to abundant food may be required to trigger significant breeding events in the two study species (Predavec 2000). Brown et al. (2008) provided food and water supplements to house mice (*Mus musculus*) study sites located in semi-arid Victorian agricultural land. The treatment resulted in a very limited impact on mouse population growth and size (Brown et al. 2008). Although dispersal from the study area or the removal of mice by predators may have explained the lack of mouse population increase, the authors concluded that a limiting factor(s) other than food and water restricted mouse numbers during the study period.

The fact that a significant relationship existed between the presence of Uluru mala pouch young and rainfall in the three months prior to the surveys suggests that the supplementary feed provided did not fully compensate for times of lower rainfall and thus lower wild food availability. At least two explanations for the relationship between the presence of pouch young and rainfall are plausible. First, that the supplementary food is not providing all the necessary nutritional requirements to sustain high levels of breeding (Predavec 2000). Alternatively, breeding and births in mala are triggered by rain events and consequent condition of wild food plants. Second, insufficient mala

within the trapping area are using the food and water supplements to significantly affect the number of pouch young present in the trapped samples. Although mala home range studies at Uluru showed that not all mala visited the supplements (Chapter 5), dietary investigation showed that the supplementary feed contributed significantly to overall mala diet. Further, trapping was concentrated in the vicinity of the feed and water stations, and therefore individuals consuming supplements were more likely to be caught. Thus, the first hypothesis appears more likely to explain the relationship between the presence of pouch young and rainfall in the previous three months regardless of the presence of supplementary feed and water. The fact that no statistically significant relationship was found between the presence of pouch young and rainfall six months prior to the survey date may indicate that the time period is too long to identify mala breeding and birthing changes resultant from such rain events. Future experiments would be required to form robust conclusions regarding the relationship between rainfall and the percentage of female mala with pouch young.

4.4.4 Pouch young and sex ratio

Polygyny, where a male mates with several females (Krebs 1994), is common among macropod species (Croft 1989, Jarman 1991). Behavioural observations suggests mala employ this mating system (Lundie-Jenkins 1993). In a relatively small, closed, male-biased population of a polygynous species, one may assume that each receptive female would have a number of potential breeding partners. Although some males may be excluded from breeding by more dominant animals, a high rate of female breeding would be expected in favourable environmental conditions. The positive correlation identified at Uluru between the ratio of males to females trapped and the percentage of females with pouch young must be considered in light of there being no significant difference found between the number of males and females in the population (Figure 4.7). However, if such a bias was representative of the population as a whole, this significant positive correlation could arguably be expected.

4.4.5 Comparison of Uluru and Watarrka mala populations

The primary aim of threatened species translocations is the establishment of (or in the case of an augmentation, assisting in the preservation of) self-sustaining wild populations. Therefore, a sufficient number of animals must be released to ensure this goal is achieved.

However threatened species, by their nature, exist only in small numbers, and consequently releases will typically involve relatively small groups of animals (Stanley Price 1991). These fledgling populations are susceptible to genetic, demographic and environmental extinction pressures. Both the Uluru and Watarrka mala populations experienced rapid population increase after founder release. Both populations have, thus far, avoided succumbing to either demographic or environmental threats. The fact that Uluru mala are sheltered from a number of these extinction pressures through intensive management (fire prevention, exclusion of predators, provision of supplementary food and water) is no doubt responsible for the program's short-term success. It appears that founder group size, at least with a minimum of 24 individuals, is not critical to the initial successful establishment of translocated mala populations in predator free areas. However, as discussed in Chapter 2, long-term problems may result from the release of a small number of founders. A consequence of small release cohorts may be low genetic diversity and matings between closely related animals, resulting in inbreeding depression. Such low genetic variation will reduce fitness, may minimize the chance of adaptation to future environmental change, and ultimately endanger the establishment of a viable population (Miller et al. 1999, Sarre and Georges 2009, Sigg 2006, Smith and Hughes 2008). Franklin (1980) proposed two minimum effective population sizes for both short and long-term population viability. A minimum effective population size of 50 was suggested for immediate population survival, and 500 individuals to ensure the maintenance of genetic variability in the long-term. However, it was acknowledged that these figures were initial estimates, and that future study would refine minimum effective population sizes (Franklin 1980). The Uluru founder population numbered 24 individuals, and effective population size may have been less than 20; less than half the estimated number suggested by Franklin (1980) as necessary for short-term survival. The

release of 81 mala at Watarrka, where management is similar to that at Uluru, comfortably exceeds the minimum proposed by Franklin (1980) for short-term persistence. However, both founder groups are very much smaller than Franklin's (1980) suggested 500 individuals required to offset potential genetic issues in the long term. The conservation of mala in Australia is managed by the national Mala Recovery Team, comprised of representatives from a variety of agencies involved in attempts to preserve the species. A central objective of the Recovery Team is to manage the existing disjunct mala populations as a single metapopulation, primarily to address concerns regarding the genetic health of the species (pers. obs.). Genetic analysis of the individual populations will be used to guide the strategic movement of animals between facilities to maximise genetic variation within the metapopulation (pers obs). It is hoped that this action will alleviate genetic extinction pressures acting on both the Uluru and Watarrka mala populations as a consequence of their relatively small founder sizes.

4.4.6 Management implications

My study was undertaken, in part, to identify optimum strategies for future mala translocation and management. As mala translocation success in arid/semi-arid mainland Australia is reliant on the use of predator-proof enclosures (Chapter 3), the following recommendations are made for this type of program (although recommendations may also be applicable to translocations to predator-free offshore islands). The Uluru mala reintroduction has shown that short-term translocation success can be achieved with a relatively small, male-biased release cohort. Managers of future mala translocations may therefore consider releasing similar founder groups with confidence. However, the potential long-term genetic issues associated with small founder groups and unbalanced gender ratios will still need to be addressed, presumably through managing individual mala colonies as a metapopulation. I recommend that initial mala population monitoring should occur every six months after release. This provides the potential to identify threats to translocation success when the population is at its most vulnerable, that is at low population numbers. When managers are satisfied that the population has established in the short term, interval periods between monitoring may be extended. Determining the number of mala KTBA at each monitoring period may be an important

addition to mark-recapture calculations. Considering both these figures may permit program managers to better assess population trends.

My study did not attempt to calculate the impact of supplementary food and water provision on mala population increase, so specific experiments would be required to test the presumed benefits of supplements to population growth. However, dietary analysis (Chapter 6) has shown that mala are using the supplementary food provided, and that for some animals it may contribute a substantial proportion of their diet. Further, if the source population is provided with supplements (as was the case with the Watarrka National Park mala population in the Uluṟu reintroduction example) similar provisioning at the release site will ensure an abundant, recognizable food source for the release cohort. Considering the minimal cost of food and water supplement infrastructure, and its installation and maintenance, I recommend that future mala translocations provide this form of post-release assistance.

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Chapter 5. A study of the home range of mala at Uluṟu – Kata Tjuṯa National Park

5.1 Introduction

Attempts to reintroduce mammalian species to the Australian arid/semi-arid zone have been thwarted by an inability to control introduced predators (Bellchambers 2001, Christensen and Burrows 1994, Langford 2000, Short et. al 1992, Priddel and Wheeler 2004, Southgate 1994, Chapter 3 this thesis). As a result, recent conservation efforts have often focused upon the construction of specially designed enclosures that permit the release of animals into predator-proof refuges. However, the spatial restriction of animals necessitates the monitoring of a population with respect to enclosure carrying capacity. Failure to identify when a population has exceeded carrying capacity can result in animal welfare issues including poor individual health, intraspecies aggression and ultimately the starvation of animals (Bulinski et al. 1997, Carlstead and Shepherdson 1994, Lebarbenchon et al. 2007, Mike Sutherland pers. com.).

Radio tracking is an important post-release monitoring technique that can be used to gain a variety of information critical to the assessment of translocation success. Such data include the dispersal patterns of the founder group, behaviour of animals under different management regimes, assessment of release protocols, post-release survival and cause of death of individuals, habitat use by founder animals and suitability of the release site environment (Dunwoody et. al 2009, Hardman and Moro 2006a, Hardman and Moro 2006b, Lundie-Jenkins 1998, Moseby and O'Donnell 2003, Short et. al 1992). Radio tracking can also be used to estimate the home range size of animals released into predator-proof enclosures. Such estimates, coupled with other measurable variables such as body condition and reproductive status, could be useful in determining carrying capacity (Finlayson and Moseby 2004). A radio tracking study comparing home range

sizes of reintroduced burrowing bettongs (*Bettongia lesueur*) maintained at different densities was completed within predator-proof enclosures at Roxby Downs in South Australia (Finlayson and Moseby 2004). Finlayson and Moseby (2004:462) concluded that although reproductive rate may be a superior gauge of population size with respect to carrying capacity, behavioural parameters such as home range size may also provide useful information.

The failure of previous mala translocations to the wild (Langford 1999, Langford 2000, Lundie-Jenkins 1998) dictated that the reintroduction of the species to Uluṟu – Kata Tjuṯa National Park would be restricted to a predator-proof enclosure. The growing Uluṟu mala population is less than six years old, and does not appear to have reached the carrying capacity of the 170 hectare enclosure into which the animals were reintroduced. I therefore had the opportunity to calculate mala home range size at both low and higher population densities. Radio tracking at Uluṟu also provided the opportunity to test assumptions regarding the benefits of large, predator-proof enclosures over small, intensively managed captive breeding facilities. That is, it could be argued that large, predator-proof enclosures provide a more 'natural' environment for threatened species as they permit animals to range freely, browse natural foods, and interact with conspecifics and other taxa without restriction. If mala home ranges were found to be similar within populations reintroduced to predator-proof enclosures as those released to the wild (whilst being mindful of long and short term climatic differences and floristic variation between release sites), this would support the assumption that large enclosures provide for the species' natural ranging requirements. However, if mala home ranges within predator-proof enclosures were significantly different to those living in the wild, this assumption would be challenged. Home range data collected from mala reintroduced to Sangsters Bore in the Northern Territory (Lundie-Jenkins 1998) and Peron Peninsula in Western Australia (Hardman 2006) presented the opportunity to compare the two situations. This study also permitted the investigation of the importance of supplementary feed provision to such semi-captive populations. Feed has been made available since the release of mala at Uluṟu for two reasons. First, supplementary feed was provided at Watarrka National Park (the source of the Uluṟu founder population),

and it was deemed prudent to provide similar facilities at Uluru to cater for animals which may have become dependent upon artificial feed. Second, it is assumed that the provision of abundant food contributes to higher breeding rates amongst the population. If radio tracking showed study animals living at high density close to the feeders, and feeding primarily on supplementary food rather than browsing in the remainder of the enclosure, it could be argued that the predator-proof enclosure is no more 'natural' than a small, intensive breeding facility. Arguments could then be raised for the removal of supplementary feeders in an effort to preserve and promote free-browsing behaviour, or the adoption of intensive breeding facilities if these prove to cover all management requirements at lower cost. However, if tracking studies showed animals using broad home ranges which may, or may not, include the feed stations, then it could be concluded that predator-proof enclosures may provide a sufficiently 'natural' environment to promote feeding behaviour akin to wild-living mala.

This radio tracking study of mala at Uluru was therefore undertaken with three primary aims. First, to estimate home range sizes at both low and higher population densities in an effort to monitor enclosure carrying capacity. Second, to discover if differences exist between mala ranging behaviour within the Uluru enclosure and that exhibited by mala reintroduced to the wild. Third, to ascertain the importance of supplementary feed stations to mala, in order to gather information regarding mala foraging behaviour.

5.2 Methods

5.2.1 Background to the study

Commencing in 1989, captive bred mala were reintroduced to Sangsters Bore by the (then) Conservation Commission of the Northern Territory (CCNT) in an effort to re-establish the species in the wild (Lundie-Jenkins 1998). As part of this program, 51 animals were fitted with radio collars to examine mala dispersal post-release (Lundie-Jenkins 1998). Home range estimates for study animals released at Sangsters Bore were calculated within the first six months of release, 6-12 months after release (n = 10), and where possible again after 12-18 months (n = 5, Lundie-Jenkins 1998:113). Radio-towers equipped with antennae, receiver units and compasses were used to collect

locational fixes through triangulation. Further, squats (nests) were located during the day on foot and recorded using a compass and Global Positioning System (GPS) unit (Lundie-Jenkins (1998). Diurnal location of mala also enabled the accuracy of triangulation estimates to be checked (Lundie-Jenkins 1998). The Incremental Area Analysis (IAA) function of the RangesIV software package (Kenward 1990) was used to determine the minimum number of fixes required to make a confident estimate of home range size (Lundie-Jenkins 1998). IAA plots the increase in range size provided by each additional location fix, and an asymptote is reached when no area increase is achieved by the provision of further points (Lundie-Jenkins 1993). Fixes obtained for each animal were randomly sorted and combined to provide an overall estimate of the minimum number of fixes required. Harmonic Mean Contours (HMC) were used to calculate home range estimates (Lundie-Jenkins 1993).

In 2001, an experimental reintroduction of mala (and also the banded hare-wallaby *Lagostrophus fasciatus*) was undertaken on Peron Peninsula (Hardman 2006, Hardman and Moro 2006a, Hardman and Moro 2006b). This translocation program was conducted to test the importance of release protocol for hare-wallabies, investigate the importance of nesting sites, and determine if the release site environment was able to sustain reintroduced animals after many years of grazing by introduced species (Hardman 2006, Hardman and Moro 2006a, Hardman and Moro 2006b). As the gathering of mala home range data was central to the program, the sixteen captive bred mala were fitted with radio collars prior to release (Hardman 2006, Hardman and Moro 2006a, Hardman and Moro 2006b). Two methods were used to collect location fixes for reintroduced mala over three tracking periods. First, antennae, receiver units and compasses were used by researchers on foot to determine nocturnal ‘active’ fixes by bi- or tri-angulation (Hardman 2006). Second, after individual animals were tracked to their diurnal squat sites using an antenna and receiver, a GPS unit was used to determine location. The IAA function of RangesV software package (Kenward and Hodder 1996) was used to determine the number of fixes required to obtain a confident estimate of home range size (Hardman 2006). Only mala for which sufficient fixes could be obtained were used in

data analyses. The Minimum Convex Polygon and fixed kernel methods using 95% of fixes were used to calculate mala home range estimates in RangesV.

The Uluru mala colony was founded with animals sourced from Watarrka National Park. A description of the reintroduction is provided in Chapter 4.

5.2.2 General Methods: Uluru

Throughout this chapter, the term 'home range' broadly follows the definition of Burt (1943:351): '…that area traversed by the individual in its normal activities of food gathering, mating, and caring for young'. Ideally, the radio tracking study undertaken at Uluru should use the same methods as those of previous studies in order to permit accurate comparisons of results. However, for several reasons it was decided that location fixes would be acquired on foot rather than through the use of set receiving points. Economic and logistical factors limited the use of triangulation towers, and it was considered that fixes obtained through physically tracking the study animals to their location would be more accurate. Further, tracking animals on foot permitted me to check if the radio collars were negatively affecting individuals. In addition, it was hoped that observations of the collared animals would contribute to the behavioural component of the study (refer Chapter 6).

Radio tracking was conducted during two study periods. First, in order to determine home range estimates at low population density, two males and one female were tracked during July-August 2007, and a further 2 females and a male in June-July 2008. Following two years of population growth, a second period of tracking was conducted in June-August 2010, where a further four males and two females were collared. Before each tracking period, mala were trapped during routine population monitoring using treadle traps (Thomas trap, Western Australia), transferred to hessian bags, and fitted with radio collars (Titley Scientific, Ballina, New South Wales). Only adult mala were collared to ensure collar fit remained relatively constant. The desire to collar equal numbers of males and females was thwarted by my inability to trap suitable females; that is mature animals without large pouch young. All tracking was done on foot using a

portable receiver unit (Telonics TR-4, Telonics Incorporated, Arizona; Titley Australis 26k, Titley Scientific, Ballina, New South Wales) and three element Yagi antenna (Titley Scientific, Ballina, New South Wales). Animals were tracked to within an estimated 30 metres of their position, then circled in ever decreasing loops until their location could be identified to within approximately 5-10 metres. Location fixes were then obtained using a global positioning system unit (Garmin GPS 76, Garmin, Kansas). At least two squat sites were found per animal during the day, whereas other location fixes were obtained across all hours of darkness. A small torch was used to permit quiet and safe movement through the enclosure during times of low moonlight. Methods employed to calculate population densities are described in Chapter 4. The density estimate for the 2007/8 period was based on trapping data recorded in April 2008. Supplementary food is provided within the Uluru mala enclosure via four feed stations (galvanised steel feeders, Magnus Australia) that are continuously stocked with commercially produced macropod pellets ('roo food', Laucke Mills, Australia).

Comments on field methodology

The field techniques I used were successful in radio tracking mala to an acceptable level of accuracy. Study animals were sighted periodically during tracking, which confirmed that the audible tone emitted by the transmitter collars was being accurately interpreted. However, the time consuming nature of individually tracking each animal on foot dictated a small sample size. Although accuracy may be compromised by using triangulation towers, this method permits the rapid collection of location fixes for many animals. However, as stated above, resource limitations did not permit the construction and use of triangulation towers. Further, financial constraints limited the number of radio collars used in the study.

When radio tracked animals were re-trapped and transmitters removed at the conclusion of the study, no evidence of visible injury resulting from the collars was found. Further, animals were seen moving and browsing freely whilst wearing the collars. I also observed a female that had successfully raised a pouch young to near independence whilst collared. However, two misfortunate incidents occurred during the tracking study.

First, one of the collars fell off a study animal after the locking nut came loose. The locking nuts fitted to all subsequent animals were secured with general purpose adhesive. Second, one of the 2010 study animals died when the radio transmitter hanging beneath the animal's neck became caught in the tray of one of the supplementary feed stations. The animal was located shortly after death, and although no intensive post-mortem was undertaken, the lack of apparent injury to the animal suggests it succumbed to capture myopathy. Considering that 10 other animals had worn the collars for many months without misfortune, and that five of these mala had visited the feed stations innumerable times, it was concluded that the death of this study animal was a rare occurrence and that it was appropriate to continue the study.

5.2.3 Home Range Analyses

IAA encompassing 100% of fixes was performed using the computer program Ranges6 v1.215 (Kenward et al 2003) to ascertain the minimum number of fixes required to reasonably estimate home range size. Plots of home range size and number of fixes were deemed to have reached asymptote when further fixes contributed 5% or less to the total home range size. For the majority of animals, sufficient fixes were obtained that additional fixes did not contribute any increase to home range size. Unlike Lundie-Jenkins' (1998) analysis of the data collected during the release of mala to Sangsters Bore, individual incremental area plots were used, rather than combining data for all collared animals. Further, fixes were added chronologically to the IAA. Logistical factors dictated this methodology, as it took around 45 minutes of tracking to obtain each fix.

Two methods were used to estimate mala home range size in this study. First, estimates using the Minimum Convex Polygon (MCP) approach were calculated using Ranges6 v1.215 to permit comparison between the 10 Uluru study animals and with the animals released on Peron Peninsula (Hardman 2006). The use of this method also provided opportunities for future comparisons with studies of other taxa (Fisher and Owens 2000, Kie et al 1996). Second, to enable comparisons with Lundie-Jenkins' (1998) home range estimates, Ranges6 v1.215 was also used to calculate estimates using the Harmonic Mean

Contours (HMC) approach based upon 95% of fixes. For HMC calculations, Tracking Resolution within Ranges6 v1.215 was set at 19 metres, as this was the minimum resolution at which 95% of fixes were incorporated within a single, continuous contour for the majority of study animals. Due to small sample sizes and skewed data, I used the Wilcoxon Rank Sum test (also known as the Mann-Whitney *U* test) to assess variation in mala home range sizes. The tests were run in PHStat2 (Levine et al 2008). ArcGIS version 9.2 was used to plot all mala location fixes on a satellite image of the enclosure, thereby permitting visual analysis of home range overlap between individuals, and the location of squat sites within the home range. The resulting maps also provided evidence of the importance of supplementary food and water points to mala home range, which was further examined by determining the Harmonic Mean Centres for each study animal using Ranges6 v1.215.

When comparing Uluru mala home range estimates with Lundie-Jenkins' (1998) calculations, only data collected 6-12 months and 12-18 after release to Sangsters Bore was used. Although data were also collected by Lundie-Jenkins (1998) within the first six months of release, this was not considered so as to exclude the effect of initial dispersal from the release point on home range size. When comparing Uluru mala home range estimates with Hardman's (2006) calculations, only two tracking sessions conducted at Peron Peninsula were considered as insufficient data was provided to permit comparisons with the third session.

5.2.4 Spatial Autocorrelation

Swihart and Slade (1985) warned that using autocorrelated data (where the location of a study animal is influenced by its previously identified position) can compromise the accuracy of home range estimates. Although autocorrelated data exert a greater influence on home range estimates based on statistical probability, inaccuracies such as underestimation of home range size can result from using such data with non-statistical models (Swihart and Slade 1985). In order to minimise the risk of autocorrelation within time/resource constraints, consecutive fixes for a particular individual were collected at least one hour apart. As 3-6 study animals were tracked during any night, and no more

than 3 fixes for any animal were collected during a single night, most fixes for an individual were recorded at a far greater time interval. Although arbitrary, the one hour minimum time between fixes has been used and accepted in previous radio tracking studies of marsupials (Masters 2003, Moseby and O'Donnell 2003).

5.3 Results

I was able to obtain sufficient data to permit home range analysis for five of the six collared animals tracked during each of the two study periods (Table 5.1). Home range estimates for individuals were calculated for mala tracked in 2007/8 and 2010 using both MCP and HMC (Table 5.1).

Table 5.1 IAA results and home ranges estimates for radio collared mala at Uluru

Study animal	**Tracking period**	**Total no. fixes obtained**	**No. of fixes req. to obtain plot asymptote**	**MCP home range est. 90% fixes (ha)**	**MCP home range est. 95% fixes (ha)**	**HMC home range est. 95% fixes (ha)**
M531	2007/08	23	16	15.77	17.80	10.14
M568	2007/08	28	15	15.23	15.84	12.27
M503	2007/08	23	21	10.48	11.73	11.74
F553	2007/08	27	11	19.34	20.96	21.09
F0601	2007/08	26	21	13.8	16.78	16.45
M575	2010	21	16	26.41	28.65	8.56
M190	2010	18	14	10.40	10.40	5.09
M114	2010	23	20	8.59	14.88	13.89
F977	2010	21	19	3.20	8.82	6.96
F500	2010	23	19	5.91	6.14	7.77

Gender home range size averages were calculated for mala tracked in 2007/8 (Figure 5.2) and 2010 (Figure 5.3) using both MCP and HMC.

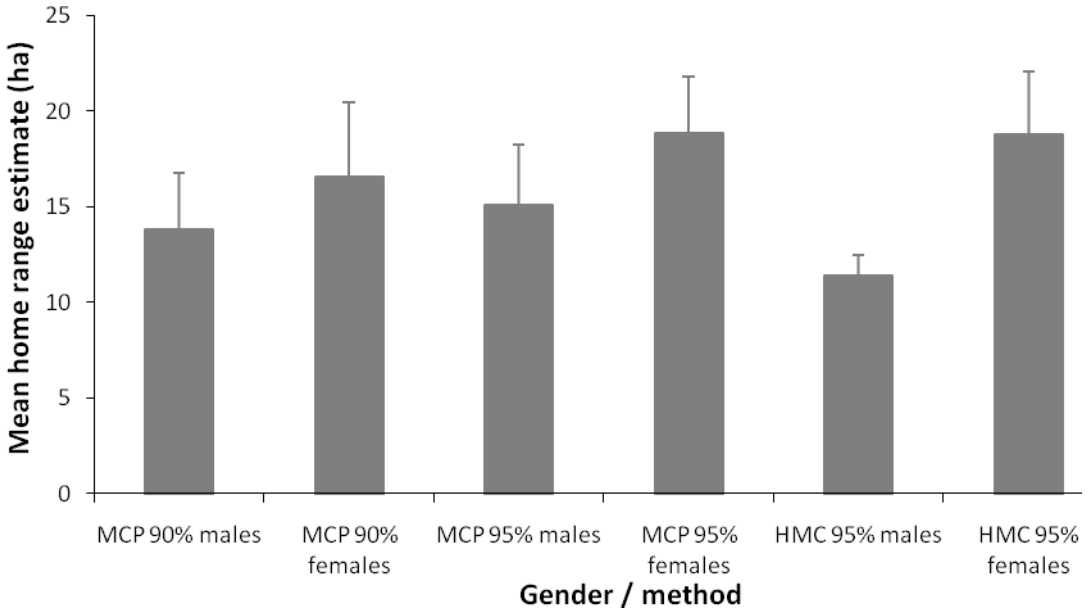

Figure 5.1 Average home range sizes for male and female mala tracked in 2007/8. Standard deviations are shown above home range estimate bars

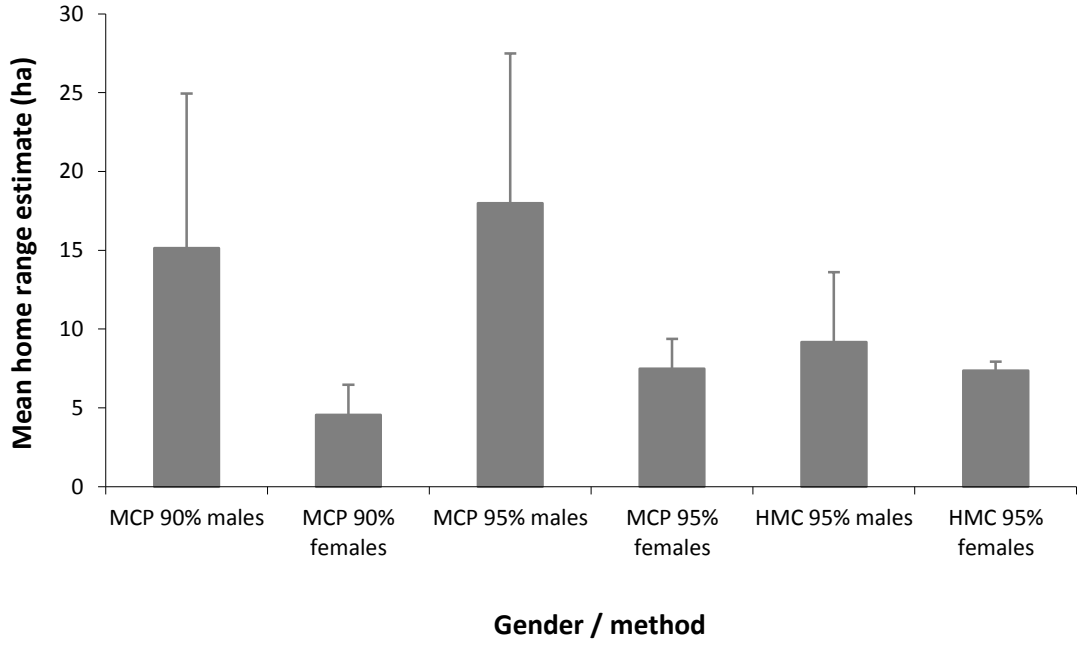

Figure 5.2 Average home range sizes for male and female mala tracked in 2010. Standard deviations are shown above home range estimate bars

The diurnal and nocturnal location fixes for all study animals are plotted over satellite images of the Uluru mala enclosure in Figures 5.3 (July-August 2007), 5.4 (June-July 2008) and 5.5 (June 2010). The MCP for each of these animals is also shown.
There was no significant difference in home range estimates between animals tracked in 2007/8 and 2010 for each of the three methods used (Two-tailed Wilcoxon Rank Sum test, MCP 90%: $Z = 1.567$, $p = 0.117$; MCP 95%: $Z = 1.358$, $p = 0.175$; HMC: $Z = 1.732$, $p = 0.083$). Similarly, when the home range estimate for males (combined across both tracking periods) was compared to that of females there was no significant difference (MCP 90%: $Z = -0.853$, $p = 0.394$; MCP 95%: $Z = -0.610$, $p = 0.522$; HMC: $Z = 0.426$, $p = 0.670$).

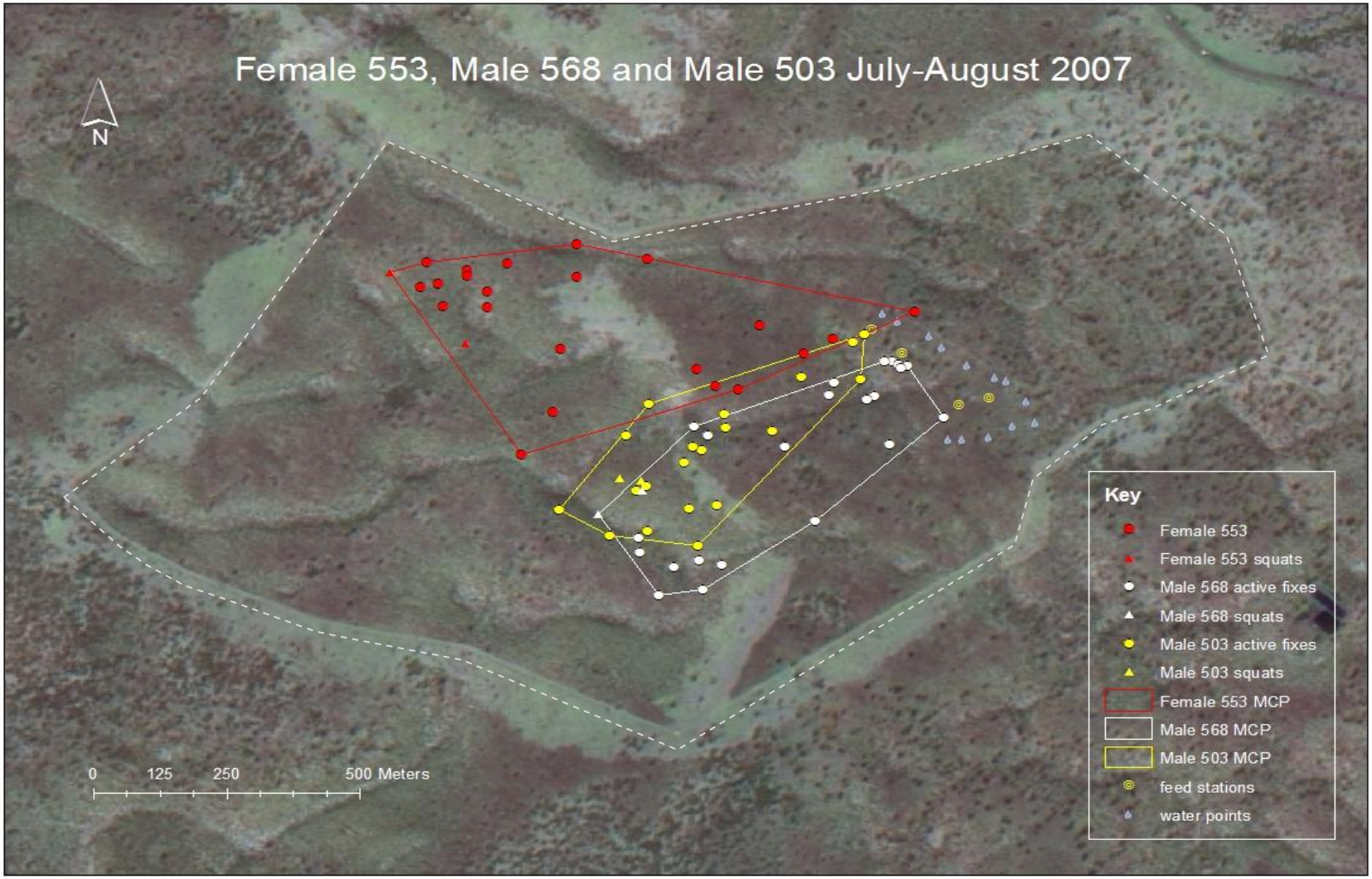

Figure 5.3 Diurnal/nocturnal location fixes and MCP for mala tracked in July-August 2007. Grey dotted line shows enclosure boundary

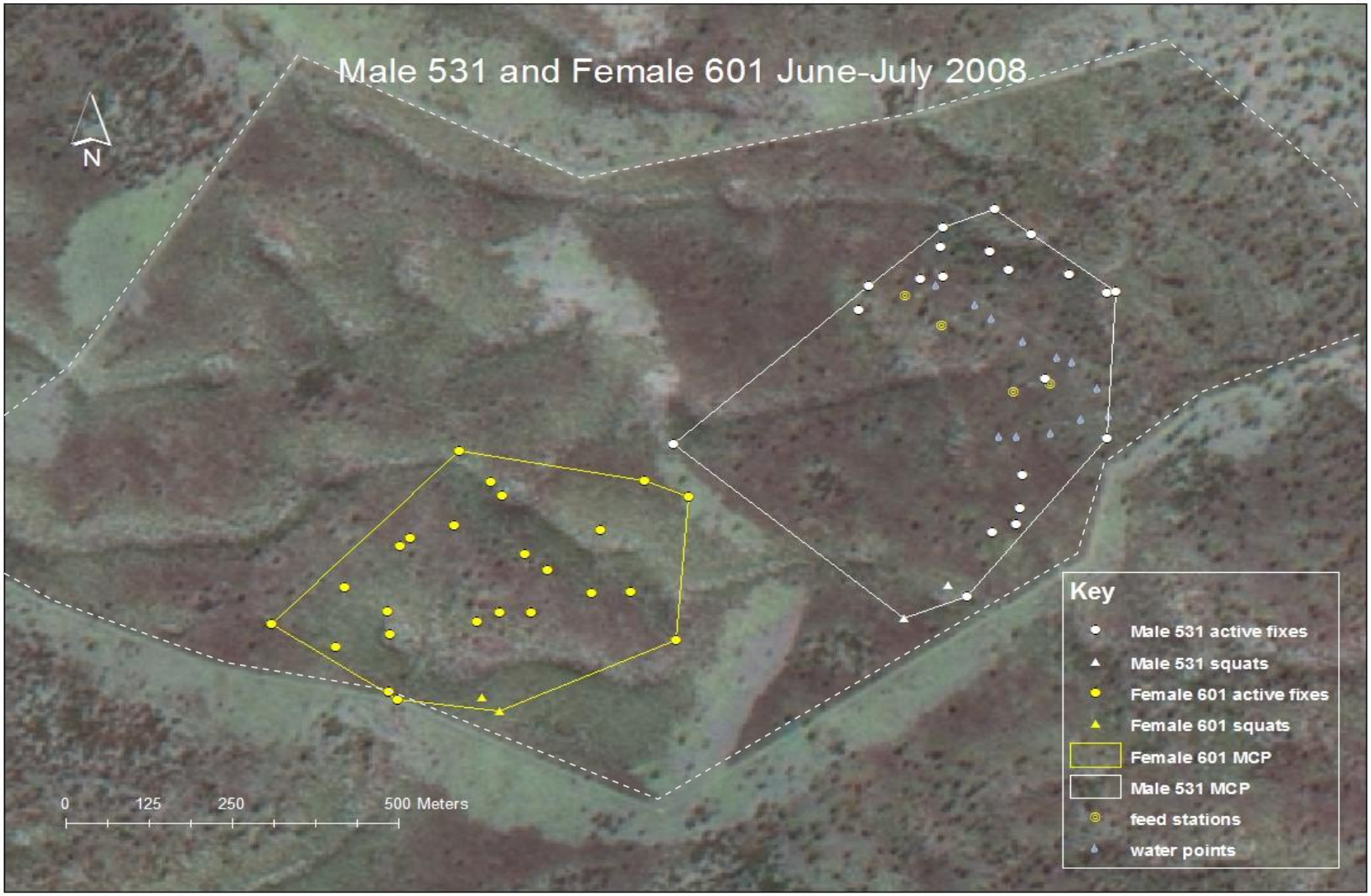

Figure 5.4 Diurnal/nocturnal location fixes and MCP for mala tracked in June-July 2008. Grey dotted line shows enclosure boundary

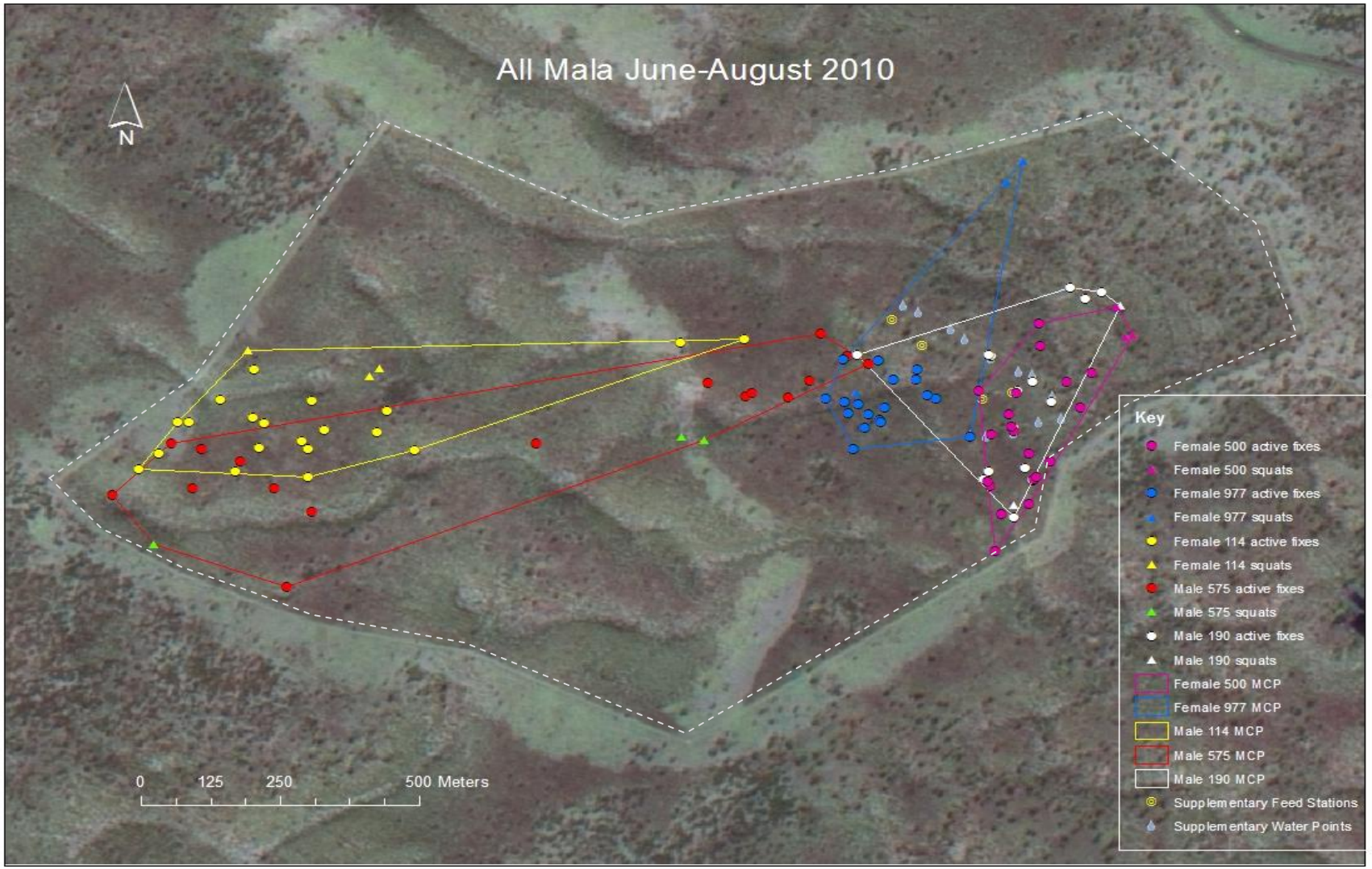

Figure 5.5 Diurnal/nocturnal location fixes and MCP for mala tracked in June-August 2010. Grey dotted line shows enclosure boundary

5.3.1 Home range and supplementary food/water points

The spatial ecology of mala tracked in 2007/8 showed varying relationships with supplementary feed stations and water points. One mala was not found near (that is within approximately 50 metres of) the supplements during the study (Female 601, Figure 5.4), and two other animals were only found in the vicinity on two or less occasions (Female 553, Male 503, Figure 5.3). Conversely, Male 568 (Figure 5.3) and Male 531 (Figure 5.4) were found near the supplementary food and water six and seven times respectively. The mala tracked in 2010 showed similar variability in the proximity of location fixes to the artificial food and water. Two mala, Males 575 and 114, were not found near the supplementary food and water points (Figure 5.5), and Female 997 was located near the supplements only three times (Figure 5.5). However, 50% of the fixes obtained for Female 500 and 67% of those of Male 190 were found near the feed stations and water points (Figure 5.5). To further investigate the importance of the supplements in determining home range use, the Harmonic Mean Centre of each animal's range was calculated using Ranges6 v1.215 (Table 5.2). This permitted the distance to be measured from the activity centre to the nearest supplement. Only the Harmonic Mean Centre of Male 568 was in the immediate vicinity of the supplementary food and water points, whereas for all other animals the Harmonic Mean Centre was at least 100m, and up to 1204 m, from these resources (average 490m, s.d. 398m, Table 5.2).

Table 5.2 Distance between the Harmonic Mean Centre of each study animal's home range and supplementary food and water points

Study animal	Distance from Harmonic Mean Centre to nearest supplementary food/water point (metres)	Supplementary food and water points within home range
F601	919	no
M531	137	yes
M568	0	yes
F553	776	yes
M503	574	yes
M190	205	yes
F500	100	yes
M114	1204	no
F997	170	yes
M575	328	no

5.3.2 Home range overlap

As two temporally distinct tracking periods were undertaken at low population density (July-August 2007, June-July 2008), mala home range overlap is only considered for animals tracked contemporaneously. The home ranges of Female F553 and Male 503 during the first tracking period showed little overlap, and only at the extreme edge of each animal's range (Figure 5.3). Female 553 and Male 568 did not share any home range area (Figure 5.3). However, considerable home range overlap occurred between Male 503 and male 568, including the squat sites for both individuals (Figure 5.3). Only two animals were tracked during the second study period, and the home ranges of these two animals, Male 531 and Female 601, did not overlap (Figure 5.4).

Of the animals tracked in 2010, Male 114 and 575 shared a proportion of their home ranges, particularly in the western part (Figure 5.5). Male 114 did not overlap home range with any other study animal, and Male 575 shared only a very small part of its home range with Female 977 (Figure 5.5). Female 977 and Male 190 showed overlap when 100% of fixes were used to draw a MCP, however if the outlier in the west of Male 190's home range is not considered, the overlap is removed (Figure 5.5). However, significant home range overlap is apparent for Male 190 and Female 500 (Figure 5.5).

5.3.3 Location and use of squats

Diurnal squat sites were located on two occasions for five of the study animals, and three times for the remaining mala. A standard time interval between the identification of nest sites could not be achieved due to logistical constraints. The squat sites used by mala during the 2007/8 tracking period were generally all located on the outer edge of the home range (Figures 5.3 and 5.4). Male 503 was the only animal which nested more centrally within its range (Figure 5.3). In 2010 (Figure 5.5), a similar pattern was observed for four of the study animals (F500, M114, M190, M575). The remaining animal, Female 977, used a squat site within a cluster of active fixes, however it also used two other sites nearly 400 metres remote from all other fixes (Figure 5.5). The distance between squats for individual mala during both tracking periods ranged from 20m to

1000m (average 190m, s.d. 279m, Figure 5.6). Only one individual was found occupying the same squat site on multiple occasions (M503, Figure 5.6).

Considering the three animals tracked contemporaneously over July-August 2007, Female 553 selected nesting sites remote from those of Males 568 and 503 (Figure 5.3). Conversely, the squat sites of the two males were located relatively close together, with the closest being only around 14 metres apart. However, the two animals were not using these closest squats on the same day, but rather were 52 metres and 111 metres apart on the two occasions that contemporaneous fixes were obtained (Figure 5.3). As stated above, the home ranges of the two animals tracked in in 2008 did not overlap, and consequently the squat sites of these individuals were remote (Figure 5.4).

Of the mala tracked in 2010, only Male 190 and Female 500 nested within each other's home range (Figure 5.5). All other squats were located solely within the home range of the individual mala. Further, Male 190 and Female 500 appear to have occupied the same squat site 11 days apart (Figure 5.6). Inaccuracy with the radio tracking technique employed in this study (including inherent GPS inaccuracy and a desire not to approach a nesting animal so closely that it flushed) makes it impossible to state categorically that the two mala nested in exactly the same site. However, from physically tracking to the location of the fixes I am confident that this was the case.

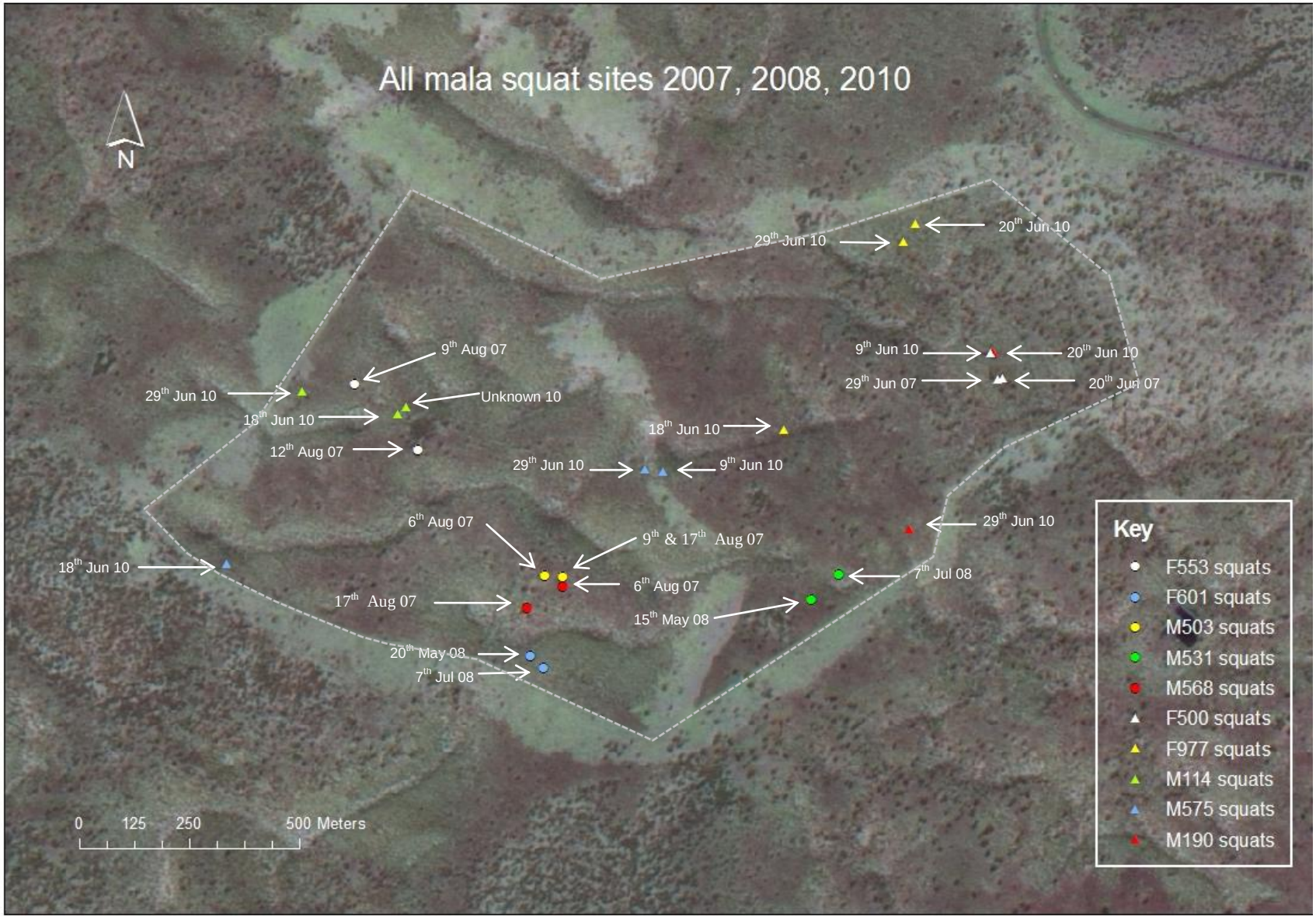

Figure 5.6 Location and date of occupancy of all mala squat sites during the study period. Dotted line shows enclosure boundary

5.3.4 Comparison of mala home range size between Uluru and Sangsters Bore

When Uluru mala home range estimates (both study periods, HMC 95% of fixes; average 11.40 ha, s.d. 4.82) were compared to those of Sangster Bore (both study periods; average 56.76 ha, s.d. 56.33), results showed that Sangsters Bore animals had significantly larger home ranges (Wilcoxon Rank Sum test, Z = -2.646, p = 0.004). Further, unlike Uluru mala, male home ranges estimates (both study periods combined; average 104.94 ha, s.d. 61.95) at Sangsters Bore were significantly larger than those of females (average 24.63 ha, s.d. 15.85; Wilcoxon Rank Sum test, Z = 2.635, p = 0.004).

5.3.5 Comparison of mala home range size between Uluru and Peron Peninsula

When Uluru mala home range estimates (both study periods, MCP 95% of fixes; average 15.2, s.d. 6.61) were compared to those of the first study period conducted at Peron Peninsula (n = 8, average 10.78, s.d. 3.08), there was no significant difference in size (Wilcoxon Rank Sum test, Z = -1.51, p = 0.131). However, home range estimates calculated for mala in the second Peron Peninsula study period (n = 3, average 4.57 ha, s.d. 0.20) were significantly different to those of Uluru mala (Wilcoxon Rank Sum test, Z = -2.54, p = 0.011).

5.4 Discussion

5.4.1 Home range size and population density

Changes in mala home range size may occur as the population increases within the enclosure. Intraspecific aggression may restrict home range size, or alternatively mala may have to increase their range in order to access sufficient resources. Such a change may provide an indication of the proximity of the population to the carrying capacity of the enclosure. No significant difference in home range size was found between mala tracked in 2007/8 and 2010, where population estimates were 67 and 116 respectively (Chapter 4). Several possible explanations may account for this pattern; the two population sizes, and consequent densities, were not sufficiently dissimilar to produce a significant difference in home range size; mala home range size remains stable (although quite variable between individuals) regardless of population density; mala density within the enclosure has yet to reach the point where home range is affected; the amount of

supplementary feed consumed by mala increased, thereby permitting a higher density of animals to be sustained in the same area; that my study's small sample size was insufficient to permit robust statistical analysis of the data and therefore is not detecting actual differences. In order to test the above hypotheses, additional tracking studies would be required. However, trapping results suggest that mala density at Uluru has yet to reach a point where animal wellbeing is compromised. Health checks of animals are carried out by a veterinarian during regular trapping of the population. An increase in fighting injuries and/or individual parasite load, and a general decrease in fitness and fecundity, may suggest that carrying capacity has been reached or overshot. Monitoring of fecundity rates and individual animal health at Uluru in June 2010 did not show that mala were stressed, and therefore the fact that home range size has not changed over time is likely to suggest the current population is below the carrying capacity of the enclosure. Completing radio tracking studies is relatively expensive and very labour-intensive. Consequently, as trapping results and health checks are less resource-intense and provide a ready source of information, I suggest that this work, in conjunction with the monitoring of preferred mala food plants identified in Chapter 6, provide the basis for future assessments of the carrying capacity of the Uluru enclosure.

5.4.2 Effect of supplementary food and water on the home ranges of mala at Uluru

The data showed that half of the radio tracked individuals may have obtained a potentially considerable proportion of their nutritional requirements from the supplementary feed stations (see Chapter 6 for corroboration of this point). However, Harmonic Mean Centres calculated for all study animals showed only a single individual focused its activity amongst the supplementary food and water points. Further, the data showed the remaining animals did not use the supplements at all, or were rarely found within the vicinity. As the majority of study animals were collared after being trapped near the food/water points, it could be assumed that the use of artificial food/water by the study animals is not representative of animals living remote from the supplements. The data therefore suggests that a significant percentage of mala living within the Uluru enclosure are able to find adequate nutrition from natural sources. The range of food plants browsed by mala, as listed in Chapter 6, further reinforces this conclusion. This is

a significant point when assessing the costs and benefits of managing predator-proof enclosures, and suggests that with respect to nutrition, Uluru mala appear to be satisfactory candidates for future wild release.

5.4.3 Comparison of home range size between mala at Uluru and Sangsters Bore

As discussed above, differences between home range size estimates of mala at Uluru and Sangsters Bore may challenge assumptions regarding the benefits of large, predator-proof enclosures. It must be considered that the mala location data collection techniques used by Lundie-Jenkins (triangulation towers) and my study (tracking on foot) were different. Further, the tracking resolution used by Lundie-Jenkins within the Ranges software is unknown, although it was presumably suitable for the tracking method employed. In addition, if population density does indeed affect mala home range size, then the difference in density between Uluru and Sangster's Bore must also be kept in mind. Assuming that such differences in data collection have not contributed significantly to the observed variance in mala home range estimates at the two study areas, this result could suggest several scenarios. First, the spatially limited environment of the Uluru enclosure (and of both the Watarrka enclosure from which the mala were translocated and the intensive captive breeding environment of their forebears), has resulted in a permanent change in ranging behaviour. Second, mala have not suffered a behavioural change, but rather are finding sufficient food, nesting sites and breeding partners within the home ranges they are establishing such that it is unnecessary to form larger ranges. Although comparisons showed Sangsters Bore mala established significantly larger home ranges, some range estimates at Uluru were comparable, or larger, than the estimates for some individuals at Sangster Bore. All home range estimates calculated at Uluru (n = 10, HMC, 95% of fixes) were under 30 hectares, and five of the Sangsters Bore animals (n = 15) also established ranges up to this size (Figure 5.7).

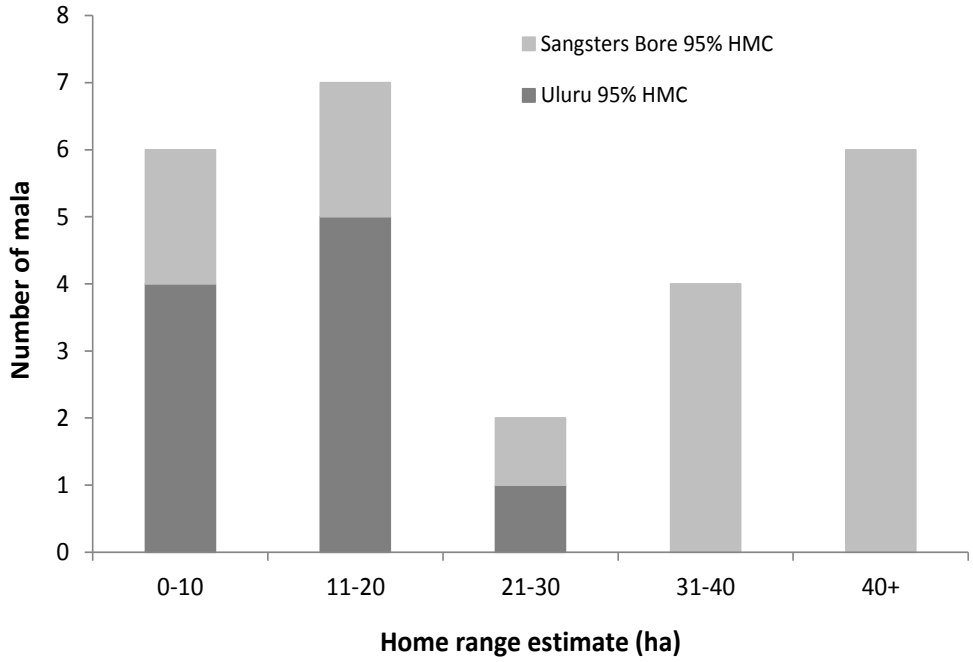

Figure 5.7 95% HMC home range estimates for mala at Uluru and Sangsters Bore

Therefore, regardless of whether animals are ranging in the 'wild' or within artificial confines, it appears that when sufficient resources are available, mala will establish relatively small home ranges. Further, the Sangsters Bore home range data are skewed by three very large estimates of ranges exceeding 150 hectares. These animals were males, which may have established such large ranges in order to locate females for breeding (Lundie-Jenkins 1998). Lundie-Jenkins (1998) calculated mala density at 6.65 per km^{-2} from data collected over two years of intensive tracking work. However, estimates of density during the two tracking periods at Uluru were 39.4 per km^2 in 2007/8 and 68.2 per km^2 in 2010. Thus, it could be argued that male mala at Sangsters Bore may have required larger home ranges in order to find potential breeding partners. The relatively high mala densities, and therefore breeding opportunities, at Uluru may also explain the lack of sex-based differences in home range size when compared to Sangsters Bore. As animals released at Sangsters Bore were sourced from the mala intensive breeding facility in Alice Springs, it would be logical to conclude that any behavioural changes resulting from spatial confinement would be more pronounced in these animals

than those reintroduced to Uluru from other large, predator-proof enclosures. Consequently, it is unlikely that the difference identified between Sangsters Bore and Uluru mala home range size is a consequence of behavioural change. Therefore, I suggest that although home range estimates were significantly different between the two study sites, this is likely due to a requirement for Sangsters Bore mala to range more widely to find adequate food and mating opportunities, rather than behavioural change in Uluru mala or shortcomings in the enclosure's ability to adequately provide for mala ranging needs. This conclusion is supported by the results of the comparison between Uluru mala home ranges and those estimated for animals released at Peron Peninsula (section 5.4.4 below).

5.4.4 Comparison of home range size between mala at Uluru and Peron Peninsula

The data collection method used to obtain active fixes at Peron Peninsula was different to that adopted for the Uluru study. However, as contemporaneous fixes were taken from mobile receivers (that is by researchers on foot) rather than fixed towers, the impact of methodological differences on results is likely to be less than those resulting from the Tanami Desert study. Hardman (2006) believed that home range estimates calculated for the first tracking period were affected by the provision of supplementary food. He found that home ranges were smaller during the second phase of tracking when mala were not travelling between nesting sites and food supplements (Hardman 2006). Considering that supplementary food and water are provided within the Uluru enclosure, it could be argued that the Uluru study is more directly comparable to the first Peron Peninsula tracking period. As home range estimates for Uluru mala and those tracked at Peron Peninsula during the first study period were not significantly different, it appears that the predator-proof enclosure at Uluru is providing adequately for the ranging behaviour of mala. The fact that Peron Peninsula animals occupied significantly smaller home ranges than Uluru mala in the second tracking period further suggests that mala are not negatively affected by the spatial restriction of the predator-proof enclosure.

5.4.5 Home Range Overlap

A thorough study of mala home range overlap would necessitate the fitting of radio collars to all animals within a designated area. Consequently, this study provides only an indication of overlap between a small number of individuals, which are likely to reside in the same general area as a relatively large number of other mala which were not collared and tracked. It may be argued, therefore, that my results will underestimate the true extent of range overlap. However, some generalisations can be made. The data show that male mala do not defend exclusive territories, but rather permit other individuals to be active, and nest, within their home ranges. Hardman (2006) concluded similarly after radio tracking mala released at Peron Peninsula. Although no home range overlap between females was observed, this is likely due to the difficulty in trapping females appropriate for collaring (adult animals without large pouch young) which resulted in a female sample size of four. Study of female range overlap is further restricted by the fact that contemporaneous data were only collected for two of these animals. Considering the density of mala observed within the vicinity of the supplementary food and water points, I expect that a comprehensive study of all females in this area would show considerable overlap between individuals. This conclusion is supported by Hardman (2006) who recorded substantial female home range overlap at Peron Peninsula. Range overlap between males and females at Uluru observed in both tracking periods, and particularly in 2010, would likely be extended if the sample size was increased. The propensity of Uluru mala to share resources within their home range is consistent with Lundie-Jenkins (1998) study of free-ranging reintroduced mala. In addition, Hardman (2006) found that range overlap of mala released to Peron Peninsula was greater when supplementary food was provided, further supporting the conclusion that rich food sources will be shared by a number of individuals.

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Chapter 6. A study of mala diet at Uluṟu - Kata Tjuṯa National Park and comparison with diet of free-ranging mala in the Tanami Desert

6.1 Introduction

After previously ranging over a vast area of arid and semi-arid Australia, mala became extinct on the mainland in 1991 (Langford 1999). Consequently, the foci of subsequent recovery efforts have been captive breeding and translocation (Chapter 1). An understanding of species' ecology is critical to maximising the chance of translocation success, however a paucity of such knowledge often exists for rare and threatened taxa (Booth 1988). This lack of knowledge extends to mala diet, affecting the ability of the translocation proponents to select appropriate release sites.

Three studies of mala diet were undertaken before the species became extinct on mainland Australia. The first was conducted in the Tanami Desert 450 km north-west of Alice Springs by Pearson (1983, 1989) after a period of above average rainfall. Two methods were used to identify dietary components; signs of grazing and microscopic faecal pellet analysis (Pearson 1983, 1989). Further, the discovery of a dead animal provided the opportunity to compare the stomach and rectal contents of an individual, which Pearson (1983) concluded were not significantly different. He thus determined that microscopic analysis of faecal pellets was an acceptable method of mala diet analysis. Pearson (1983) also conducted vegetation surveys at the study site in order to assess proportional availability of plant species to mala. Faecal pellet samples showed mala browsed a broad range of plants, and preferred grass and sedge seed, soft green grass shoots, sedges and some dicotyledonous material (Pearson 1983:vii, 1989). A strong preference for grass leaf over stem was also detected.

Lundie-Jenkins et al. (1993) also studied the diet of the last wild mala population in the Tanami Desert. Faecal pellets were collected between July 1986 and December 1987 to permit microscopic analysis, and direct observation of animals also provided data for the study. Vegetation surveys were undertaken during pellet collection to assess the availability of different plant species to mala. Lundie-Jenkins et al. (1993) found that mala diet was highly variable, and comprised a wide variety of plant species. The authors concluded that although the leaves and stems of grasses and sedges were dietary staples, seeds and fruits were preferred when available (Lundie-Jenkins et al. 1993:492). Lundie-Jenkins et al. (1993) also stated that insects were intentionally consumed by mala during times of nutritional stress.

A reintroduction of mala to the site of the last wild populations in the Tanami Desert began in 1989 (Lundie-Jenkins 1998). As part of post-release monitoring, Lundie-Jenkins (1998) conducted a further study of mala diet. Faecal pellets were collected between March 1991 and July 1993 to permit microscopic analysis (Lundie-Jenkins 1998). Vegetation transects were surveyed to ascertain plant diversity and abundance. Lundie-Jenkins (1998) found mala browsed a wide range of plants, and made use of food supplements provided as part of post-release assistance. Grasses of the genus *Eragrostis* were the single most preferred dietary item, although monocot and dicot species were consumed at similar levels overall (Lundie-Jenkins 1998). Supplementary feed made up approximately a third of total diet.

Mala became regionally extinct in the southern part of Central Australia during the late 1950s (Langford 2000). Although Indigenous people held knowledge of mala diet, no systematic, scientific analysis of the diet of the species was completed prior to its disappearance from this part of its range. The reintroduction of mala to Uluru provided an ideal opportunity to conduct such a study. Dietary analysis of the Uluru mala population was undertaken with three primary aims. First, to identify the main food plants consumed by mala. Monitoring protocols for a number of these species could then be established to identify browsing impact at increasing population size, thereby assisting in the calculation of the Uluru predator-proof enclosure's carrying capacity. Second, to

establish the importance of supplementary feed to mala diet. This would provide important data regarding the level of 'natural' foraging behaviour displayed by mala at Uluru. Third, to contribute to the existing knowledge of mala diet, including a potential increase in known food species. A better understanding of mala dietary requirements would permit the more accurate identification of suitable future reintroduction sites for the species. To achieve this aim, the results of the Uluru dietary analysis were compared to those studies previously undertaken in the Tanami Desert.

6.2 Methods

6.2.1 Rationale for using faecal pellet analysis

I analysed faecal pellets to study mala diet, and this method was chosen for a number of reasons. First, the process is non-invasive and therefore acceptable for use with threatened species such as mala. Second, a previous study found no significant difference between the proportions of plant materials found in the stomach and faeces of mala (Pearson 1983), and thus the technique may be considered accurate for the taxon. Third, faecal pellet analysis is the most widely used and accepted method of macropod diet analysis. Last and perhaps most importantly, using this technique provided the opportunity to compare results with previous mala dietary studies, all of which were based on faecal pellet analysis.

6.2.2 Sampling protocol

Initially, five equally-spaced pellet collection sites were selected around the internal fenceline of the Uluru mala enclosure. A collection site at one of the supplementary feed stations was also chosen. Fresh pellets, identified by their black, shiny appearance, were collected from these sites in January, March and June of 2006. As mala became active across the enclosure, and the number of animals began to increase, an additional five pellet collection sites were randomly chosen. Samples were gathered from this increased number of sites in July 2006, October 2006, and April 2009. Pellets were sent to Elizabeth Jefferys, Principal of EAJ Consultants (Pagewood, New South Wales) for preparation and analysis. Of the total pellets collected at Uluru, 20 samples of five scats were selected. Each sample contained material gathered from specific sites at a particular

time. These 20 samples were prepared by crushing and mixing the five pellets before extracting a single 2 x 2 x 1-cm sample. This sample was then placed through four sieves of 0.58 mm, 0.41 mm, 0.26 mm and 0.13 mm mesh. Any material passing through the smallest sieve was collected on filter paper. Each sample was thus divided into 5 sub-samples of differing particle size. Each sub-sample was placed onto labeled 25 x 76.2-mm microscope slides and placed on a hotplate at 30 degrees Celsius to dry. When dry, the slides were covered with a mounting medium (Euparal) and a 22 x 50-mm slip.

6.2.3 Plant reference collection

I collected samples of 74 plant species found within the Uluru mala enclosure to create a reference collection to aid faecal fragment identification (Table 6.1). Where possible, leaf, stem and seed samples were obtained, and supplementary feed and soil samples were also collected. Reference slides were created by grinding these materials with a mortar and pestle to imitate mastication by mala. Ninety-five percent ethanol was added, the sample ground for a second time, and then material removed with a pipette and jewelers forceps. After evenly spreading the material onto microscope slides, a mounting material (Euporal) and cover slip were added. Samples were examined under a binocular microscope at 40x and 100x magnification, and identified to the lowest possible taxonomic level. Plant material was divided into eight categories; leaf, stem, seed, insect, supplementary food leaf/stem, supplementary food seed, soil, and 'unknown'. Quantitative evaluation of each food category was equivalent to the relative occurrence evaluation in Hansson (1970). The area of food materials covered on each of the sample's five sub-sample slides was 22 x 55-mm. A total of 50 fields were scored for each slide, with each food category being recorded as present or absent. The methods of sample and reference slide creation, and analysis of samples, described above have been successfully used in a number of previous dietary studies of Australian herbivorous mammals (Firth et al. 2005, Fox et al. 1994, Luo et al. 1994, Nano et al. 2003).

I examined the relationship between dietary variables and a number of rainfall variables. Specifically, the amount of leaf, stem and seed found within faecal pellets was correlated with rainfall occurring both three and six months prior to the collection of samples

(rainfall data sourced from Commonwealth of Australia 2010). Similarly, the percentage of both *Poaceae* and supplementary food were also tested for correlation with rainfall. Last, correlation analysis was performed between the number of mala present in the enclosure, and the percentage of supplementary food found in faecal pellet samples. Tests were run in PHstat2 (Levine et al. 2008).

Table 6.1 Plant species collected from the Uluru mala enclosure to create a reference collection for the identification of plant fragments in mala faecal pellet samples

CASUARINACEAE	*Allocasuarina decaisneana*
PROTEACEAE	*Grevillea eriostachya*
	Grevillea stenobotrya
	Grevillea juncifolia
	Hakea subarea
	Hakea divericata
SANTALACEAE	*Exocarpus sparteus*
	Santalum lanceolatum
LORANTHACEAE	*Lysiana exocarpi*
GYROSTEMONACEAE	*Gyrostemon ramulosus*
PORTULACACEAE	*Calandrinia sp prob balonensis*
	Calandrinia sp prob reticulata
CAESALPINIACEAE	*Senna artemesioides*
MIMOSACEAE	*Acacia melliodora*
	Acacia ligulata
	Acacia ramulosa
	Acacia aneura
	Acacia maitlandii
	Acacia tetragonophylla
	Acacia minyura
FABACEAE	*Crotalaria cunninghami*
	Daviesia arthropoda
	Leptosema chambersii
	Swainsona microphylla
EUPHORB IACEAE	*Euphorbia wheeleri*
	Euphorbia australis
	Euphorbia tannensis
CHENOPODIACEAE	*Rhagodia eremaea*
	Scleroleana johnsoni
	Enchylaena tomentosa
	Salsola kali
AMARANTHACEAE	*Ptilotus obovatus*
	Ptilotus polystachyus
	Ptilotus latifolius
SAPINDACEAE	*Dodonea viscosa*
MALVACEAE	*Sida ammophila*
	Alyogyne pinoniana
	Abutilon otocarpum
GOODENIACEAE	*Leschenaultia divericata*
	Goodenia cyclopter
	Dampiera cinerea
	Scaevola parvifolia
	Scaevola basedowii
BRUNONIACEAE	*Brunonia australis*
ASTERACEAE	*Podolepis canescens*
	Leucochrysum stipitatum
	Calotis erinacea
	Chrysocephalum eremaeum
POACEAE	*Eriachne helmsii*
	Paraneurachne muelleri
	Enneapogon polyphyllus
	Enneapogon avenaceus
	Aristida holathera
	Aristida contorta
	Eragrostis eriopoda
	Cymbopogon obtectus
	Paractaenum refractum
	Paspalidium reflexum
	Triodia schinzii
	Triodia pungens
	Amphipogon caricinus
MYOPERACEAE	*Eremophila longifolia*
	Eremophila latrobei latrobei
	Eremophila gibsonii
	Eremophila willsii
SOLANACEAE	*Solanum coactiliferum*
	Solanum centrale
VERBENACEAE	*Newcastelia spodiotricha*
BORAGINACEAE	*Trichodesma zeylanicum*
CONVULVULACEAE	*Bonamia erecta*
MYRTACEAE	*Aluta maisonneuvei*
STERCULIACEAE	*Rulingia loxophylla*

6.2.4 Methods used in previous mala diet studies

All previous studies of mala diet used microscopic faecal analysis to determine dietary composition. A detailed description of methods employed for each study is provided in Pearson (1983, 1989), Lundie-Jenkins et al. (1993) and Lundie-Jenkins (1998). Briefly, Pearson (1983, 1989) used a point sampling method when conducting microscopic analysis. Instead of considering all plant fragments within a microscope field (the method used for the Uluru analysis), material lying across the intersection of lines on a grid was identified. Both Lundie-Jenkins et al. (1993) and Lundie-Jenkins (1998) based analyses upon random searches of sample slides and the identification of 50 plant fragments. Although these procedures differ to my study, assuming all provide a random sample of material (the objective of each method) they are not likely to provide significantly different results. Further, all are equally affected by the issue of variable digestibility (Barker 1986), and are not biased by fragment size.

General comparisons were made between the mala dietary composition at Uluru and the Tanami Desert. These included using the results of faecal pellet analysis to calculate and compare Levins' standardized measure of niche breadth for each of the four studies of mala diet: $B_A=(1/\sum p_i^2)-1/n-1$, where p_i = the proportion of occurrence of each food plant in the mala diet, and n = the number of plant species in the mala diet (Levins 1968, Pavey et al. 2008, Welden and Welden 2002).

6.3 Results

6.3.1 Major dietary categories

Microscopic analysis of mala faecal pellets revealed evidence of minimal mastication of food by mala. Much of the food material was in relatively large pieces, with little sign of chewing or grinding. The major dietary categories identified within the samples were supplementary food (43%) and grasses (40%; Figure 6.1). When supplementary food and soil are discarded from the analysis, the data reveal that only 6% of faecal material was comprised of non-grass species.

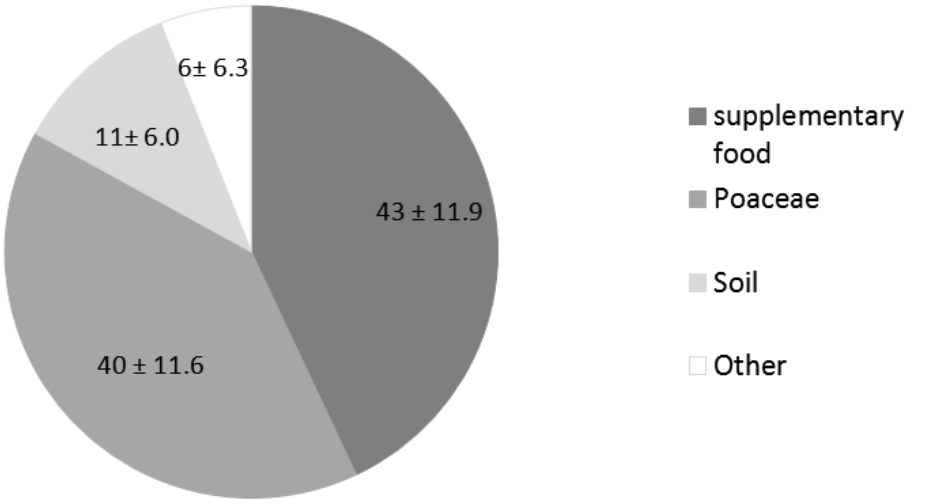

Figure 6.1 Average percentage (± standard deviation) composition of Uluṟu mala faecal pellet samples

6.3.2 Preferred plant parts

Each of the 20 samples contained a higher proportion of leaf material than stem or seed, and in only one sample was more seed than stem identified. When all samples were averaged, leaf material was clearly the most frequently eaten plant part (Figure 6. 2).

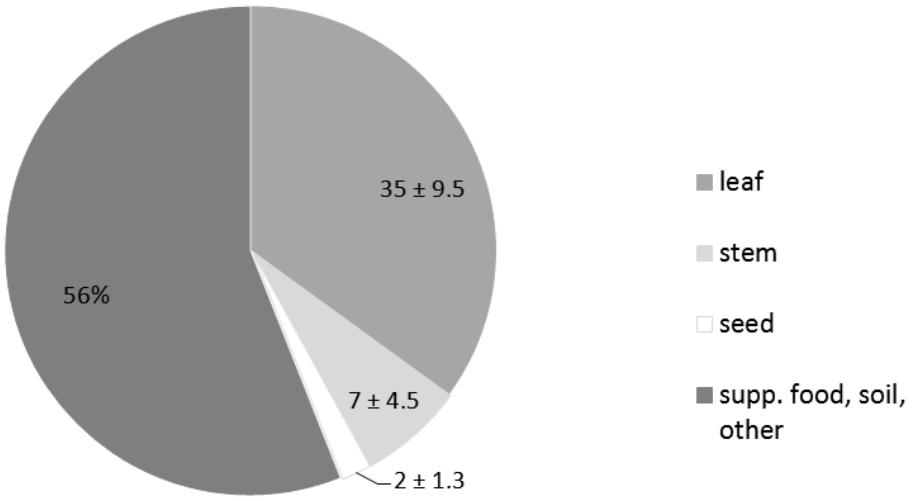

Figure 6.2 Average percentage (± standard deviation) of leaf, stem, seed and supplementary food/soil/other present in all Uluṟu mala faecal pellet samples

6.3.3 Plant and insect species present in faecal samples

A total of 27 native species were consumed by mala at Uluru during the survey period, 20 of which were previously unknown as mala dietary components (Table 6.2). Twenty-four species were found in 10 or less of the 20 pellet samples, with the remaining three taxa found in 15, 19 and 20 of the samples. *Eragrostis eriopoda* and *Aristida contorta* were the most abundant native species, being found in 19 and 20 of the samples respectively. In addition to naturally occurring floral species, supplementary food was present in all samples and averaged 43% (s.d. 12.3) of total sample material. Although invertebrates were found in 60% of faecal pellet samples, they never made up more than 4% of total material. Soil was present in all samples, and ranged from 1% to 21% of total material (average 11.15%, s.d. 5.96). The seeds of 19 of the total 27 species were consumed by mala. No evidence of flowers, pollen, root material, or fungal spores was found.

Although no methodical investigation of mala diet through direct observation was attempted, I witnessed mala browsing a variety of plants during data collection for other chapters of this thesis. All these taxa were also identified in the faecal analysis, with the exception of *Grevillea stenobotrya*, the leaves of which I observed an individual mala consuming on a single occasion.

6.3.4 Fluctuations in sample composition and correlation with rainfall

Fluctuations in both rainfall and the percentage of different plant parts (leaf, stem and seed) found in mala faecal pellet samples were observed during the study period. However, no correlation was found between rainfall in the previous three and six months before sample collection and the average percentage of leaf (three months: $r = -0.58$, $t = -0.61$; six months: $r = -0.44$, $t = -0.90$, $d.f = 4$, $n = 6$, $P > 0.05$), stem (three months: $r = -0.80$, $t = 2.695$; six months: $r = 0.74$, $t = 2.234$, $d.f = 4$, $n = 6$, $P > 0.05$) or seed (three months: $r = -0.148$, $t = -0.299$; six months: $r = -0.141$, $t = 0.142$, $d.f = 4$, $n = 6$, $P > 0.05$) identified (Figure 6.3).

Table 6.2 Dietary items in mala faecal pellets collected from sampling sites inside the Uluru enclosure

Flora					
Family	**Genus, species**	**Dietary item**	**% of samples / number of samples**	**Ave. % all samples, (lowest % single sample – highest % single sample)**	**Previously recorded mala dietary component**
Proteaceae	*Hakea divericata*	leaf	5%, 1	0.25 (0.25 - 0.25)	
		seed	20%, 4	0.69 (0.25 - 2)	
	Hakea lorea (suberea)	seed	5%, 1	0.25 (0.25 - 0.25)	
Santalaceae	*Exocarpus sparteus*	seed	10%, 2	0.38 (0.25 – 0.5)	
Portulacaceae	*Calandrinia sp*	seed	5%, 1	0.5 (0.5 – 0.5)	
Mimosaceae	*Acacia melliodora*	seed	5%, 1	0.5 (0.5 – 0.5)	yes
Fabaceae	*Crotalaria cunninghamii*	seed	5%, 1	0.25 (0.25 - 0.25)	
Chenopodiaceae	*Scleroleana johnsoni*	leaf	10%, 2	4.5 (4 – 5)	
Amaranthaceae	*Ptilotus polystachyus*	seed	10%, 2	0.38 (0.25 – 0.5)	
	Ptilotus latifolius	seed	5%, 1	1 (1 – 1)	
Malvaceae	*Alyogyne pinoniana*	stem,	10%, 2	1.5 (1 – 2)	
	Sida ammophila	leaf,	5%, 1	0.25 (0.25 - 0.25)	
		seed	5%, 1	1 (1 – 1)	
Sterculiaceae	*Rulingia loxophylla*	leaf	30%, 6	4.67 (1 – 15)	
		seed	5%, 1	1 (1 – 1)	
Boraginaceae	*Trichodesma zeylanicum*	leaf	34%, 7	3.14 (1 – 5)	
		seed	5%, 1	1 (1 – 1)	
Verbenaceae	*Newcastelia spodiotricha*	leaf	5%, 1	0.25 (0.25 - 0.25)	yes
Solanaceae	*Solanum centrale*	seed	45%, 9	0.69 (0.25 – 1)	
	Solanum coactiliferum	seed	10%, 2	0.75 (0.25 – 1.25)	
Myoporaceae	*Eremophila latrobei lat*	seed	5%, 1	0.5 (0.5 – 0.5)	
Goodeniaceae	*Goodenia cycloptera*	seed	20%, 4	0.56 (0.25 – 1)	
Unknown	*Unknown dicotyledon A*	leaf	5%, 1	0.25 (0.25 - 0.25)	
		seed	15%, 3	4 (1 – 6)	
	Unknown dicotyledon B	seed	5%, 1	0.5 (0.5 – 0.5)	
Poaceae	*Aristida contorta*	leaf	100%, 20	15.75 (3 – 27)	
		stem	85%, 17	3.38 (0.5 – 11)	
	Eragrostis eriopoda	leaf	85%, 17	14.29 (2 – 24)	yes
		stem	14%, 14	5.3 (1 – 11)	

Table 6.2 *continued*

Flora					
Family	**Genus, species**	**Dietary item**	**Number / % of samples**	**Ave. % all samples, (lowest % single sample – highest % single sample)**	**Previously recorded mala dietary component**
	Triodia pungens	leaf	65%, 13	3.63 (0.25 – 7)	yes
		stem	34%, 7	2.29 (0.5 – 7)	
		seed	15%, 3	0.67 (0.5 – 1)	
	Triodia schinzii	stem	10%, 2	0.75 (0.5 – 1)	yes
	Amphipogon caricinus	seed	5%, 1	0.5 (0.5 - 0.5)	
	Cymbopogon obtectus	leaf	10%, 2	4 (1 – 7)	
		stem	5%, 1	4 (4 – 4)	
	Eriachne helmsii	leaf	40%, 8	2.59 (0.25 – 7)	
		stem	10%, 2	2.50 (1 – 4)	
Supplementary food		leaf and stem	100%, 20	33.05 (18 – 59)	
		seed	95%, 19	11.63 (0.5 – 27.5)	
Fauna					
EUGAMASIDAE	*Mesostigmata sp*		5%, 1	0.5 (0.5 – 0.5)	
FORMICIDAE	*Iridomyrmex sp*		25%, 5	1.1 (0.5 – 2)	
UNKNOWN, Order Coleoptera	unknown		20%, 4	2.25 (1 – 4)	
TERMITIDAE	*Nasutitermes*		10%, 2	1 (1 – 1)	
Other					
Soil			100%, 20	11.15 (1 – 21)	

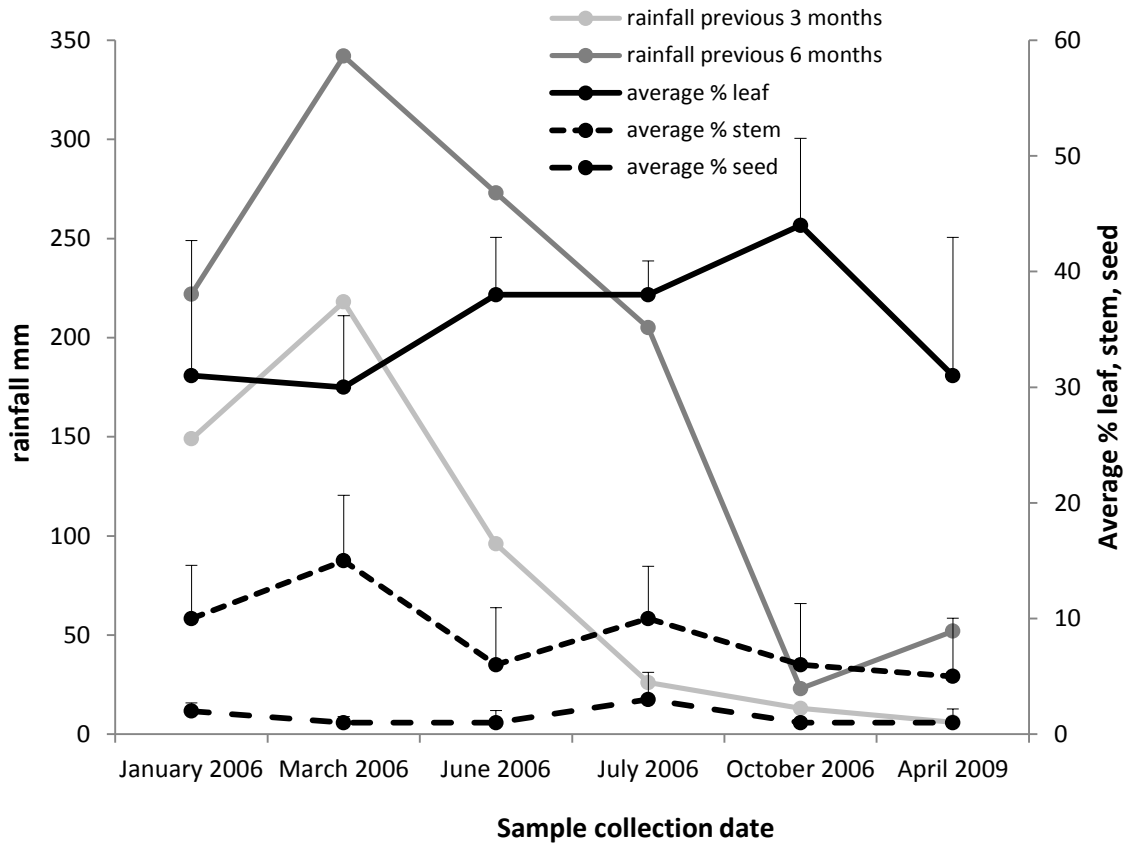

Figure 6.3 Average percentage of leaf, stem and seed found in mala faecal pellet samples collected during the study period (error bars show standard deviation), and rainfall in the previous three and six months prior to sample collection.

The average percentage of Poaceae found in samples collected over the study period showed little variation (Figure 6.4). No correlation was found between rainfall in the previous three and six months prior to the collection of faecal pellet samples and the percentage of Poaceae present (three months: $r = 0.431$, $t = 0.954$; six months: $r = 0.643$, $t = 1.679$, d.f. = 4, $n = 6$, $P > 0.05$).

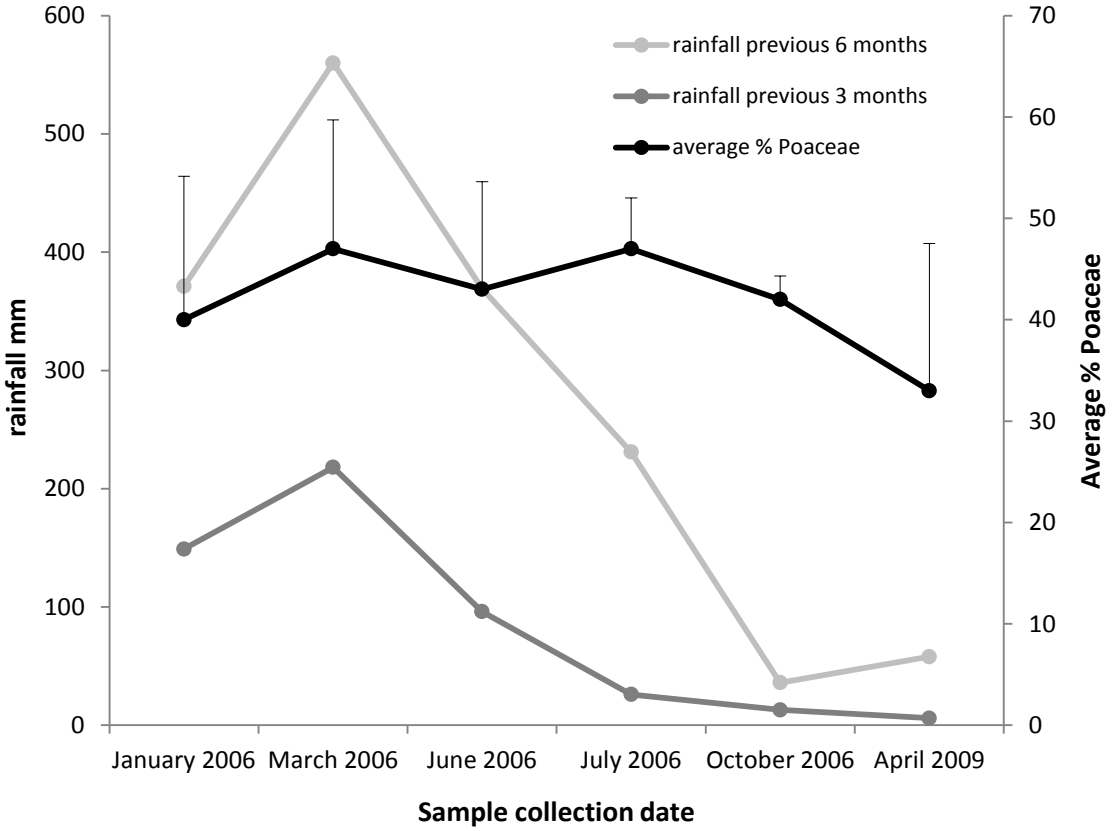

Figure 6.4 Average percentage of Poaceae in mala faecal pellet samples at Uluru (error bars show standard deviation), and rainfall in the previous three and six months prior to the collection of samples.

The average percentage of supplementary food present in mala faecal pellet samples ranged from 33% to 54% (Figure 6.5). No correlation existed between total rainfall either three or six months prior to the collection of samples and presence of supplementary food (three months: $r = -0.395$, $t = -0.861$; six months: $r = -0.395$, $t = -0.861$, $d.f = 4$, $n = 6$, $P > 0.05$).

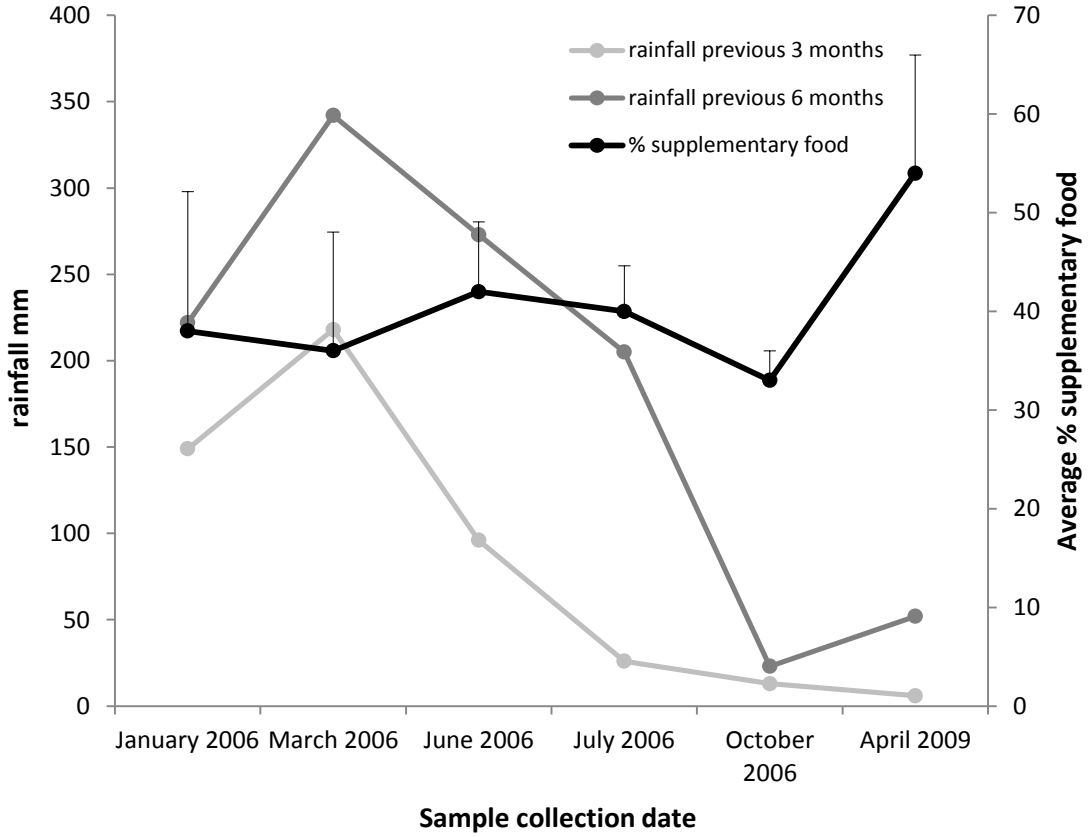

Figure 6.5 Average percentage of supplementary food in Uluru mala faecal pellet samples (error bars show standard deviation), and rainfall in the previous three and six months prior to collection

The average percentage of *Triodia sp*. material found in mala faecal pellet samples at Uluru ranged from 0.34% to 7.5% (Fig 6.6). The average percentage of insects found in the samples was minimal and showed little variation (Fig 6.6). No correlation was found between either of these variables and rainfall in the previous three or six months before samples were collected (*Triodia sp*., three months: r = 0.399, t = 0.871, six months: r = 0.522, t = 1.222; insects, three months: r = 0.015, t = 0.03, six months: r = -0.208, t = -2.00, d.f. = 4, n = 6, P > 0.05)

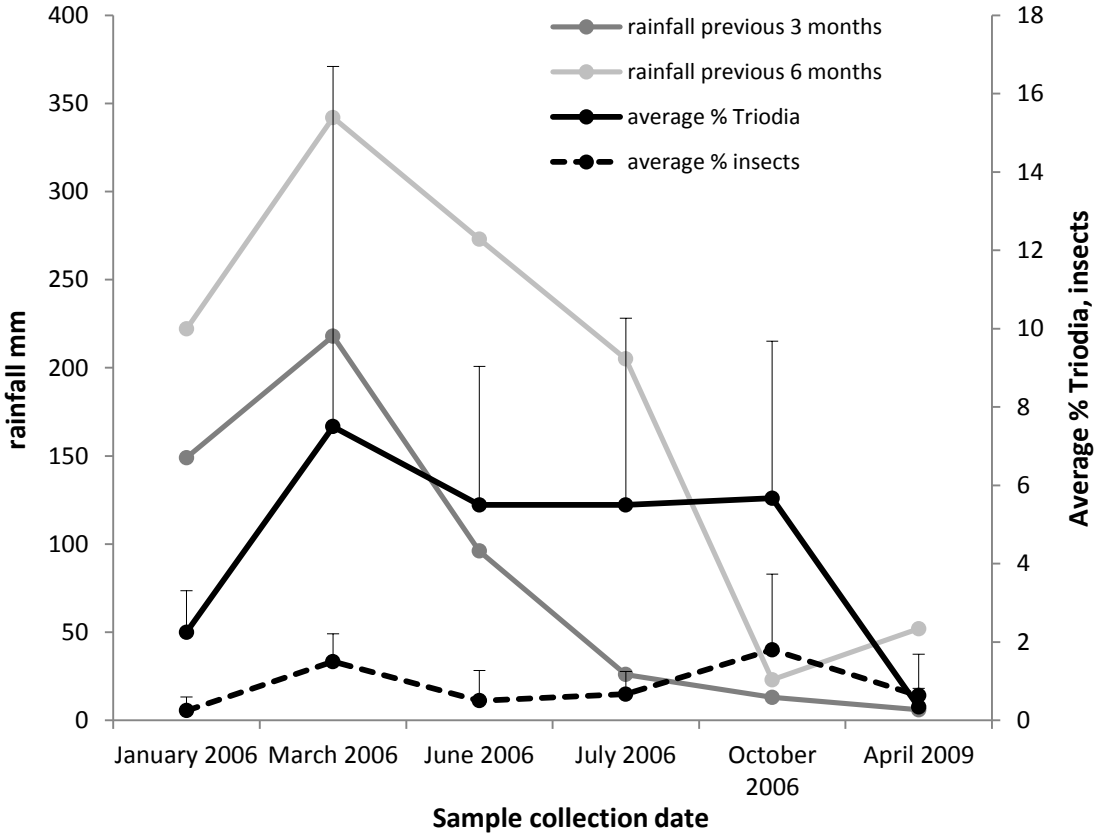

Figure 6.6 Average percentage of both *Triodia sp*. and insects in Uluru mala faecal pellet samples (error bars show standard deviation), and rainfall in the previous three and six months prior to collection

6.3.5 Percentage of supplementary food in samples and population density

There were four occasions when population data were collected around the same time as faecal pellets (Chapter 4), permitting the comparison of average percentage of supplementary food present in pellet samples at increasing mala density (Figure 6.7). Although no correlation existed between these two variables ($r = 0.528$, $t = 0.879$, $d.f = 2$, $n = 4$, $P > 0.05$), an upwards trend in both population estimate and the percentage of supplementary food found in samples was observed over the last two sample collection dates.

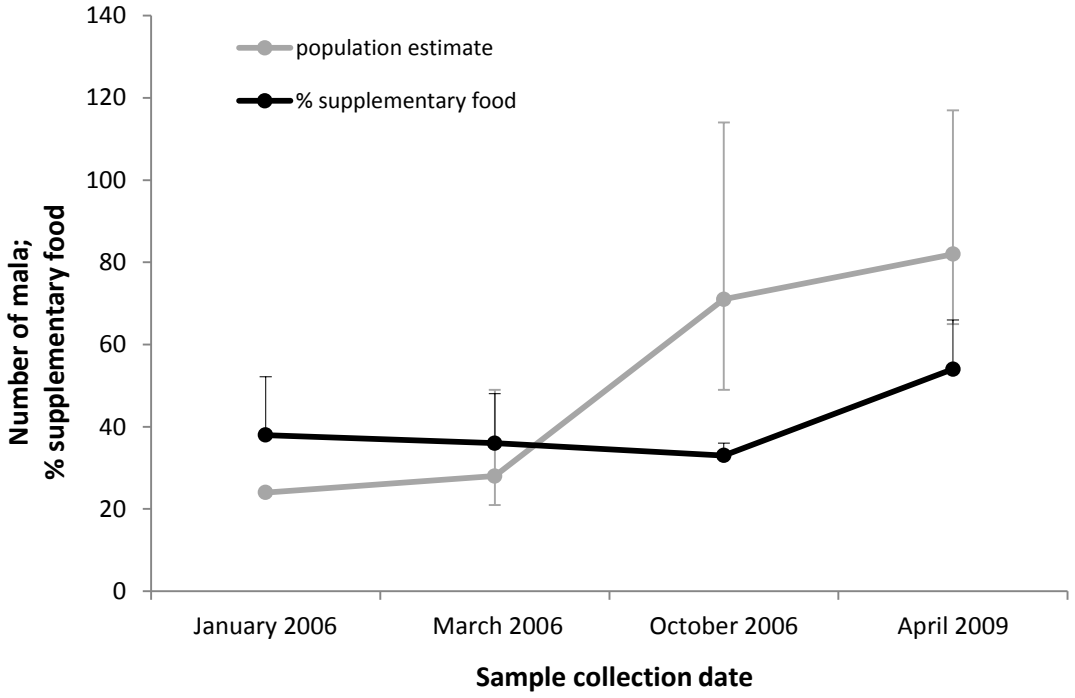

Figure 6.7 Estimate of the number of mala within the enclosure (error bars show lower and upper 95% confidence intervals; January 2005 estimate is based on the number of founders released) and average percentage of supplementary food present in faecal pellet samples (error bars show standard deviation) collected during the survey period

6.3.6 Dietary comparison between Uluru and Tanami Desert: niche breadth

The calculation of Levins' standardized measure of niche breadth permitted the comparison of all four mala dietary studies with respect to the range of food plants consumed. Levin's equation produces a number between 0 and 1, providing a scale between a narrow niche (value nearing 0) to a broad niche (value nearing 1; Pedersen 1999, Pavey et al. 2008). Although all faecal pellet analyses showed that mala browse broadly, the calculation of Levins' measure of niche breadth indicated a relatively narrow dietary range for the species (Table 6.3). It appears that only a small number of plant species (relative to the total species eaten) provided the majority of the food consumed (Table 6.3).

Table 6.3 Levins' standardized measure of niche breadth calculated for the three mala dietary studies undertaken in the Tanami Desert, and my study of Uluru mala

Study	Levins' standardized measure of niche breadth
Pearson (1983)	0.35
Lundie-Jenkins et al. (1993)	0.44
Lundie-Jenkins (1998)	0.24
Uluru study	0.22

Results were similar across the four studies, with Lundie-Jenkins (1998) study and my investigation of mala diet at Uluru producing nearly identical values (Table 6.3).

6.4 Discussion

6.4.1 Dietary comparison between Uluru and Tanami Desert

Prior to its extinction on the mainland, mala occupied a large portion of the western half of continental Australia. The Uluru and Tanami Desert locations where dietary research was undertaken are at the eastern edge of this former range. However, the locations differ in climate as the Tanami Desert study site is sub-tropical, whereas the Uluru site, some 500km south, is within Australia's semi-arid zone. As a consequence, plant species vary considerably between the two locations (Lundie-Jenkins et al. 1993, Pearson 1983, this study). Despite this, the dietary patterns between the two areas have a number of similarities. The wild and reintroduced animals in the Tanami Desert, and those at Uluru, both browsed a broad range of floral taxa (Table 6.4; plant fragments that could not be identified to species level have been omitted from the table). It is interesting to note that both the study of wild mala (Lundie-Jenkins et al. 1993), and my study of the Uluru population, identified 27 individual mala food plants. Further, only two of these floral taxa were common to both analyses. Pearson (1983) also identified a large suite of food plants (28 species) as did Lundie-Jenkins in his study of reintroduced animals (1998, 23 species). Traditional Ecological Knowledge of mala diet within the south-central part of its former range (Chapter 8) identified 24 mala food plants, including 18 species not recorded in other studies (Table 6.4).

Table 6.4 All known mala food plants from faecal analysis (black ticks) and direct observation/evidence of grazing only (grey ticks). Asterisk in Traditional Ecological Knowledge column indicates only one informant stated this plant was eaten by mala (please refer to Chapter 8 for details regarding TEK collection). Plant fragments which could not be identified to genus level have been omitted

Family	**Genus, species**	**Pearson (1983)**	**Lundie-Jenkins et al. (1993)**	**Lundie-Jenkins (1998)**	**Uluru study**	**Traditional Ecological Knowledge**
Moraceae	*Ficus platypoda*					✓*
Proteaceae	*Grevillea juncifolia*		✓			✓
	Grevillea stenobotrya				✓	
	Grevillea eriostachya					✓
	Hakea divericata				✓	
	Hakea lorea (suberea)				✓	
Santalaceae	*Exocarpus sparteus*				✓	
	Santalum acuminatum					✓
	Santalum lanceolatum					✓*
Loranthaceae	*Amyema maidenii*					✓
	Lysiana murrayi					✓
Gyrostemonaceae	*Gyrostemon ramulosa*					✓*
Nyctaginaceae	*Boerhavia coccinia*					✓*
Portulacaceae	*Calandrinia remota*	✓	✓	✓		
	Calandrinia sp.				✓	✓
	Portulaca oleracea					✓*
Chenopodiaceae	*Neobassia astrocarpa*	✓	✓	✓		
	Halosarcia halocnemoides	✓	✓			
	Halosarcia sp.		✓			
	Scleroleana clelandii		✓	✓		
	Scleroleana johnsoni				✓	
Caesalpiniaceae	*Senna sp*					✓*
Mimosaceae	*Acacia dictyophleba*		✓			
	Acacia wiseana		✓			
	Acacia melliodora				✓	
	Acacia ligulata					✓
	Acacia ramulosa					✓*
	Acacia murrayana					✓
Fabaceae	*Indigofera georgei*	✓				
	Tephrosia brachycarpa	✓				

Table 6.4 *continued*

Family	Genus, species	Pearson (1983)	Lundie-Jenkins et al. (1993)	Lundie-Jenkins (1998)	Uluru study	Traditional Ecological Knowledge
	Crotolaria cunninghamii				✓	
	Leptosema chambersii					✓
Amaranthaceae	*Hemichroa diandra*		✓	✓		
	Ptilotus fusiformis		✓			
	Ptilotus latifolius				✓	
	Ptilotus polystachyus				✓	
Stackhousiaceae	*Stackhousia intermedia*	✓				
Malvaceae	*Abutilon otocarpum*			✓		
	Alyogyne pinoniana				✓	
	Sida ammophila				✓	
Sterculiaceae	*Rulingia loxophylla*				✓	
Myrtaceae	*Melaleuca glomerata*	✓	✓			
	Melaleuca lasiandra		✓			
	Aluta maisonneuvii					✓*
Rubiaceae	*Hedyotis pterospora*	✓				
Boraginaceae	*Trichodesma zeylanicum*				✓	
Verbenaceae	*Newcastelia spodiotricha*	✓			✓	
Frankeniaceae	*Frankenia sp.*	✓				
Solanaceae	*Solanum centrale*				✓	✓
	Solanum coactiliferum				✓	✓*
	Duboisia hopwoodii					✓
	Solanum cleistogamum					✓*
Myoporaceae	*Eremophila latrobei*				✓	
Goodeniaceae	*Goodenia virgata*	✓	✓	✓		
	Goodenia cycloptera				✓	
	Goodenia sp.			✓		
Stylidiaceae	*Stylidium desertorum*	✓				
Surianaceae	*Stylobasium spathulatum*					✓
Brunoniaceae	*Brunonia australis*		✓	✓		
Lauraceae	*Cassytha filiformis*		✓	✓		
Asteraceae	*Pluchea tetrantha*		✓	✓		

Table 6.4 *continued*

Family	Genus, species	Pearson (1983)	Lundie-Jenkins et al. (1993)	Lundie-Jenkins (1998)	Uluru study	Traditional Ecological Knowledge
Poaceae	*Aristida browniana*	✓				
	Aristida contorta				✓	
	Aristida holathera		✓	✓		
	Eragrostis eriopoda	✓		✓	✓	
	Eragrostis falcata	✓	✓	✓		
	Eragrostis speciosa	✓				
	Eragrostis sp.		✓	✓		
	Eriachne aristidea	✓	✓	✓		
	Eriachne helmsii				✓	
	Eriachne obtusa		✓	✓		
	Triodia pungens	✓	✓	✓	✓	
	Triodia schinzii	✓	✓		✓	
	Triodia sp.			✓		✓*
	Amphipogon caricinus				✓	
	Cymbopogon obtectus				✓	
	Yakirra australiensis		✓	✓		
	Digitaria radulans			✓		
	Eulalia fulva			✓		
Cyperaceae	*Bulbostylis barbata*	✓				
	Cyperus bulbosus	✓				
	Cyperus concinnus	✓				
	Cyperus conicus	✓				
	Cyperus sp.		✓	✓		✓*
	Fimbristylis caespitosa	✓				
	Fimbristylis dichotoma	✓				
	Lipocarpha microsephala	✓				
Supplementary feed				✓	✓	

Despite the broad range of food plants recorded, niche breadth analysis showed that only a small proportion of these species provide the majority of the food consumed by mala across all studies. The nearly identical niche breadth measures for both Lundie-Jenkins (1998) study and my mala diet investigation are likely due to the provision of supplementary food at both the Tanami and Uluru study sites, and that substantially more supplementary food material was found in faecal pellet samples at both these sites than any other dietary item.

The considerable variation in vegetation between the Tanami and Uluru study sites did not permit meaningful calculations of dietary overlap between the studies. However, mala selected similar types of plants, and a number of the same plant species, in both locations. Five floral species present at Pearson's (1983) Tanami Desert study sites also occurred within the Uluru predator-proof enclosure, namely *Acacia melliodora, Newcastelia spodiotricha, Triodia pungens, Triodia schinzii,* and *Eragrostis eriopoda.* All of these species were consumed by mala at Uluru, and only *A. melliodora* was not eaten by Tanami wild mala (Pearson 1983, 1989). Of the 27 floral species recorded as Tanami mala dietary components by Lundie-Jenkins et al. (1993) *Grevillea juncifolia, A. melliodora, Brunonia australis, Aristida holathera, Triodia pungens* and *Triodia schinzii* are also found at Uluru. Of these, *A. melliodora, T. pungens* and *T. schinzii* were found in Uluru faecal samples. Lundie-Jenkins (1998) identified four species consumed by mala at the Tanami reintroduction site which also occur at Uluru. *T. pungens* and *E. eriopoda* were also eaten by mala at Uluru, however *B. australis* and *A. holathera* were not recorded in Uluru faecal samples.

Similar to my findings, dietary studies undertaken in the Tanami Desert revealed that mala favoured grasses, with *Eragrostis* and *Aristida* species contributing a significant proportion of mala diet (Pearson 1983, 1989; Lundie-Jenkins et al. 1993, Lundie-Jenkins 1998). In addition to consuming similar plant types and species, mala at both Uluru and the Tanami Desert both selected similar plant parts. Leaf was the preferred plant component across all species at both study sites (Pearson 1983, 1989, Lundie-Jenkins et al. 1993, Lundie-Jenkins 1998). Further, Lundie-Jenkins et al. (1993), Lundie-Jenkins

(1998) and Pearson (1983, 1989) concluded that seeds were an important part of mala diet. Considering Uluru mala ate the seeds of 19 different plants, and the contribution of these seeds to overall floral biomass within the enclosure was very small, the same conclusion could be drawn for my study.

The fact that composition of mala diet in both the Tanami Desert and at Uluru appears to be very similar is unsurprising. Although the two sites have climatic and floristic differences, mala would presumably have similar nutritional requirements in both locations. Animals would therefore browse similar taxa to fulfill these needs. The Uluru study therefore confirms the broad-browsing nature of mala (whilst identifying that a relatively small number of species contribute the majority of dietary intake), and provides an additional 20 previously unknown (to microscopic analysis) mala food plants. After the observation I described above (section 6.3.3), *Grevillea stenobotrya* can also be added to this list. Information gathered for Chapter 8, and included in Table 6.4, suggests that further species not identified in faecal analysis are also likely to be consumed by mala within the enclosure. Two of these species, *Amyema maidenii* and *Lysiana murrayi* would be unlikely to be found in faecal pellets for at least two reasons. First, these two mistletoe taxa are found on *Acacia* species (*Amyema maidenii* exclusively) (Moore 2005, Quirico 1999) which are relatively rare within the enclosure. Second, mala would be restricted to consuming the species when plant parts fell to the ground. Indigenous informants also stated that mala consumed the nectar of *Grevillea eriostachya, Grevillea juncifolia*, *Leptosema chambersii*, and *Aluta maisoneuveii*. Lundie-Jenkins et al. (1993) also observed mala consuming *G. juncifolia* in the Tanami Desert. These nectar-bearing plants are common or abundant within the enclosure, however all traces of nectar would be digested by the animal. Unless pollen was also ingested with the nectar, traces of these food plants would not be found in faecal samples.

Variation was apparent between the dietary studies with regard to the contribution of insects to mala diet. Due to the small number of insect remains found within faecal samples during his study period (considered a wet year), Pearson (1983) determined that insects were likely to have been ingested incidentally. Similarly, my study suggests that

insects were not consumed intentionally during the Uluru study period. Conversely, Lundie-Jenkins et al. (1993) suggested that insects were eaten intentionally by mala, and may provide dietary nitrogen during dry periods. In his 1998 study, he stated that the faecal sample preparation technique he employed resulted in the under representation of insect material (Lundie-Jenkins 1998). However, he also concluded that the provision of supplementary food for reintroduced animals may have negated the need for mala to consume insects to fulfill nutritional requirements. This hypothesis may also be true for the Uluru population. Further study is therefore required to test Lundie-Jenkins et al.'s (1993) assertion regarding the importance of insects to mala diet.

6.4.2 Rainfall and dietary composition

As part of ongoing management within the Uluru enclosure, small controlled burns (totaling less than 3 hectares annually) were lit during the study period to ensure suitable habitat for mala. Consequently, in addition to climatic influences, the vegetation available to mala as browse changed due to management action. Although these burns only affected relatively small areas of the enclosure, they should be noted when considering the results of the study.

I did not measure the relative abundance of plants within the mala enclosure, and as a consequence cannot comment on mala food preferences. Rainfall is a key determinant of plant diversity and abundance in the semi-arid environment (Westbrooke et. al 2005). Consequently, mala would presumably experience times of food stress when preferred natural food plants are absent or in short supply, leading to changes in dietary composition. For example, Johnson (cited in Pearson 1983:92) suggested that mala consume proportionally more *Triodia* species when dry periods lead to poor conditions, and this assertion was supported by Lundie-Jenkins et al. (1993). In addition, Lundie-Jenkins (1998) found that supplementary feed comprised a larger proportion of mala diet during such times. Dietary adjustments would presumably also occur to take advantage of times of relative abundance. This was observed in the Tanami Desert study, where mala decreased their use of supplementary feed when preferred natural food plants were available (Lundie-Jenkins 1998). Therefore, as considerable variation in rainfall was

experienced over the Uluru survey period, changes in the average percentage of mala dietary components could be expected. However, no evidence of a correlation between the average percentage of major mala dietary components and rainfall was found. For example, low rainfall in the previous three and six months before samples were collected did not equate to a significant increase in the amount of *Triodia* consumed. Further, higher rainfall did not correlate with an increase in seed consumption. At least two explanations are possible for this result. First, that rainfall fluctuation was too small, or rain fell during the wrong season, to produce changes in the availability of mala food plants. Second, that the consumption of supplementary food (also found not to be correlated with rainfall) provided a constant source of nutrition regardless of environmental flux, and that mala availed themselves of this food to a similar level at all times. This second explanation may be supported by the fact that Uluru mala and their forebears have been held in free-ranging, semi-natural conditions where supplementary food has been provided for over 20 years.

It could be reasonably assumed that as the number of Uluru mala increased over the study period, more pressure was brought to bear on the finite food resources contained within the enclosure. A consequence of this could have been an increased reliance on supplementary food. Although the data showed no significant correlation between mala population estimates and the average percentage of supplementary food found within samples, an upward trend in both mala population size and the percentage of supplementary food found in samples was observed in the last two sampling periods. As the time between these periods was substantial (30 months), this trend could indeed prove significant. Further sampling is required to identify whether this pattern has continued before conclusions can reliably be drawn.

6.4.3 Assessment of study methods

As briefly stated in section 6.2, I chose faecal pellet analysis as the method of mala diet investigation for a variety of reasons. Direct observation of feeding behaviour to assess the dietary composition of relatively cryptic, nocturnal species (such as mala) within a structurally complex environment is highly problematic (Kohn and Wayne 1997,

Valentini et al 2009). Further, the removal of gut content samples from animals, requiring the sedation or killing of the individual, was not considered acceptable due to both the conservation status of mala and limited available resources. However, accuracy problems associated with microscopic faecal analysis have been identified, including the issue of variable digestibility of different species and plant materials (Barker 1986). Such variation may lead to the inability to detect certain food components, and proportional misrepresentation of certain species in faecal pellets (Jordan 2005, Pearson 1983). Although certain foods may not be detected by faecal pellet analysis, Pearson (1983) found no significant difference between the proportions of plant materials found in the stomach and faeces of mala. This finding suggests that although a small proportion of foods may go undetected, microscopic faecal analysis is an acceptable technique to determine the composition of mala diet.

Four samples contained dicotyledenous material from two species that could not be identified. These two species made up between 0.5% and 6% of the samples (average 3.2%, s.d. 2.8). As these species were included in all analysis, as 'unknown dicotyledon A' and 'unknown dicotyledon B', the fact that they could not be identified has no bearing on the overall pattern of mala diet detected; that is that monocotyledenous species and supplementary feed dominate mala diet, and leaf is the most commonly consumed plant part. However, as two samples contained 5% and 6% of 'unknown dicotyledon A' seed, the species appears to be of some local significance to mala. The plant reference collection compiled for the study appears to have lacked these species, and further work would be required to identify these two additional mala food plants.

6.4.4 Management implications

This study was undertaken in part to identify preferred mala food plants that could be monitored to aid the assessment of the Uluru enclosure carrying capacity. Although 27 different species were consumed by mala, nearly all were, by volume, only minor dietary components. The fact that such species may be relatively rare in the enclosure, or show little evidence of mala browsing, would make attempts to identify changes in biomass of such species very difficult. Therefore, I suggest the monitoring of *Aristida contorta* and

Eragrostis eriopoda, as these species are both relatively common in the enclosure, and are eaten in volume by mala. The number of individual plants within set quadrats could be monitored over time, and indices of browsing pressure calculated. Further, the regeneration of these species after management burns could be examined. Such information, coupled with animal health and reproduction data, would permit an assessment of mala population levels with respect to enclosure carrying capacity.

A spatially and environmentally limited captive breeding facility has the potential to affect animal behaviour, including the ability to recognise and obtain natural foods (Chapter 2). Part of the reason for favouring a relatively large, predator-proof enclosure for housing mala at Uluru was to provide a 'natural' environment to encourage the maintenance of 'wild' behaviours. My study of mala diet provided the opportunity to assess whether or not this aim has been achieved with regard to feeding behaviour. Although mala can browse freely within the enclosure, the provision of supplementary food may theoretically change feeding behaviour substantially (Waples and Stagoll 1997, Putman and Staines 2004). Assuming that the supplements are palatable, available to all animals (ie 'dominant' mala do not restrict 'submissive' individuals from accessing supplements), and contain the necessary nutrients for good health, mala may choose to feed exclusively at feed stations. Free access to rich, concentrated and reliable food sources would presumably be very attractive within the semi-arid environment. My study has shown that the food provided at Uluru is indeed a major component of mala diet, but not to the exclusion of natural foods. Mala are still browsing a large variety of plant species, and consuming, on average, as large a percentage of native grasses as supplements. The importance of natural foods is further supported by the fact that six of the samples were collected from the supplementary feed stations (and thus would presumably contain predominantly supplementary feed), and that some mala do not include the feed stations in their home ranges (Chapter 5). Thus, I conclude that mala at Uluru have retained the ability to successfully recognise and obtain natural foods, and that these foods remain an important dietary component despite the presence of supplementary feed. In order to preserve these food recognition and foraging skills, the

Uluṟu population, and future reintroduced populations, should be maintained in large predator-proof enclosures.

A further aim of my study was to provide information, from a dietary perspective, about the suitability of future reintroduction sites for mala in the arid/semi-arid zone. From my findings, suitable releases sites would support a broad range of different food plants including preferred grass species and drought fodder. Suitable shelter species would also be present. The primary importance of grasses of the genera *Aristida* and *Eragrostis* suggests that their presence would be highly beneficial in the release area. Further, *Triodia* would be required to provide nesting sites and potentially feed during prolonged dry periods. Both *Eragrostis eriopoda* and *Arstida contorta* are widespread across semi-arid Australia, as are the genera as a whole (Cunningham et. al 1981, Jessop 1981). Similarly, *Triodia* species cover vast tracts of land (Moore 2005). Therefore, the selection of a release site supporting these taxa at a local level should be straightforward. Thus, a botanically diverse reintroduction site supporting *Eragrostis, Aristida* and *Triodia* would appear to be most suitable to provide for the dietary needs of mala.

6.5 Acknowledgements

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Chapter 7. A Study of the behaviour of mala at Uluṟu – Kata Tjuṯa National Park, and comparisons with behavioural observations from the Arid Zone Research Institute, Alice Springs

7.1 Introduction

The decline of faunal species in the wild may prompt wildlife managers to establish captive breeding colonies in an effort to assist their preservation. Captive colonies provide security against the loss of a species should wild populations become extinct, and successful breeding may produce animals that can found new, or supplement existing, populations (Kelly 1999, Magin et al 1994, Maguire et al 1990).

Several factors may limit the effectiveness of captive breeding for eventual wild release, including changes in behaviour as a result of the artificial captive environment (for discussion of this phenomenon refer to Chapter 2). As the ultimate aim of most captive breeding programs is the eventual release of animals to the wild, such changes in behaviour may negatively impact future translocation success (Lacy 1994, McLean, Lundie-Jenkins and Jarman 1994, Pople et al 2001, Short et al 1992, Snyder et al 1996, Waples and Stagoll 1997). However, activity considered abnormal to that observed in the wild may simply be appropriate reactions to the captive environment, as opposed to an indication of unsuitability for release (Carlstead 1996, Mathews et al 2005). To avoid such erroneous judgment, comparing behaviour of captive bred animals with that of wild conspecifics (where available) in an identical, controlled environment may provide an accurate way of identifying individuals unsuitable for release (Mathews et al 2005).

Currently, captive breeding of threatened Australian macropods can be divided into two broad categories: intensive, highly managed populations housed in zoological parks and

other research facilities; and ‘free range’ populations held within predator-proof enclosures in which animals interact in a semi-natural setting. A variety of advantages and disadvantages are inherent in each of these methods, however this chapter is primarily concerned with behavioural considerations.

Although animal ranging is restricted, large predator-proof enclosures more closely resemble a species’ natural habitat than intensive captive breeding facilities. Within predator-proof enclosures, individuals are able to freely interact with other members of the population (including selecting breeding partner[s] within the behavioural framework of the particular taxon), human contact is minimal, and (depending on population size in relation to carrying capacity) feeding is not necessarily reliant on supplementation. Although predator-proof enclosures are designed to exclude introduced terrestrial carnivores, animals living within the enclosure may still experience predation from aerial hunters (raptors and owls), and some native terrestrial species (such as goannas). Consequently, selection for individuals capable of avoiding certain predators may continue in predator-proof enclosures, whereas this will not occur in intensive captive breeding facilities. The behaviour displayed by ‘free-ranging’ individuals may therefore be a better representation of ‘natural’ activity than that exhibited by animals in intensive breeding facilities. If this were to be the case, it could be argued that animals bred within predator-proof enclosures would be more suitable for release to the wild than their captive bred counterparts. No methodical study of mala (*Lagorchestes hirsutus*) behaviour was undertaken prior to the extinction of the last wild populations in 1991 (Langford and Burbidge 2001). Various factors including low animal density, the physical environment, and the flightiness of the species (Lundie-Jenkins 1993) would have made such a study extremely difficult to conduct. Without such ‘baseline’ information, a comparison between captive bred and wild behaviour is not possible.

Despite there being no record of wild mala behaviour, a detailed behavioural study was conducted by Lundie-Jenkins (1993) within the (now defunct) captive breeding colony at the Arid Zone Research Institute (AZRI) in Alice Springs, Northern Territory. Lundie-Jenkins conducted his research between 1986 and 1988, completing 86 hours of

observation over 22 nights. A raised platform was used to watch mala, which were housed in small groups (3-4 animals) within 9-16m x 6m enclosures. These enclosures were floodlit specifically for the study (Lundie-Jenkins 1993).

The reasons for undertaking this study into mala behaviour at Uluru were twofold. First, to identify previously unrecorded behaviours, and re-examine interpretations of known activities to increase the overall knowledge of the species. Second, to compare and contrast the behaviour of animals held in vastly different conditions to ascertain the effect of the captive environment on mala behaviour. As the AZRI colony was established in 1981, it is likely that the majority of animals observed by Lundie-Jenkins over the period 1986-1988 were not the original wild founders, but rather their descendants. As a result, the mala behaviour displayed may have been modified from 'wild' behaviour by the captive environment. Whilst bearing in mind the differences between 'permanent' behavioural changes and the adoption of 'temporary' activity appropriate to captive surroundings, observed differences between AZRI and Uluru mala behaviour may have management implications. If mala at Uluru display different behaviour to those observed by Lundie-Jenkins, and if it is assumed that the Uluru predator-proof enclosure provides a more natural environment than the pens at AZRI, then it could be argued that behaviourally, mala from predator-proof enclosures are better candidates for translocation to the wild. This reasoning was behind the recommendation made by senior Walpiri traditional owners to the Northern Territory Government that the original predator-proof enclosure at Lake Surprise be built (Langford 2000). Consequently, predator-proof enclosures should be preferred as breeding facilities for mala, and perhaps other small macropods.

7.2 Methods

7.2.1 The captive population

The Uluru mala population was sourced from Watarrka National Park, where the Northern Territory government manages a 120 hectare predator-proof enclosure. The Watarrka population was founded in 2000-2001 by mala translocated from a similar enclosure on the floodplain of Lake Surprise in the Tanami Desert. Captive animals from

the AZRI colony provided the founders for the Lake Surprise population. As a consequence, all of the mala at Uluru observed as part of this study have spent their entire lives in predator-proof enclosures. In the following sections, I take the Uluru population to represent animals that have lived and reproduced in semi-natural conditions for over 20 years, and the AZRI population of Lundie-Jenkins as animals born and bred in captivity.

7.2.2 Recording of raw data

The considerable differences in structure, location and environment between the two observation sites meant that it was not possible to fully replicate the methodology used by Lundie-Jenkins. Where possible, however, similar methods were used.

Observations of mala were made on foot within the UKTNP predator-proof enclosure over the period May to August 2009. Two methods were used to gather behavioural data, with the primary approach being the recording of activity with a video camera (Sony HDRHC9). Using a red-filtered torch for illumination, footage of mala behaviour was captured on digital video tape, and later downloaded onto a laptop computer. In addition, observers recorded oral descriptions of mala behaviour onto a digital voice recorder (Olympus DS-2300). Sound files were then downloaded onto a laptop computer for analysis. Audio notes regarding interactions between animals, and general descriptive information, were also recorded. Early in the research period, an infra-red night vision scope was trialed with the voice recorder. However, it was found that mala could not be easily distinguished through the scope when partially obscured by vegetation, and consequently the technique was abandoned. Two observers collected data during the initial three nights of fieldwork whilst methodology was finalised. The subsequent work was conducted by a single observer. Observations were made over all night hours between sunset and sunrise, however the majority of behaviour was recorded between sunset and 2200. With the exception of the final night, fieldwork was only undertaken on nights where the moon phase was between the 3rd and 1st quarter in an effort to ensure consistent levels of light.

During the fieldwork period, the entire enclosure area was searched for mala. However, animals were only located and observed within 30 metres of the supplementary feed stations and water points. All data was thus collected within a 2.5 hectare area. Although mala have been encountered in all areas of the enclosure during other periods of fieldwork (radio tracking, trapping and incidental sightings), and evidence of mala presence is ubiquitous, walking through the area searching for mala proved fruitless. This inability to locate mala may have been largely due to the vegetation present within the enclosure. In areas other than those burnt under controlled conditions to provide appropriate habitat for mala, the area is thickly vegetated with mature spinifex (*Triodia* sp.) and thryptomene (*Aluta maisoneuvii*) communities. The state of the vegetation may have affected observations in two ways. First, as mala are a physically small macropod (head-body 330mm males, 375mm females; Johnson and Burbidge 1995), they are easily concealed by vegetation. Second, it was very difficult for an observer moving on foot to travel quietly through the enclosure. As mala are easily startled, it may be assumed that some animals moved away from the approaching researcher. Further evidence of the difficulty in observing mala in the broader enclosure is provided by radio tracking studies (Chapter 5). During this fieldwork, researchers closed to within five metres of target animals, however sightings were extremely rare.

7.2.3 Construction of ethogram

After initial behavioural observations were made, an ethogram for *L. hirsutus* was constructed (Table 7.1). This list was supplemented throughout the study as further fieldwork resulted in the discovery of additional behaviours. Where a particular behaviour correlates with that defined by Lundie-Jenkins (1993), and where I agree with the description provided, this definition is presented.

Table 7.1 Mala behaviour ethogram derived from observations at UKTNP

Name of behaviour type	**Description**
Movement	
Rapid	Fast-paced, bipedal movement where upper body is held in an approximately horizontal position. Movement may include rapid changes of direction.
Moderate	Bipedal movement where gait is noticeably slower, appears more 'balanced' and is directionally less erratic than the 'rapid' movement. Upper body is held in an approximately horizontal position.
Slow	This behaviour is the same as that defined by Lundie-Jenkins as 'investigative' (1993:31): "a slow walk with four paws placed on the ground and the hind feet brought forward slowly in unison beneath the body. During this movement, the hind feet extended forward to a position where their centres were adjacent to, and outside the forepaws. The tail appeared to provide some support in this gait…".
Single jump	Animal uses hind feet to jump forwards up to a height of 25 centimetres from a standing position, or during the 'slow' movement, to clear an obstacle.
Standing	
Startled	This behaviour is the same as that defined by Lundie-Jenkins (1993:31) as 'alert': "A bipedal stance with the vertebral column perpendicular to the ground and forearms slightly extended forward from the body." May be accompanied by a rapid ducking motion where head is pulled downward until upper spine is near horizontal.
Alert	This behaviour is the same as that defined by Lundie-Jenkins as 'undisturbed': "A bipedal stance with the vertebral column curved ('hunched') and forearms held loosely in front of the body". Although not common, may be accompanied by a rapid ducking motion where head is pulled downward until upper spine is near

	horizontal.
Feeding and drinking	
Feeders	This behavioural category includes all activities associated with procurement and ingestion of commercially produced 'kangaroo pellets' at supplementary feed stations. Specific behaviours include obtaining, manipulating and chewing food, and changes in posture associated with these activities whilst stationary.
Natural	This behavioural category includes all activities associated with procurement and ingestion of naturally occurring vegetation. Specific behaviours include obtaining, manipulating and chewing food, and changes in posture associated with these activities whilst stationary.
Drinking	Any activity associated with acquiring water from artificial watering points.
Grooming	
Scratching with hind foot	This behaviour involves use of one hind foot to scratch neck region, side of head, ear, upper chest area, shoulder area.
Grooming with forepaws	This behaviour involves use of forepaws to scratch scrotum, chest/abdomen, flank area, and muzzle. It also includes the use of a single forepaw to scratch mid or lower back area, head/neck.
Licking	This behaviour involves the licking of forepaws in both sexes, and in females licking the inside of the pouch or pouch opening.
Unknown grooming	This behaviour involves grooming where the precise nature of the grooming activity cannot be determined (e.g. animal turns upper body away from the observer to groom back of body).

Other	
Investigative sniffing	This behaviour involves a stationary animal sniffing the ground or vegetation, and may include animal grasping vegetation with forepaws.
Interactions	
Investigative sniffing – interaction	During this behaviour, one animal approaches another and sniffs at the posterior of its body.
Mutual investigative sniffing	During this behaviour, two animals simultaneously sniff each other's noses.
Aggressive approach	This behaviour involves an animal directly approaching, and moving into the immediate space of, a second animal, resulting in the second animal moving away.
Brief chasing	This behaviour involves an animal chasing a second animal for a distance of less than 5 metres.
Tail waving	During this behaviour, the focal animal, whilst in the presence of a second animal, stands with body horizontal to the ground and moves tail laterally in a wave-like movement. Vertical tail movement in a similarly fluid fashion may also occur.

7.2.4 Splitting data into bouts and behaviours

Initially, the behaviour of all mala encountered (where technically possible) was recorded. However, during the last night of fieldwork, priority was given to instances when two or more mala were present in a given area. This approach was taken for two reasons. First, mala appeared to be behaving 'predictably'; that is no new behaviours were being observed. Second, few interactions had been recorded during the study period. This data has been omitted in the following discussion regarding the proportion of total time spent on each behaviour.

After data from both audio and video recordings were downloaded onto a laptop computer, individual 'bouts' of behaviours were identified. I define a 'bout' as a temporal period of behaviour(s) performed by a particular individual. Each bout commenced when the focal animal changed behaviours from the type being exhibited when observations were first initiated. This ensured inaccuracies in behavioural frequency calculations were avoided by excluding fragments of particular behaviours. If, during a bout of activity, an individual was lost from sight, or was obscured to the point where behaviour could not be confidently identified, the bout was terminated and a second bout commenced when unobstructed vision returned. This permitted the accurate determination of the relative frequency of each behavioural type during a bout. Mala behaviour undertaken during each bout was measured to the second using a digital stopwatch, and allocated to particular categories as described in the ethogram. In addition to this quantitative data, vocal descriptions of observations of interest were recorded.

7.2.5 Data Analysis

The relative frequency of each mala behaviour observed at Uluru was analysed using the Kruskal-Wallis test, and significance determined using a non-parametric multiple-comparison procedure (Dunn 1964). In order to compare mala behaviour at Uluru and AZRI, comparative behavioural categories were created to correspond to Lundie-Jenkins (1993) methods. The Wilcoxon Rank Sum test was used to identify significant

differences between the mean percentages of total activity time per mala behavioural category at the two sites.

7.3 Results

Over 10 nights, a total of 34.5 hours of fieldwork yielded just under 4 hours of quantitative mala behaviour observations (Table 7.2). Due to the cryptic, flighty nature of the species, and the state of the vegetation within the enclosure, no data on the activity of mala within or near squats were recorded, and little 'wild' browsing was witnessed. Consequently, this study only provides information on the active, ranging aspect of mala behaviour within an area providing focused, rich food sources.

Table 7.2 Summary of UKTNP mala behaviour data collection effort, result and relevant astronomical data (Geoscience Australia 2005)

Date 09	Number of Observers	Number of Bouts Observed	Total Minutes Recorded	Mean bout length	Standard Deviation	Sun-rise	Sun-set	Moon-rise	Moon-set	Moon phase
19/05	2	30	24	0.78	0.68	0717	1807	0148	1416	last quarter
20/05	2	41	52	1.27	1.18	0718	1807	0242	1448	last quarter
21/05	2	25	24	0.96	1.74	0718	1806	0338	1522	last quarter
25/05	1	10	12	1.2	1.28	0720	1805	0759	1838	new moon
26/05	1	24	23	0.96	1.37	0721	1805	0906	1944	new moon
21/07	1	34	30	0.88	1.03	0729	1815	0629	1717	last quarter
29/07	1	16	11	0.69	0.60	0725	1819	1157	0046	first quarter
30/07	1	28	37	1.32	1.06	0725	1820	1238	0144	first quarter
01/08	1	15	25	1.67	1.66	0724	1821	1411	0336	first quarter
11/08	1	7	22	3.19	2.40	0717	1825	2304	0953	full moon

The total time spent by all mala undertaking all behaviours except drinking (225 bouts), and the percentage of total observation time per behaviour witnessed, is shown in Table 7.3 and Figure 7.1. Bouts which included drinking were not included, as this behaviour could only be observed when researchers intentionally waited at supplementary water drippers. Inclusion of these bouts would therefore not be an accurate representation of the relative frequency of this behaviour.

Table 7.3 Total time spent undertaking all behaviours by mala observed at UKTNP

Name of behaviour type	Total time spent undertaking activity (seconds)	Percentage of total observation time
Movement		
Rapid	46	0.32
Moderate	319	2.23
Slow	2214	15.50
Single jump	11	0.08
Standing		
Startled	69	0.48
Alert	7140	49.99
Feeding		
Feeders	2905	20.34
Natural	68	0.48
Grooming		
Scratching with hind foot	41	0.29
Grooming with forepaws	49	0.34
Licking	188	1.32
Unknown grooming	19	0.13
Other		
Investigative sniffing	1117	7.82
Interactions		
Investigative sniffing - interaction	28	0.20
Mutual investigative sniffing	28	0.20
Aggressive approach	15	0.11
Brief chasing	20	0.14
Tail waving	6	0.04

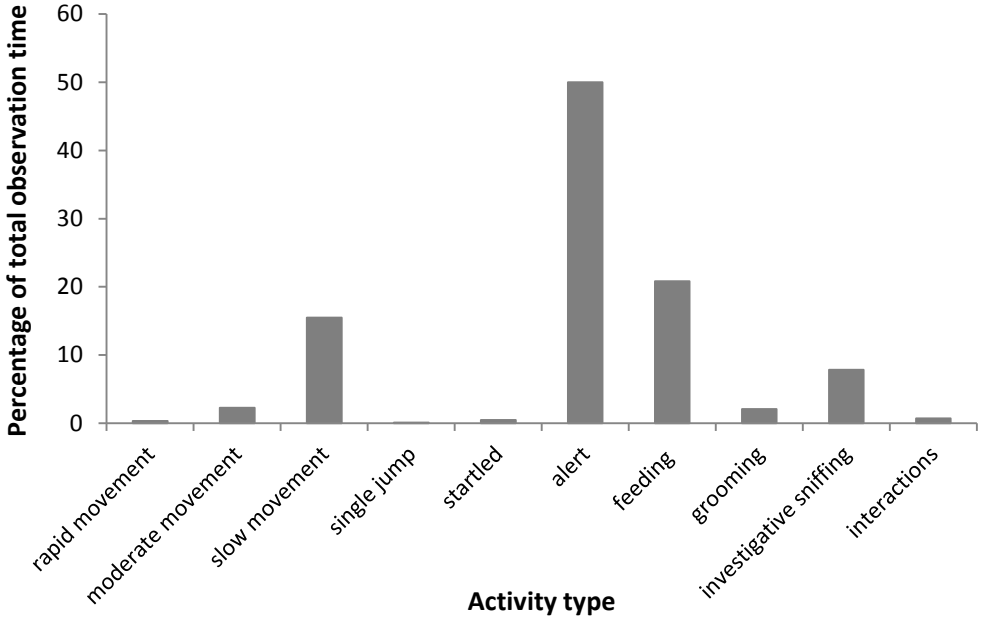

Figure 7.1 Percentage of total observation time spent by mala at Uluru undertaking each behaviour type/group

7.3.1 Relative frequency of behaviour

Significant differences were detected in the times mala spent performing each of the recorded behaviours (Kruskal-Wallis test). The most commonly recorded behaviour was the 'alert' state, which accounted for nearly 50% of total observation time, and was performed significantly more than all other behaviours with the exception of feeding (multiple-comparison procedure, Dunn 1964). The time spent feeding at feed stations (20%) was also significantly greater than that devoted to the 'rapid movement', 'standing jump', 'startled' and 'interactions' behaviours. 'Slow movement' (15.5% of total observed time, performed significantly more than 'single jump', 'startled', 'interactions' and 'rapid movement' behaviours) and 'investigative sniffing' (7.8%) were the only other behaviours which accounted for greater than 5% of the total observation time. 'Rapid movement' and 'moderate movement', the 'startled' posture, grooming behaviours, those associated with interactions, and feeding on natural vegetation were rarely observed. The 'tail waving' behaviour was witnessed only once.

7.3.2 Peak activity periods

No rigorous investigation was made into peak activity periods during this study. However, through the course of fieldwork, mala appeared to be more active during two periods; between one and three hours post-sunset, and between 3.5 hours and 1.5 hours pre-sunrise. Although during these periods mala were seen in higher numbers, the hours between these apparent activity peaks also yielded observations.

7.3.3 Vocalization

Alarm squeaks were often heard during fieldwork. These utterances were short and high pitched in nature, and were emitted only singly. Generally, a fleeing animal did not produce a squeak when initially startled, but rather moved at rapid pace for 5-10 metres before emitting the alarm vocalization. On one occasion outside the fieldwork period a low growl was heard. Although it could not be positively established that mala made the sound, it is the most likely candidate considering this vocalization has been recorded for *L. hirsutus*, (McLean et al 1994) and there are no other species likely to produce this sound present within the enclosure.

7.3.4 Boldness/flightiness

The vast majority of mala at Uluṟu showed a high sensitivity to the presence of humans and conspecifics. Although all mala observed have spent their entire lives in some form of predator-free environment, they remained alert to perceived threats. Alarm squeaks uttered by individuals would trigger the 'alert' posture in conspecifics. Mala were easily startled by conspecifics, and on one occasion the arrival of a small rodent, the spinifex hopping mouse *Notomys alexis* (body mass average 35 grams; Breed 1995), caused a mala to flee. Generally, whilst observing an individual at relatively close range (up to approximately five metres), the researcher could not move positions without causing the subject to move rapidly away, at times emitting an alarm squeak.

There were, however, notable exceptions. On several occasions, sub-adult animals feeding at the supplementary feed stations appeared unperturbed by the presence of the observer. I was able to walk slowly past a feeding sub-adult without triggering a flight

response (quantitative behavioural data were not recorded during this time). Sub-adults also appeared less concerned by the approach of other mala. Further, on rare occasions, mala would slowly approach the stationary observer, passing within less than a metre. Not all animals which displayed this behaviour were sub-adults.

Most observations were undertaken at a distance of 3-15 metres from the subject. At these distances, mala appeared to be aware of the presence of the observer, and would often look in their direction whilst carrying out other behaviours. Red-filtered light was used to illuminate the subject, as there is some evidence that red light is less intrusive to macropods (Wolf and Croft 2008). Indeed, the use of red light during capture and handling of mala appeared to reduce the stress levels of animals when compared to unfiltered 'white' torch light (pers. obs.). Although the use of artificial light clearly affected mala behaviour, no other practical method was available that would eliminate discovery of the observer by mala.

7.3.5 Behaviour patterns

Over the course of a bout, the focal mala characteristically displayed a suite of behaviours, some of which were displayed multiple times. For example, when moving slowly over a distance of several metres, a mala typically used the 'slow' gait for a few seconds, then interrupted the movement by adopting the 'alert' posture. After a pause, the animal would resume the 'slow' gait for a few seconds, before beginning a short period of 'investigative sniffing'. The alert posture was sometimes then adopted before an animal moved on. Thus, short periods of many different behaviours all interspersed with one another appears to be typical of the species.

By definition, some actions were particularly short in duration. The 'single jump' always took less than a second, and 'brief chasing' was always concluded in under 4 seconds. Interactive behaviours were also short, with the longest recorded being just under 4 seconds in length. Conversely, although some other behaviours could also be conducted for brief periods, on other occasions the same actions were undertaken for considerable lengths of unbroken time. Several mala adopted the 'alert' stance for more than 60

seconds, with the longest recorded being 97 seconds. Relatively long periods of 'investigative sniffing' were also observed, the longest lasting nearly 39 seconds.

Feeding from the stationary feeders was consistently carried out for longer periods than any other behaviour, but was highly variable. The average length of feeding behaviour was 40 seconds (n = 71, s.d. ranging from 0.1 to 101.6), however eight feeding periods lasted over a minute, four feeding periods were over three minutes in duration, and one greater than four minutes. On one occasion, a mala was observed feeding for over eight minutes, but this individual's feeding behaviour was not included in the data, as it had commenced feeding prior to the arrival of the observer.

7.3.6 Interactions

Only 24 periods of interactive behaviour between mala ('investigative sniffing – interaction', 'mutual investigative sniffing', 'aggressive approach', 'brief chasing'), were quantitatively recorded. In the following presentation and discussion regarding mala interactions, the terms 'dominant' and 'subordinate' are used to describe aggressive and passive individuals. Unintentional 'interactions', such as the inadvertent frightening of one animal by another, are also discussed.

By far the most common observation when more than one mala was present in a particular location was individuals appearing to ignore each another. Animals were often observed feeding side by side at the supplementary feed stations, seemingly unperturbed by the presence of others. On occasions, I witnessed up to four animals feeding simultaneously. Two other interactive scenarios, although relatively uncommon when compared to the 'ignore' response, were the next most frequently encountered. The first involved an individual mala moving away from a second animal without overt aggression by the 'dominant' animal. The second involved overtly aggressive behaviours. Non-aggressive interactions generally involved a mala approaching a second stationary individual at a supplementary feed station. On numerous occasions, the seemingly non-aggressive approach of a second mala resulted in a feeding individual moving away. For example, one 'pace' at 'slow' movement by an aggressor towards the subordinate animal,

located 1.5 metres away, resulted in that individual moving still further away at rapid pace. In contrast to these relatively benign interactions, aggressive exchanges between individuals were observed. Animals present at feeding stations would chase new arrivals away, usually for only a short distance (up to five metres). During one bout, an aggressive animal chased an approaching individual(s) away from the feed station on three successive occasions, apparently actively defending its exclusive access to the food source. In addition, a longer chase was witnessed, with one mala pursuing another over 10-15 metres before both were lost from view. As this action occurred approximately 15 metres from the nearest feed station, it is unclear what triggered the chase. Interestingly, not all mala present during aggressive exchanges between individuals were involved, or affected, by the behaviour. On one occasion, a sub-adult mala was left to feed unhindered whilst a series of aggressive interactions occurred between other adult mala present.

Not all chases appeared aggressive in nature. For example, two mala were seen feeding approximately 30 centimetres apart; one female with a visible offspring in the pouch, whilst the gender of the second animal could not be established. When the female moved away from the feed station, the second animal followed, eventually closing in to within touching distance. At this point, the mother moved further away at a moderate pace. Although it appeared that the mother did not want to be in close proximity to the second animal, the interaction was quite different to the aggressive chases also observed.

After interactions involving dominant and submissive animals, the seemingly unintentional frightening of one animal by another was the next most frequently observed interaction. During one incident, a mala adopting the ‘moderate’ gait resulted in a nearby animal fleeing. This did not seem to be an aggressive movement; rather it appeared simply that the movement alone startled the second animal. On two occasions, the approach of a third animal resulted in two feeding animals taking flight. During these incidents, the arrival of the third animal appeared to frighten the other two mala, even though the approach was not aggressive in nature. Conversely, one animal was observed approaching a feeding mala relatively rapidly, which caused no reaction in the stationary

mala. 'Mutual frightening' was also observed. The arrival of an individual at a feed station sometimes resulted in both the approaching and feeding animals fleeing when they became aware of each other's presence. On other occasions, after appearing to notice one another for the first time, both animals would move off at 'moderate' pace.

Aggressive mala appeared, at times, to tolerate the presence of 'subordinate' animals. On two occasions, individuals were seen to chase a second mala away from a feed station, but permit the second animal to return and feed in close proximity. In one instance, after a period of time the two individuals engaged in 'mutual investigative sniffing', after which the second animal again moved away. The aggressive animal repeated this behaviour a short time later, permitting a second mala which it had previously chased away to return and feed.

The single incident of 'tail waving' recorded at Uluru involved an individual using the 'slow' gait, interspersed with 'alert' posture, to approach a feed station. As it neared the feed station, the focal mala appeared to become aware of the presence of another, and then proceeded with the 'tail waving' behaviour as described in the ethogram (Table 1). After a few seconds of this activity the focal animal was aggressively chased away by the second individual.

7.3.7 Drinking

Although drinking has not been recorded in wild populations (Lundie-Jenkins 1993), mala at Uluru were observed using the water points provided. These water points comprise a pressure regulator and 'dripper' valve, held at mala head-height by a steel frame. A total of five bouts of behaviour including drinking were recorded. During these bouts, mala would lick the dripper valve to obtain water, alternating this behaviour with the 'alert', 'investigative sniffing' and 'slow' behaviours. Mala obtained water for between one and 12 seconds at a time (average 5.6 seconds, s.d. 3.1).

7.3.8 Incidental observations

Two behaviours not witnessed during data collection for this chapter were observed during other fieldwork. On one occasion, a female mala with young at foot were encountered for a short period of time. During observations, the offspring, whilst standing perpendicular to the female, put its head in the female's pouch and presumably began suckling. On a second occasion, a mala was observed slowly approaching a female stationary at one of the feed stations. When within approximately 1.5 metres of the female, the approaching mala stopped and assumed the 'alert' posture. This animal then stood upright, thrust its pelvis forward, and took one pace forward with its right hind leg. It maintained this posture for a short period of time (less than 10 seconds), then crouched down and began sniffing at the ground where it had been standing. After moving into the 'alert' posture, the animal again assumed an upright position, thrust forward its pelvis, and took a step forward with the right hind leg. The animal then repeated the sniffing behaviour noted above. After completing this activity, the animal moved towards the female, which moved away at moderate gait. The animal then pursued the female at moderate pace for over 5-10 metres until they were lost from view.

7.3.9 Comparisons with Lundie-Jenkins' study of mala at AZRI

Comparative behavioural categories for the Uluru study were created to correspond with Lundie-Jenkins' (1993) seven mala behavioural categories (Table 7.4). The mean percentage of total activity time for each behaviour at AZRI could then be plotted against the relative behavioural categories at Uluru (Figure 7.2).

Table 7.4 Corresponding mala behavioural categories – AZRI and UKTNP studies

AZRI behaviour	Comparative behavioural category for Uluru study
Nesting	No nesting behaviour observed
Feeding	Feeding – feeders, Feeding – natural
Investigative	Slow movement, investigative sniffing
Traveling	Moderate movement, rapid movement, single jump
Grooming	Unknown grooming, self-grooming with hind foot, self-grooming with forepaws, licking
Standing	Alert, startled
Interacting	Investigative sniffing – interaction, mutual investigative sniffing, aggressive approach, brief chasing

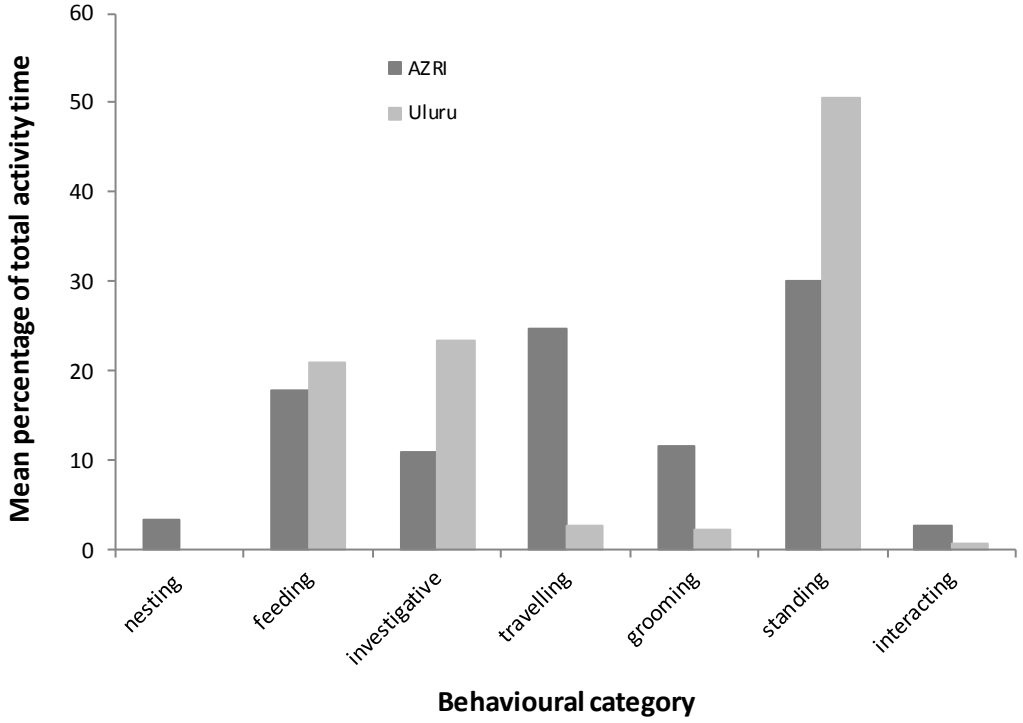

Figure 7.2 Mean percentage of total activity time for each behaviour at AZRI, with the comparative behavioural categories at Uluru

The data show differences between the mean percentage of total activity time per behavioural category between AZRI and Uluru (Figure 7.2). Two-tailed Wilcoxon Rank Sum test results revealed all differences with the exception of 'feeding' to be statistically

significant (‘investigative’ Z = -3.18, p = 0.0014; ‘traveling’ Z = 3.18, p = 0.0015; ‘grooming’ Z = 3.18, p = 0.0014; ‘standing’ Z = -3.18, p = 0.0014; ‘interacting’ Z = 2.24, p = 0.025). As mentioned above, no nesting activity was observed at Uluru. However, mala at AZRI spent only a relatively small amount of time undertaking this activity, and direct observation of nests located at Uluru would be required to permit comparison with the AZRI data.

7.4. Discussion

7.4.1 Frequency of behaviours displayed by mala at UKTNP

Animals that live in groups derive benefits from the herd environment including increased overall predator vigilance and the sharing of warning signals (Kaufmann 1974). As mala are generally solitary animals (Lundie-Jenkins 1993), they do not enjoy such advantages. Considering the constant need to be aware of predators (perceived or otherwise), competitors and conspecifics, and the general ‘flightiness’ observed in the species, it is not surprising that the ‘alert’ stance was the most commonly observed. In contrast, the ‘startled’ posture was infrequently seen. However, mala were frequently observed using the ‘rapid’ movement as an immediate reaction to a perceived threat, as opposed to adopting the ‘startled’ pose prior to flight. The ‘startled’ posture may therefore be an increased state of vigilance adopted when danger may be imminent, but not immediate. The fact that mala were frequently seen feeding is no doubt a product of the fact that the majority of observations were conducted in the vicinity of the supplementary feed stations. ‘Investigative sniffing’ appeared, in most cases, to be linked to feeding, and consequently this behaviour type was also commonly observed. Little quantitative data regarding ‘rapid’ movement was recorded. This is most likely due to mala appearing to engage in this behaviour relatively infrequently, and that individuals would often disappear from view whilst exhibiting this behaviour making it impossible to record the duration of the behaviour.

Interactions between mala accounted for less than 1% of observed behaviours recorded during the study. Considering mala are generally solitary animals, this lack of interaction is not unexpected. However, as the supplementary feed stations are a rich and reliable

food source, they undoubtedly act as a focal point for mala. Consequently, a higher animal density than that found in 'wild' populations would be expected in the vicinity of the feed stations. Although groups of up to four or even five animals were often seen at these points, interactions of an aggressive nature were uncommon. When they were witnessed, aggressive interactions appeared to be driven by access to the food source. From the study data, it could be argued that some form of hierarchy within the Uluru mala population exists, as presumably aggressive interactions would not occur if no such structure was present. Further, their infrequency may indicate an existing social structure in which dominance was established and respected. The presence of concentrated, high-quality food sources at Uluru is likely to encourage the expression of such hierarchical behaviour.

7.4.2 Comparisons with Lundie-Jenkins' study of mala at AZRI

Although mala within each population displayed a similar suite of behaviours, statistically significant differences in occurrence of behaviour between Uluru and AZRI were observed for all but one behavioural category. Assuming predator-proof enclosures provide a more natural environment than small breeding pens, these differences may have implications for the sourcing of mala for future translocation efforts. However, reasonable explanations for these differences also require consideration.

Regarding all behaviours, the amount of time spent 'traveling' showed the largest discrepancy between AZRI and Uluru mala. Several factors may explain this difference. First, although mala at Uluru have an enormous area in which to travel when compared with the AZRI pens, animals could only be satisfactorily observed within a small proportion (2.5 ha) of this space. Therefore, much mala 'travel' time could not be identified. As part of radio tracking studies at Uluru, animals nesting over 700 metres away were recorded at supplementary feed stations. If movement time between nest sites could have been recorded at Uluru, 'traveling' behaviour would no doubt be a more significant component of the overall time budget. Second, AZRI animals were more likely to encounter other mala during all activity times as a consequence of the small size of the enclosures. Such encounters may lead to 'traveling' behaviours, as mala avoid

aggression, or other undesired attention, from conspecifics. Third, the confined nature of the AZRI pens may have triggered stereotypical locomotive behaviours in the mala. Animals restricted by space and limited stimuli have been known to exhibit such obsessive and repetitive behaviours (Jackson 2003, Swaisgood and Shepherdson 2005), although no mention of this phenomenon was made by Lundie-Jenkins (1993). Frequent interaction with other mala, and the potential for mala to be expressing stereotypical behaviour within the small AZRI pens, may also explain the difference in time spent undertaking the 'standing' behaviour between AZRI and Uluru.

AZRI mala conducted investigative behaviour for only half the amount of time spent by Uluru animals. This discrepancy may be due to the relatively small size of the AZRI enclosures. Presumably, the 3-4 AZRI mala housed together would quickly become familiar with their limited captive environment. Uluru mala, however, had a substantially larger area in which to move, and would presumably encounter a larger number/range of olfactory stimuli over the course of a night. Conversely to investigative behaviour, animals at AZRI groomed for over five times as long as those at Uluru. Lundie-Jenkins (1993) observed that the majority of this activity occurred when mala first emerged from their squats, although shorter periods of grooming were also recorded during the remaining hours of activity. Thus, the discrepancy regarding grooming between the two populations may be due to the fact that Uluru mala were not observed undertaking this initial bout of grooming. Another explanation may be that the time spent on this activity is another manifestation of stereotypical behaviour, as excessive grooming been identified in captive mammals (Carlstead 1996).

Differences in the spatial environment between AZRI and Uluru are likely to be the cause of the significant difference in time spent interacting with conspecifics. Although the supplementary feed stations at Uluru provided a focal point for mala, sufficient area was available for animals to avoid interaction if desired. Conversely, the very limited space available to mala at AZRI would presumably have made avoiding interactions impossible.

In addition to the difference in time spent undertaking particular behaviours between the two populations, a number of behaviours noted by Lundie-Jenkins (1993) were not witnessed at Uluru. None of the complex behaviours between female and offspring noted by Lundie-Jenkins (1993) were recorded, although an incidental observation of a young at foot suckling from outside the pouch was made during other work. Mutual grooming, infrequently observed by Lundie-Jenkins (1993), was not encountered, nor were the 'sexual checking' and 'attempted mating' behaviours. The 'extremely violent' interactions witnessed between 'mature animals of one sex and sub-adults or juveniles of the other sex' (1993:31) at AZRI were also not observed at Uluru. As interactions of any description between mala were infrequently seen at Uluru, it is not surprising that the breadth of interactions witnessed within the small, relatively densely populated confines of the AZRI pens were not recorded.

Lundie-Jenkins describes alarmed mala emitting loud squeaks and fleeing for cover 'releasing a drawn-out hissing sound' (1993:31). Although alarm squeaks were common, this hissing was not heard during observations at Uluru. However, mala held in hessian bags during trapping work often breathe heavily producing such a sound. The stamping of hind feet occasionally observed by Lundie-Jenkins (1993) prior to fleeing, was also not witnessed at Uluru. This may again be a consequence of the ease of observation at AZRI, rather than the lack of such behaviour at Uluru. A single behaviour type, 'tail waving', was recorded at Uluru but not noted during Lundie-Jenkins (1993) study. Either this activity was not witnessed at AZRI, or it was not explicitly described. Considering the detail with which Lundie-Jenkins described all other aspects of mala behaviour, and the apparent rarity of 'tail waving', it is more likely that this particular action was not seen at AZRI.

Differences in methods between my study and that of Lundie-Jenkins also made other comparisons of behaviour difficult. Unlike the AZRI study, the gender of most Uluru mala could not be determined. Consequently, attempting to interpret interactions between individuals was problematic. Lundie-Jenkins (1993:31) describes female non-aggressive interactions thus: 'The dominant female would continue to move closer to and

investigate the subordinate female until the latter moved off.' Similar behaviour was definitely observed during the Uluru study, however the gender of all animals involved in such interactions was unknown. Lundie-Jenkins (1993) described male-male interactions as mutual fleeing, chasing and direct fights. The Uluru study witnessed mutual fleeing and chasing, but no actual physical contact between animals. Again, the gender of the animals involved could not be established. As Lundie-Jenkins (1993) noted that males would avoid confrontations within the relatively small confines of the AZRI pens, it is perhaps not surprising that no physically violent interactions between mala were witnessed. However, the incidental observation noted above involving an individual mala in the presence of a female standing with its pelvis thrust forward is likely part of male-female interaction. Lundie-Jenkins observed male mala place their forepaws on a females back and 'stand upright with his pelvic region thrust forward' (1993:32). The Uluru observation ended with female being approached, then chased, by the other. It is therefore plausible that this interaction was a female rejecting mating behaviour displayed by a male.

Lundie-Jenkins (1993) concludes that although mala are predominately solitary animals, observations suggest that dominance interactions within gender groups do occur. Drews (1993) defines dominance as '…an attribute of the pattern of repeated, agonistic interaction between two individuals, characterised by a consistent outcome in favour of the same dyad member and a default yielding response of its opponent rather than escalation'. Gregarious animals that form relatively stable groups are likely to form dominance relationships between individuals, which in turn create hierarchies within the greater group. Lundie-Jenkins questions whether dominance relationships comparable to those observed in captivity would be present in wild populations. This is based on the assumption that AZRI animals were held in higher densities than would be found in the wild, and limited fieldwork indicating that males occupy exclusive home ranges which overlap with a number of females. Although the density of AZRI mala was undoubtedly artificial, radio tracking work at Uluru indicates that male home ranges do indeed overlap (refer Chapter 5). Thus, male-male interactions are perhaps more common than Lundie-Jenkins assumed. However, regardless of this fact, they are unlikely to be more than

simple, dominant/submissive relationships between a relatively small number of frequently encountered individuals. Therefore, this study agrees with Lundie-Jenkins' conclusion that mala are unlikely to form strong dominance hierarchies in the wild.

7.4.3 Management Implications

No behavioural study of wild mala was undertaken prior to the species' extinction in the wild. Consequently, there is no way of knowing whether the behaviour displayed by mala at Uluru, or that displayed by animals at AZRI, is most similar to that of their wild forebears. Potentially, both Uluru and AZRI mala expressed an altered/depauperate suite of behaviours as a consequence of generations in their respective facilities. Further, mala at both study sites may simply have been displaying behaviours suitable to their particular environment, rather than exhibiting permanent changes from 'true' wild behaviour. If this was the case, mala transferred from Uluru to AZRI would presumably behave similarly to those raised in the intensive captive breeding facility, and animals taken from AZRI to Uluru would similarly adjust their behaviour accordingly. However, although reasonable explanations have been provided to account for the discrepancies between mala behaviour at Uluru and AZRI, the potential for these differences to indicate inappropriate behaviour unsuitable for the wild in captive bred animals should not be ignored. Within predator-proof enclosures mala are free to exhibit and maintain (within the spatial limitations of the enclosure) 'natural' behaviours such as recognising and obtaining natural foods, establishing home ranges, and interacting with both conspecifics (including the selection of breeding partners) and other taxa. Population surveys of the Uluru mala have shown strong growth and healthy individuals. Considering these results, and the potential for behavioural changes to emerge within populations held within small, captive breeding facilities over the long term (McPhee 2003, Waples and Stagoll 1997) I recommend that to maximise the chance of future reintroduction success mala populations should be housed within predator-proof enclosures.

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Chapter 9. Recommendations for the management of existing, and establishment of future, mala populations in arid / semi-arid mainland Australia

9.1 Introduction

One of the primary aims of undertaking the work contained in this thesis was to provide practical recommendations for the management of existing, and establishment of future, mala populations. As both threatened species translocation and subsequent population management are complex, multi-faceted undertakings, this thesis has explored a broad variety of issues.

Following the introductory Chapter 1, Chapter 2 presented an explanatory study of the global science of translocation biology. The multi-disciplinary aspects of undertaking translocations for threatened species conservation were presented in detail to introduce the ecological, social, cultural and economic concepts examined throughout the thesis. Next, in order to provide a contextual basis for considering the reintroduction of mala to Uluṟu – Kata Tjuṯa National Park, Chapter 3 presented an investigation and assessment of 109 Australian macropod translocations undertaken between 1969 and 2006. Data pertaining to a suite of translocation variables were presented and analysed, followed by an examination of the success or failure of the programs. This analysis clearly showed that the primary threat to Australian macropod translocation success was predation by introduced species, namely foxes and cats. Further, the control of introduced predators appeared to be significantly more important than the adoption of a particular set of translocation methodologies. If effective introduced predator control was not achieved, macropod translocations failed regardless of the release methodology employed.

Chapter 4 presented a study of population demographics in order to assess the short-term success of the Uluṟu mala reintroduction program. Following a description of

the translocation of animals from Watarrka National Park to Uluru, population estimates were presented, and a number of demographic variables investigated. The impact of rainfall at the study site on population growth and structure was also examined. In addition, a comparison with the Watarrka National Park mala population demographics was made. Both the Uluru and Watarrka mala populations experienced rapid population increase after founder release. It appears that founder group size, at least with a minimum of 24 individuals, is not critical to the initial successful establishment of translocated mala populations in predator free areas. A significant relationship was found between the presence of Uluru mala pouch young and rainfall in the three months prior to the surveys. This suggested that the supplementary feed provided did not fully compensate for times of lower rainfall (and thus 'natural' food availability) but rather that breeding and births in mala are triggered by rain events and the consequent condition of wild food plants.

Chapter 5 presented a radio-tracking study of mala ranging behaviour at Uluru, including an investigation of the suitability of using home range size as an indicator of enclosure carrying capacity. Further, the importance of supplementary feed stations was examined. This chapter also compared home range size estimates at Uluru with those calculated for mala reintroduced to the wild to assess the perceived benefits of housing threatened species in large, semi-natural enclosures. The study identified no significant difference in Uluru mala home range size at low and higher mala density. Further, home range estimates suggested that a significant percentage of mala living within the Uluru enclosure found adequate nutrition from natural sources as opposed to being reliant on supplementary feed. Analysis of radio tracking data showed free-ranging mala at Sangsters Bore had significantly larger home ranges than those of Uluru animals. Data suggested that this difference was likely due to a relative scarcity of resources at Sangsters Bore. Comparisons between home range estimates at Uluru and Peron Peninsula, where resource availability were more comparable, suggested that mala are not negatively affected by the spatial restriction of the Uluru predator-proof enclosure.

The composition of mala diet within the Uluru enclosure was examined in Chapter 6. This chapter also investigated the effect of rainfall on both the plant species, and plant part, consumed. In addition, comparisons were made between the diet of mala at

Uluru and that of the last wild population of mala studied prior to its extinction in the Tanami Desert. Further, the diet of free-ranging mala reintroduced to the Tanami Desert was also compared to the results of the Uluru study. The study revealed that mala consumed a large number of native plant species at Uluru, with the majority of dietary intake comprised of grasses and supplementary food. No correlation was found between rainfall and the amount of particular foods consumed. Supplementary food may, therefore, have provided a constant source of nutrition regardless of environmental flux over the study period, and mala availed themselves of this food to a similar level at all times. The composition of mala diet in both the Tanami Desert and at Uluru was very similar in both breadth and importance of specific plant types and parts. The study showed that Uluru mala appeared to have retained the ability to recognise and obtain natural foods.

Chapter 7 presented a study of mala behaviour within the Uluru enclosure. Individual behaviours were identified, and the average time allocated to each activity determined. Results were then compared to a previous study of behaviour observed in the species when held in a small, intensively managed captive breeding facility (Arid Zone Research Institute [AZRI], Alice Springs). Of the suite of behaviours identified by the study, Uluru mala spent significantly more time in the 'alert' standing position than any other behaviour. This was presumably due to the constant need for solitary animals to be aware of predators (perceived or otherwise), competitors and conspecifics. Both the Uluru mala and those held at AZRI displayed a similar suite of behaviours, although significant differences were apparent in the time spent undertaking each behavioural type. Although reasonable explanations for these differences were offered, the possibility that these discrepancies may indicate that captive bred mala may display behaviour unsuitable to the wild should be acknowledged.

Cultural aspects of the reintroduction of mala to Uluru were presented in Chapter 8. Traditional Ecological Knowledge (TEK) of the species held by senior Anangu was documented, and attitudes towards both the return of mala to Uluru and threatened species recovery work in general were explored. The study identified that senior people within the study area held detailed traditional knowledge of the biology and ecology of mala. However, it was also apparent that the level of mala TEK held by

Anangu was directly related to age. A significant decrease in knowledge of locally extinct animals was observed between older Anangu (early to mid-70s in age in 2009) and those only around 10 years younger. Anangu were enthusiastic about threatened species recovery work in general, and desires for the reintroduced mala at Uluru focused on both opportunities for younger generations to learn about the animals, and for the welfare of the animals themselves.

The following recommendations for conservation management of mala in arid / semi-arid mainland Australia are derived from the findings of the abovementioned chapters. Some of the recommendations presented are also applicable to other Australian macropods species.

9.2 Management recommendations

9.2.1 Intensive captive breeding facility or predator-proof enclosure?

Mala have been successfully bred in both intensive captive breeding facilities and relatively large, predator-proof enclosures. However, for both management and ecological reasons, I recommend that future populations of mala be established within enclosures. First, the cost of establishing predator-proof enclosures is likely to be comparable with, or favourable to, that of an intensive breeding facility, particularly if the latter requires built infrastructure (for example an appropriate building for food preparation or veterinary checks). Further, unlike intensive breeding facilities, ongoing maintenance and management of predator-proof enclosures is minimal. Second, the requirement to pair suitable breeding individuals within intensive breeding facilities, coupled with their relatively small spatial capacity, does not permit the rapid population growth and maintenance offered by predator-proof enclosures. Third, within predator-proof enclosures mala are free to exhibit and maintain (within the spatial limitations of the enclosure) 'natural' behaviours such as recognising and obtaining natural foods, establishing home ranges, and interacting with both conspecifics (including the selection of breeding partners) and other taxa. The management benefits described above, and the potential for behavioural changes to emerge within populations held within small, captive breeding facilities over the long term (McPhee 2003, Waples and Stagoll 1997), lead me to recommend the housing of future mala populations within predator-proof enclosures.

9.2.2 Selection of release site

Analysis of historic Australian macropod translocations clearly showed that the critical threat to program success was predation by introduced carnivores (Chapter 3). Whilst many different combinations of release methodologies were attempted, these were of little consequence if exotic predators were present. Both the local extinction and reintroduction history of mala provides a clear example of this, where even low numbers of introduced predators were sufficient to extinguish populations (Gibson et al. 1994, Langford 1999, Langford 2000; further, after the completion of this thesis, a subsequent wild release of mala in the semi-arid zone also failed). There is, therefore, no safe level of introduced predator abundance for mala reintroduction sites. Regardless of the perceived biotic, abiotic and managerial suitability of a release site, mala should not be released if introduced predators are present. Although broad scale baiting can eradicate foxes from a defined area (Saunders and McLeod 2007), there is currently no effective way of controlling feral cats in the arid/semi-arid zone (Denny and Dickman 2010). Therefore I recommend that mala are not released to the wild in these areas of continental Australia until such a cat control technique is developed. Consequently, releases to predator proof enclosures are currently the only management option for mala reintroduction to the majority of the species' former range.

This study has shown that although mala consume a broad range of food plants, grasses make up the bulk of their diet (Chapter 6). In particular, *Eragrostis eriopoda* and *Aristida contorta* were preferred food plants. When considering reintroducing mala to the central part of their former range, program managers should therefore select a floristically biodiverse sandplain / dunefield area with a good coverage of *Eragrostis* and *Aristida* species. As *Triodia sp*. dominate these sandplain / dunefield regions within the former range of mala, the presence of this genus will provide nesting sites and drought fodder (Johnson cited in Pearson 1983:92, Lundie-Jenkins et al. 1993).

My study has shown that senior Indigenous people living within the former range of mala hold detailed knowledge of the species (Chapter 8). This includes the former distribution of the animal, and areas of particular historical abundance. When selecting a release site for mala within the central part of their former range, managers

should consult, when possible, with senior Anangu as to the most appropriate location for the reintroduction. I also recommend that communities, land management authorities and cultural heritage agencies make every effort to record Traditional Ecological Knowledge (TEK) of mala and other species. This work should be a high priority as Anangu holding such knowledge are now in their senior years.

As the majority of the former home range of mala lies within sparsely populated, remote areas, consideration must be given to the practicalities of ongoing program management when selecting an appropriate release site. The history of mala recovery has shown the logistical difficulties encountered when translocations are made to remote locations (Pavey pers. comm.). In addition to requiring access for monitoring population size and health, the integrity of the enclosure itself will need ongoing vigilance. The Uluru mala program provides a clear example of this, as feral camels have collided with the enclosure fence on several occasions. Although subsequent damage has not resulted in breaches of the enclosure, the potential for significant damage by camels remains. Daily monitoring by UKTNP staff permits rapid response to fence damage and removal of camels found within the vicinity of the enclosure. Although efforts are underway to reduce feral camel populations within the arid / semi-arid zone, the potential for camel damage to predator-proof enclosures will continue to be a very real threat into the future. I therefore recommend that enclosures are built within close proximity to management agencies, or that funding be sufficient to build residential and management infrastructure at the release site.

9.2.3 Founder group composition

The Uluru mala reintroduction has shown that program success, at least in the short term, can be achieved with a single release of a relatively small founder group (24 individuals; Chapter 4). Further, a male bias in the release cohort does not threaten the establishment of the population. These findings should assist future mala reintroduction managers when they consider release group size and gender ratio. However, the potential for long-term genetic issues resulting from small founder size require consideration. The national Mala Recovery Team should work towards identifying the most appropriate way to manage the disjunct mainland mala populations to maximise genetic health, mainly through the exchange of animals.

9.2.4 Provision of supplementary food and water

The Uluru reintroduction has shown the readiness of mala to take advantage of food and water supplements (Chapter 6). The percentage of supplementary food found within Uluru faecal samples suggest that they are a significant component of mala diet. Although specific experiments would be required to identify a connection between accelerated population increase and the provision of supplements, I recommend that supplementary food and water be provided in future mala releases to predator-proof enclosures. In addition to their potential for improving animal condition and breeding rates, supplementary feed stations provide a focal point for population monitoring. A concentrated food source such as that provided by feed stations is an ideal location for conducting post-release trapping.

9.2.5 Post-release monitoring

Post-release monitoring has been a critical part of the Uluru mala reintroduction, and I suggest that all future translocations of mala include clear monitoring protocols. Initial trapping of the translocated population should occur within six months of release. Although the population would still be relatively small at that stage, trapping around supplementary feed stations should enable animals to be caught permitting the assessment of individual health and breeding condition. Subsequent trapping periods should be conducted every six months until managers are confident that the population has established (three years in the Uluru example). Although somewhat arbitrary, this decision will be based on variables such as individual health, proportion of females carrying pouch young, survivorship and population growth. Trapping data from the Uluru mala population identified differences between mark-recapture calculations and the number of mala known to be alive (Chapter 4). The number of animals KTBA may therefore be an important qualifier of mark-recapture results, and should be incorporated into the analysis of trapping results.

Post-release monitoring at Uluru included a radio-tracking study, undertaken in part to identify if mala home range size at different population densities could provide an indication of enclosure carrying capacity (Chapter 5). Although this study provided other interesting and useful data for mala management, the calculation of home range size through radio tracking proved very labour intensive. I therefore recommend that rather than using radio-tracking, the proximity of the Uluru mala population to

enclosure carrying capacity be assessed as follows. First, the results of trapping work should be used. A significant increase in the number and severity of fighting wounds, loss of condition, decrease in the proportion of females with pouch young, poor juvenile survivorship, and/or an increase in ecto/endo parasite load may provide evidence of overcrowding. Second, grasses of the genus *Eriopoda* and *Aristida* should be monitored within the enclosure. Signs of over grazing of these species may be an indication that the population has reached carrying capacity.

9.2.6 Ongoing management of release site

As fire is both a destructive and regenerative agent within arid/semi-arid grasslands, its management will be crucial to successful mala reintroduction programs. Senior Anangu who were familiar with mala prior to their local extinction have been central to the fire management of the Uluru enclosure. Annual burning programs aim to promote favoured mala food species by ensuring approximately 50 percent of spinifex within the enclosure is in an immature, regenerating state after fire. Whilst in this growth phase, important mala food plants such as *Eragrostis* and *Aristida* (Chapter 6) are common (assuming reasonable rainfall). The remaining 50 percent of spinifex communities are allowed to grow to provide suitable nesting sites for mala. When an area reaches maturity, characterised by a reduction in clump size and the formation of spinifex 'rings', it provides little cover for mala and preferred food plants are scarce. These areas are then marked for burning. I recommend a similar fire regime with respect to habitat creation is adopted in future mala translocations. In addition, the prevention of bushfire within predator-proof enclosures must be a high priority in order to avoid catastrophic loss of animals and habitat. Fire breaks should be maintained at a depth of 50-100m around the enclosure, and external/internal perimeter vehicle tracks kept clear of vegetation. Patches burnt to create suitable habitat for mala within the enclosure should be linked by a series of linear burns of minimum 30 metres width. These should be installed to break up continuous fuel, thereby ensuring fire cannot spread beyond a relatively small area within the enclosure.

9.2.7 Anangu involvement in threatened species reintroduction programs

As mentioned above, senior Anangu living within the central part of the former distribution of mala hold detailed TEK of the species (Chapter 8). There is no doubt that similar levels of knowledge would be held by Indigenous people in other areas formerly inhabited by mala. Present and future reintroductions of threatened taxa provide a unique opportunity for both Indigenous and non-indigenous people to work together for the conservation of species. Indigenous people can provide both ecological expertise and labour to reintroduction programs, and such projects provide important job opportunities within remote areas. Further, reintroduction programs have potential social benefits including the passing of traditional ecological and cultural knowledge from senior Anangu to younger generations. For the benefit of non-indigenous and Indigenous people alike, and the program as a whole, I strongly encourage proponents of future arid/semi-arid zone reintroductions to make working closely with local Indigenous people a central part of the project.

9.3 References

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