

INTRODUCTION

1.1 History and status of the Yellow-footed Rock-wallaby

The Yellow-footed Rock-wallaby, *Petrogale xanthopus*, is a species threatened with extinction (IUCN 2000). Two sub-species are currently recognized, *P. x. xanthopus* occurring in South Australia (Fig. 1.1) and New South Wales (Fig. 1.2), and *P. x. celeris* occurring in Queensland (Fig. 1.3). *P. x. xanthopus* is currently listed as Vulnerable C2a as a result of a declining and fragmented total population numbering less than 10,000 individuals (IUCN 2000). *P. x. celeris* is listed as Lower Risk (near threatened) due to an inferred population of less than 10,000 individuals (IUCN 2000). The two sub-species are reported to have diverged 180 000 years ago (Eldridge 1997c).

Edward John Eyre is reportedly the first European to sight *P. xanthopus*. A diary entry made on August 12, 1840 described him seeing large numbers of rock-wallabies at Mt Aroona (Fig. 1.1), the site of the current South Australian re-introduction. His notes included the remark 'they leapt and clambered up among the steep sides of the cliffs in a manner quite incredible' (Eyre 1841). Further early reports from around the 1850's suggested that vermin-proof fencing would be required to protect vegetable gardens from wallabies at Haywood's cottage on the Aroona sheep run (Bruce 1902). Despite their common status, the species was not classified until some years later when two specimens were collected from the Flinders Ranges in South Australia by Frederick Strange. Upon returning to England, Strange sent a specimen to the Director of the British Museum of Natural History, J. E. Gray. In 1854 Gray described the species belonging to the genus *Petrogale* (*petra*: rock; *gale*: weasel) he had erected in 1837 and named it *xanthopus* (*xanthos*: yellow; *pous*: foot), thus the Yellow-footed Rock-wallaby.

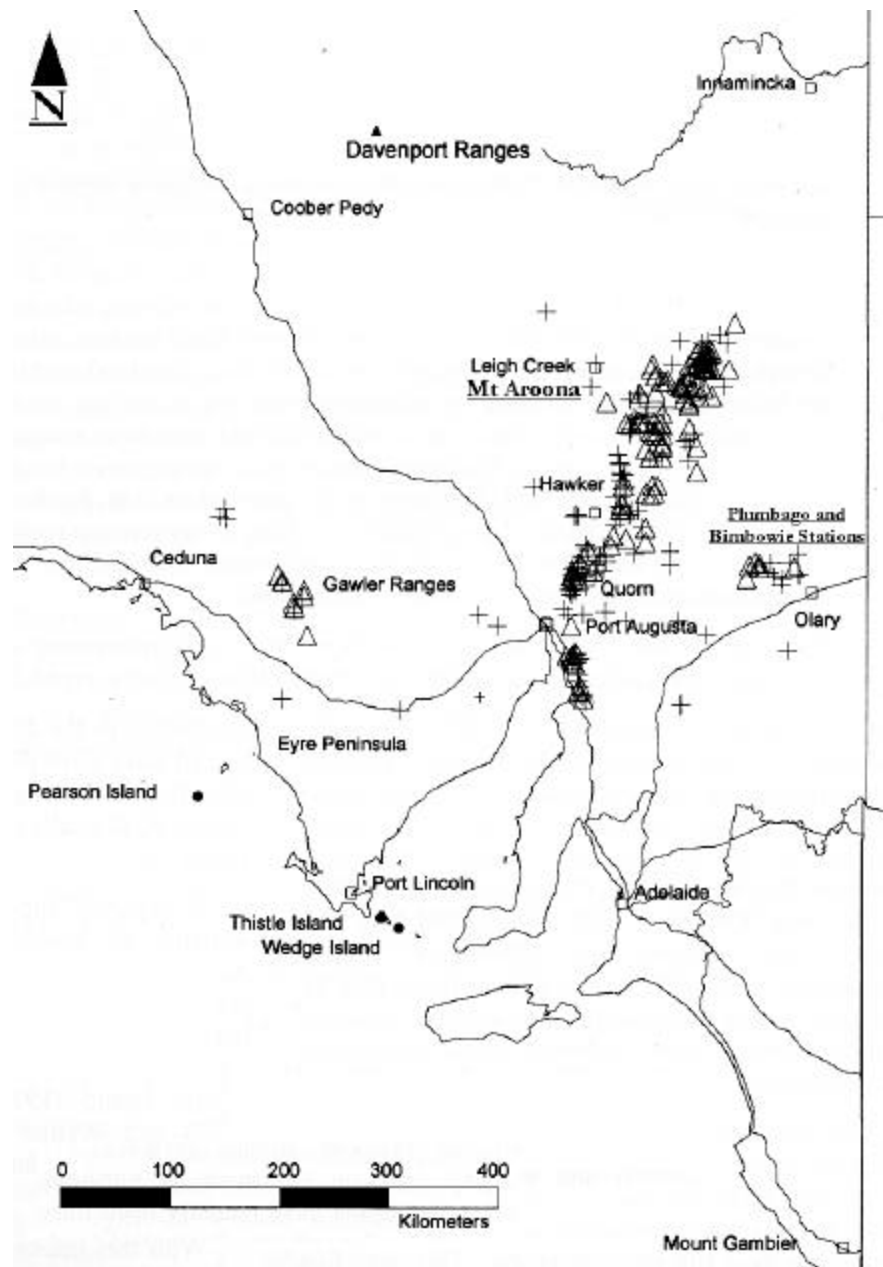


Figure 1.1 Past and present distribution of *Petrogale xanthopus xanthopus* in South Australia. + *P. x. xanthopus* recorded pre 1980, ▲ *P. x. xanthopus* recorded post 1980. Map modified from Copley and Alexander (1997).

Although only known to science for c.150 years, *P. xanthopus* or 'Andu' has long been entrenched in aboriginal culture and the traditions of the Adnyamathanha or Hills People who occupied the Flinders Ranges. Andu were hunted both for meat and byproducts such as tail sinew used to make ropes and pelts for clothing, and although Aboriginal law did not specify who could kill Andu, only initiated men were allowed to butcher the meat

and divide it among the people (Tunbridge 1991). The Adnyamathanha continued to hunt Andu after the species became protected in 1912. The practice however declined throughout the mid 1900's as mutton and other food substitutes became more readily available, and initiated men allowed to butcher the meat died (Tunbridge 1991). Nevertheless, the numbers taken for food were likely negligible compared to the wholesale slaughter of the species by Europeans earlier in the century.

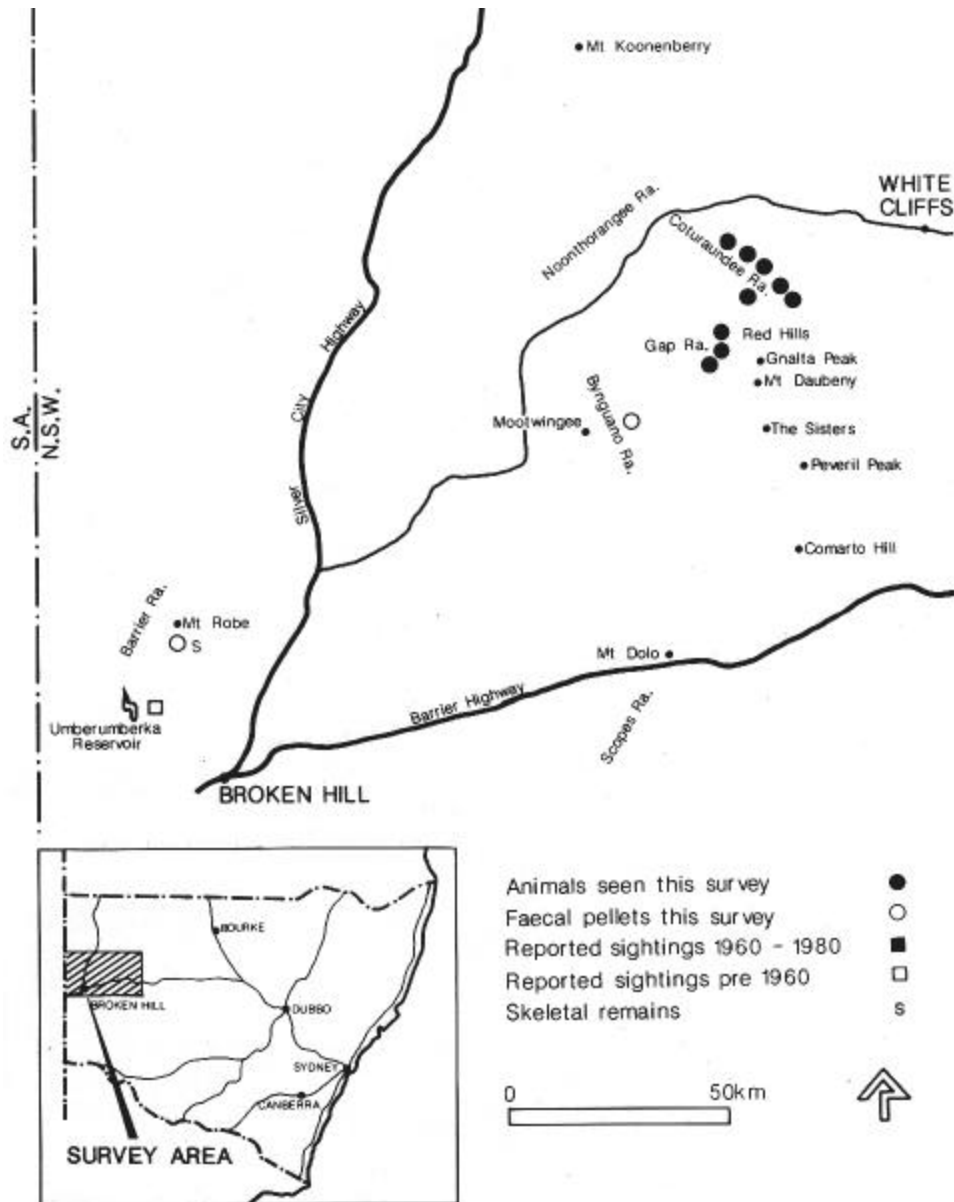


Figure 1.2 Past and present distribution of *Petrogale xanthopus xanthopus* in New South Wales. Map reproduced from Lim *et al.* (1987).

'The best sport we had was in firing at the rock-wallaby from the seat of the buggy, and watching them fall down the cliffs. Fourteen we saw fall, some were wounded, but before reaching the bottom were pretty well dead...We were indeed loathed to leave so pretty a spot. We gazed and feasted our eyes on nature's handiwork.' (The Areas Express and Farmer's Journal, 1 December 1883, as reported in Lim *et al.* 1987).

The above extract was published around the time the naturalist Thomas Ward visited the Flinders Ranges and noted 'the rock-wallaby is by far the most abundant of animals, and yet it is a much persecuted creature' (Fountain 1907 from Lim *et al.* 1987). The persecution of *P. xanthopus* intensified around the turn of the century. Aboriginal hunters were used and scalp bonus paid to eradicate the species from pastoral properties, with up to 300 wallabies being taken by a group of hunters in a single day (Lim *et al.* 1987). Hundreds of *P. xanthopus* skins were exported from Adelaide to London annually for the fur trade. In 1912 *P. xanthopus* became protected in South Australia through the Animals Protection Act, but the slaughter continued with skins being sold interstate. In 1924 Professor Wood Jones issued the warning:

'*Petrogale xanthopus* is a fitting example of an animal which needs sanctuary for its preservation and more stringent legislation efforts to check its slaughter'.

The first large-scale census of *P. x. xanthopus* population size was undertaken by Lim (1987) in 1981. *P. x. xanthopus* had only recently been discovered in New South Wales (Fox 1966) and the distribution of the species in the state was uncertain. Helicopter surveys throughout the species' range in both states estimated 6,900-8,300 animals to occur in South Australia and 180-230 in New South Wales. The species was thus classified as Endangered in New South Wales. Recent surveys in South Australia have not been as intensive and no further population estimates have been published. However Copley and Alexander (1997) reported that although the overall range of the species in South Australia has declined little in the last two decades, at least 35 out of 229 known colonies (or 15%) have become extinct since European settlement, eight (23%) of the 35 in the last 30 years (Fig. 1.1). Despite the large colonies that once existed (Lim *et al.* 1987), only eight (4%) have been recorded with more than 20 animals since surveys commenced. The removal of exotic vertebrates around some *P. x. xanthopus* colonies on Plumbago and Bimbowrie Stations in the Olary Range (Fig. 1.1) has resulted in local recent increases (Fig. 1.5, 1.6) (Copley and Alexander 1997). *P. x. xanthopus* has suffered a more significant decline in New South Wales (Fig. 1.2, 1.4). Animal sightings in that state have declined by over 60% since surveys began (Dovey *et al.* 1997; Sharp and Norton

200). Less than 5,000 *P. x. xanthopus* are now thought to exist in both states (M. Lethbridge pers. comm.; Alexander and Copley 1999).

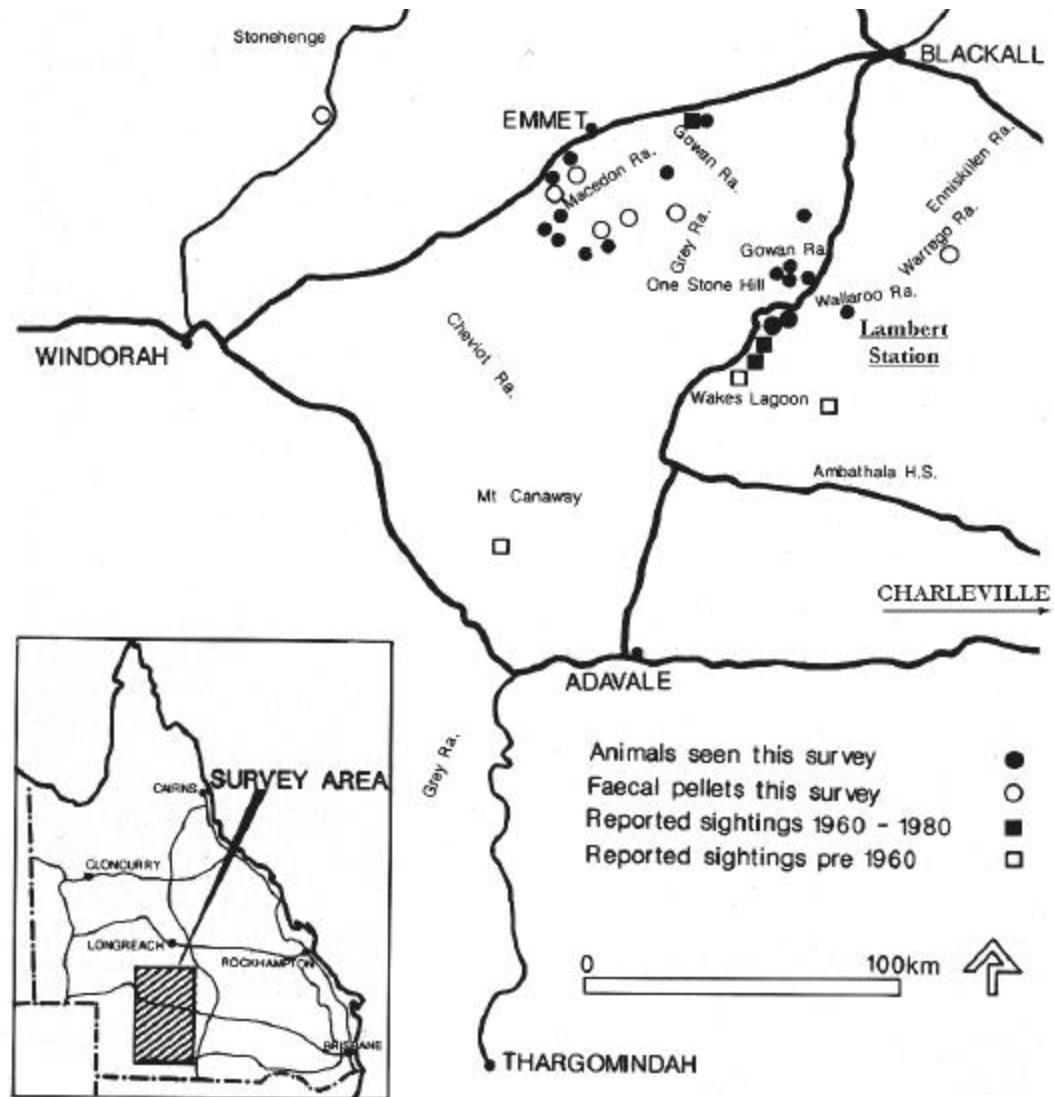


Figure 1.3 Past and present distribution of *Petrogale xanthopus celeris* in Queensland. Map reproduced from Lim *et al.* (1987).

Petrogale xanthopus celeris was first collected in Queensland in 1922 (Gordon *et al.* 1993) and described as *Petrogale celeris* (Le Souef 1924), but later regarded as a subspecies of *Petrogale xanthopus* in 1934 when reviewed by Iredale and Troughton of the Australia Museum, Sydney (Eldridge 1997a, c). Surveys for the species were undertaken in 1973-74 by

Gordon *et al.* (1978), with reliable earlier reported sightings being investigated, many unsuccessfully. Gordon *et al.* (1978), Lim *et al.* (1987) and Copley (1990) reported that the species' range was previously more extensive, with at least six known colony extinctions (Fig. 1.4), and that it was a species in decline. A more detailed survey was undertaken between 1980 and 1987, with an intensive survey in June 1984 (Gordon *et al.* 1993). The species was found at 44 sites over a 250 km² area and although not calculated with accuracy, the population was estimated to be 5,000 - 10,000 animals (Gordon *et al.* 1993). No surveys have since been undertaken. While large *P. x. celeris* colonies are known to exist, the current status of the species in Queensland is best described as unknown. The total population of *P. x. celeris* is unlikely to be over 10,000 (Lim *et al.* 1987; Gordon *et al.* 1993) and thus the sub-species should be classified as Threatened under the IUCN (2000) guidelines of C2a; a declining (Gordon *et al.* 1978; Lim *et al.* 1987; Copley 1990; Sharp 1997a) and fragmented population of less than 10,000 adults.

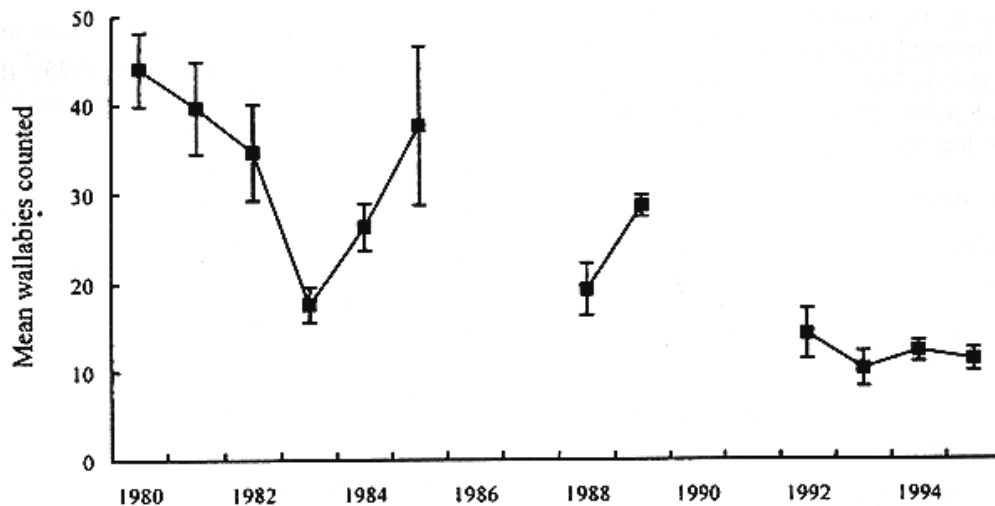


Figure 1.4 Mean number of *Petrogale xanthopus xanthopus* counted by front observer during helicopter counts of the Gap and Cotouraundee Ranges, western New South Wales, between 1980 and 1995 (95% confidence limits shown).

Figure reproduced from Sharp and Norton (2000).

1.2 Current threats to the species

Hunting of *P. x. xanthopus* is thought to have ceased in the mid 1900's, however colonies have continued to become extinct since this time (Copley and Alexander 1997). The local extinction of some colonies in the Flinders Ranges was reported to coincide with the arrival of *V. vulpes* (Lim *et al.* 1987). Lim *et al.* (1987) suggested *V. vulpes*, and possibly natural predators such as Wedge-tailed Eagles (*Aquila audax*), may be responsible for the continuing rarity of *P. x. xanthopus*. Both species are known to be abundant throughout the range of *P. xanthopus* (Lim 1987; Lim *et al.* 1987; Copley 1990; Parker 2000; Lapidge 2001). Competition for food and habitat by feral goats (*Capra hircus*) has also been linked with the decline of *P. xanthopus*, as the two species fill a similar ecological niche (Dawson and Ellis 1979; Lim 1987; Lim *et al.* 1987; Copley 1990; Maxwell *et al.* 1996; Allen 2001). Competition for food plants is reported to be most intense between the two species during drought, and may suppress the recovery of *P. xanthopus* populations post-drought (Lim 1987; Lim *et al.* 1987; Copley 1990; Allen 2001).

Both *V. vulpes* and feral cats, *Felis catus*, are known to prey on rock-wallabies (Kinnear *et al.* 1984, 1988, 1998; Spencer 1991; Hornsby 1997; Lapidge 2001, Appendix D). A study of *V. vulpes* predation on Black-footed Rock-wallaby (*P. lateralis lateralis*) colonies in the Western Australian wheatbelt was instigated in 1982. The area surrounding two of five *P. l. lateralis* colonies was controlled for *V. vulpes* with population increases of 138% and 223% occurring within six years, while two of the three other colonies declined by 14%, 85% and one increased by only 29% (Kinnear *et al.* 1988). A recent update on the study has reported population increases at the *V. vulpes* controlled colonies of 500% and 644%, and decreases of 59% and 100% (colony extinct) and one had no change (Kinnear *et al.* 1998). Despite these results, some ecologists doubt the ability of *V. vulpes* to predate *P. xanthopus* due to their larger adult size than *P. l. lateralis* (Lim *et al.* 1987). Even if adults were able to avoid predation by *V. vulpes*, it is unlikely that juveniles can.

Numerous *P. x. xanthopus* colonies in South Australia have undergone control of *V. vulpes*, *C. hircus* or both in the last decade as part of Operation Bounceback, a South Australian Department of Environment and Heritage initiative (Copley and Alexander 1997; Alexander and Copley 1999). Dramatic local increases in *P. x. xanthopus* populations have occurred (Fig 1.5, 1.6) when *V. vulpes* or both species have been

reduced on Plumbago and Bimbowie Stations in the Olary Ranges (Fig. 1.1). A similar finding has recently been reported for *P. x. xanthopus* in New South Wales (Torr and Beaton 2000). Along with further recent evidence (Hornsby 1997; Lapidge 2001, Appendix I), *V. vulpes* is indicated as the most significant current threat to the survival of *P. xanthopus*.

Figure 1.5 Estimated *P. x. xanthopus* population increase in relation to *C. hircus* and *V. vulpes* control on Plumbago Station, Olary Ranges, South Australia
(Reproduced from Alexander and Copley 1999).

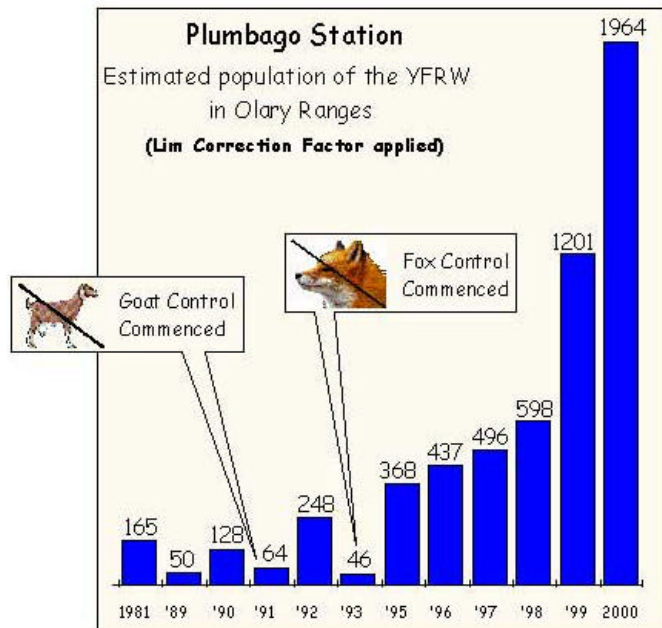
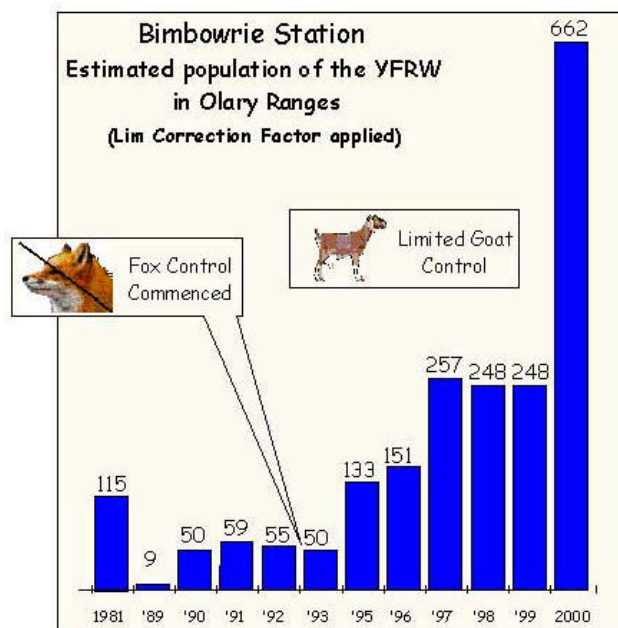


Figure 1.6 Estimated *P. x. xanthopus* population increase in relation to *V. vulpes* control on Bimbowie Station, Olary Ranges, South Australia
(Reproduced from Alexander and Copley 1999).



1.3 Reintroduction as a science

Reintroduction is defined as:

“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” (IUCN 1998),

and is generally undertaken for one of three reasons (Short *et al.* 1992):

1. **aesthetic**- restoring ecosystems to their pre-European state;
2. **conservation**- undertaken with threatened species to decrease the risk of extinction by establishing more populations in different areas; and
3. **experimental/scientific**- undertaken to study the complex interactions between a species and its habitat.

Translocation is defined as:

“deliberate and mediated movement of wild individuals to an existing population of conspecifics” (IUCN 1998),

and generally occurs to increase the size of a wild game population for hunting purposes (Griffith *et al.* 1989).

Conservation reintroductions are traditionally reserved for endangered species, many with few remaining individuals. Kleiman *et al.* (1994) suggested that a reintroduction is not warranted if there is no need to supplement the number of individuals, populations or genetics of a species. Their reasoning was that more cost effective ways to recover the species may be ignored (Kleinman *et al.* 1994). However, reintroduction of more prevalent sympatric species may have scientific merit, particularly if little is known about the endangered species' ecology or effective ways of returning them to the wild.

Reintroduction may also be to the detriment of a species that has declined to a small population. A recent example is that of the Rufous Hare-wallaby (*Lagorchestes hirsutus*), where the death of all animals released accounted for a considerable proportion of the total remaining population (Gibson *et al.* 1994; Langford and Burbidge 2001). Griffith *et al.* (1989) and Wolf *et al.* (1996) reported a reintroduction success rate of 44-53% for threatened, endangered or sensitive species versus 81-86% for native game species respectively, indicating that greater reintroduction success occurs with more secure species. Hence, reintroduction procedures are best tested on more common species until a reasonable level of success is achieved.

There has been a general call for the use of re-introduction in scientifically structured ecological experiments (Krebs 1978; McNab 1983; Griffith *et al.* 1989; Armstrong *et al.* 1994b; Serena and Williams 1994; Soderquist 1994; Soderquist and Serena 1994; Sarrazin and Barbault 1996; Burgman and Lindenmayer 1998; Seddon 1999; Fischer and Lindenmayer 2000). It has been suggested that an ideal candidate species for reintroduction is one that is known to have declined in the wild, but is not yet critically endangered, so that information gained from the reintroduction may be used to reverse the decline of that species and assist more endangered species (Serena and Williams 1994). Furthermore, the species should be well studied in the wild and have a suitably large self-sustaining captive-population. *Petrogale xanthopus* is thus the most appropriate *Petrogale* species for reintroduction; although it is vulnerable (IUCN 2000), it is less endangered than the Prosperine Rock-wallaby (*P. persephone*), Brush-tailed Rock-wallaby (*P. penicillata*) or Black-footed Rock-wallaby (*P. lateralis*) (Maxwell *et al.* 1996; IUCN 2000), it is well studied in the wild (Copley 1983; Lim 1987; Lim *et al.* 1987; Copley 1990; Lim *et al.* 1992; Sharp 1994; Sharp 1997a), and there is a large, self-sustaining captive population (c.170 animals in 15 institutions at the commencement of the current study; Slater 2001).

Despite over two decades of calls for reintroduction to be used in ecological experiments, relatively little has been published on how re-introduced animals adapt to their new environment upon release. This is possibly a result of the limited success of the procedure or because ecological monitoring normally requires extensive resources, such as staff, finance and equipment, that are often not budgeted for in reintroduction projects (Short *et al.* 1992). Thus the ability of re-introduced animals, particularly those from captivity, to adapt to a release site is largely unknown.

1.4 Reintroduction in Australia

Australia accounts for one-third of the world's mammal extinctions (Maxwell *et al.* 1996), and the highest failure rate of re-introductions (Fischer and Lindenmayer 2000). In a recent review of re-introduction success, Fischer and Lindenmayer (2000) reported that 56% (14/25) of reintroductions undertaken for conservation purposes in Australia and New Zealand had failed and that the status of 28% (7/25) was unknown. Thus only 16% (4/25) were considered successful at the time of publication. The estimates of Morris (2000) in relation to Western Australian reintroductions are however more optimistic.

Since 1971, 59 reintroductions have occurred in the state, many through the *Western Shield Project*, of which 36% have been successful, 22% unsuccessful and 42% unknown at the time of publication. Interestingly, re-introduction has been recommended for one-third of Australian vertebrate taxa, in particular 63% of mammal species, considered in nationally endorsed Action Plans (Kennedy 1992; Serena and Williams 1994; Maxwell *et al.* 1996).

Re-introductions of macropods have suffered a similar fate to those of other species. Short *et al.* (1992) reported only one successful mainland reintroduction of a macropod out of 10 attempted. However, introductions of macropods to islands without exotic predators, both in Australia and overseas, have been much more successful (>82%) (Maynes 1989; Short *et al.* 1992; Veitch 1994). In a more recent review by Morris (2000), 33% of macropod reintroductions, predominately in Western Australia, were successful, 19% unsuccessful and 48% unknown at time of publication. Of successful releases, all involved intensive fox (*Vulpes vulpes*) control and none used captive-bred stock. Thus the absence or control of exotic predators, primarily *V. vulpes* and feral cats (*Felis catus*), is essential if reintroductions of macropods are to succeed. To date, a successful mainland re-introduction of a captive-bred macropod species has not occurred.

1.5 Research objectives

At the 1994 Rock-wallaby symposium, the Royal Zoological Society of South Australia proposed a trial reintroduction of captive-bred *P. x. xanthopus* to test reintroduction procedures that could be applied to more critically endangered *Petrogale* species. The proposal was reported to be “a useful adjunct to other conservation efforts for the species” that would provide a model for potential reintroductions of the more endangered *P. lateralis* in South Australia (Copley and Alexander 1997). The author became involved in the project in April 1996 as a feral animal control, flora survey, and post-release radio-tracking volunteer. The Royal Zoological Society of South Australia, NRG Flinders and the South Australian Department of Environment and Heritage undertook the re-introduction on September 26, 1996 in the northern Flinders Ranges of South Australia. Between July 1996 and February 1997 the author undertook a B.Sc. Honours research project (Flinders University of South Australia) monitoring post-release dietary adaptation of released animals (Lapidge 1997, 2001). In December 1997 a

trapping program was established by the author for this project to allow future monitoring of the population. The reintroduction of *P. x. celeris*, initiated by the author and I. Hume in November 1997, was designed to allow comparison to that of the earlier *P. x. xanthopus* re-introduction, while further investigating captive-bred animal adaptation to the wild through extensive pre- and post-release monitoring. The aims of the study were therefore to:

1. re-introduce captive-bred *P. x. celeris* in accordance with established and original reintroduction techniques and identify successful methods of reintroduction;
2. determine how captive-bred *P. x. celeris* adjust to the wild **ecologically**, specifically through studies on fecundity, survival, home range and dispersal, and **physiologically**, specifically through studies on growth, haematology, biochemistry, vitamin E, condition, water turnover rate, field metabolic rate and food intake rate, and **genetically** by directly comparing data gathered both pre- and post-release;
3. undertake comparable ecologically and physiologically monitoring of the previously re-introduced *P. x. xanthopus* to that of *P. x. celeris*; and
4. compare post-release results from the two re-introductions to pre-release data for each sub-species, to each other, to previous research on both sub-species in the wild, and to findings for other re-introduced animals to examine the applicability of re-introduction as a conservation technique for *P. xanthopus*.

This thesis differs from conventional format in that it is not hypothesis driven. Reintroduction success or failure is multi-faceted, combining scientific, social and economic factors. The isolation of specific causes and creation of testable hypotheses could be compromised by extenuating factors, resulting in biased findings. Consequently the thesis is methodology driven and arranged in a chronically order of events, although procedures and data from the later *P. x. celeris* re-introduction are detailed first in each chapter due to them being the primary source of data for the project.

This thesis can be divided into two sections: chapters 2-4 detail pre-release techniques and baseline data, and Chapters 5-8 detail post-release monitoring techniques and research findings. Site and founder selection and preparation criteria used in the re-introductions of both sub-species are included in Chapters 2 and 3. The resources required, in particular finances, to undertake the current project are detailed in Chapter 4

because of recent concerns (IUCN 1998; Fischer and Lindenmayer 2000) about the accountability and cost-effectiveness of reintroduction as a conservation technique.

The seasonal trends at both release sites and the environmental differences between them are described in Chapter 5. Chapter 6 describes the ecological adjustments to the wild of both sub-species, Chapter 7 details their physiological adjustments, and Chapter 8 details the water, energy and food requirements of reintroduced *P. x. celeris*. Chapter 9 was to detail genetic changes that occurred in the reintroduced *P. x. celeris* populations post-release. Unforeseeable problems encountered with outsourced genetic analysis have meant that the findings cannot be presented in the thesis, but will be published at a later date. Instead, Chapter 9 summarises the major findings of the study, their contributions to the field of reintroduction biology, and the appropriateness of the technique for reintroducing captive-bred animals to conserving a species in the wild.

SITE SELECTION AND PREPARATION

2.1 Introduction

A positive relationship has been demonstrated between success of intentional releases of animals and the quality of the release site. In reviews of factors associated with reintroduction and translocation success, Griffith *et al.* (1989) and Wolf *et al.* (1996) reported a difference of 84% (n=63) and 79% (n=52) respectively in the success of animal releases to habitat judged as of excellent quality, compared to 38% (n=32) and 45% (n=33) respectively to habitat judged as of fair or poor quality. Whether the release site is in the core or periphery or even outside the species' former range was also found to contribute to success. By definition, the core of a species' range relates to extant populations rather than the historical range. Releases into the core of the species' range have on average a 75% (n=130) chance of success, versus 46% (n=52) into the periphery (Griffith *et al.* 1989; Wolf *et al.* 1996). Releases into poor quality habitat have increased in the most recent survey (Wolf *et al.* 1996), despite the establishment of guidelines for intentional releases of animals (IUCN 1987; Kleiman 1989; Stanley Price 1991, Kleiman *et al.* 1994; Beck *et al.* 1994; IUCN 1998).

Determining and addressing the factors that caused a species' local extinction is paramount prior to any re-introduction attempt (Griffith *et al.* 1989; Kleiman 1989; Stanley Price 1989; Short *et al.* 1992; Kleiman *et al.* 1994; Lindenmayer 1994; Wolf *et al.* 1996; IUCN 1998). While the main cause of decline may seem obvious, such as predation with macropods (Short *et al.* 1992), less apparent factors may also be detrimental. When the reintroduction of Quokkas, *Setonix brachyurus*, failed in 1998, despite the release of c. 700 animals over a 16-year period, overgrazing by competitors and predation by exotic

predators was blamed, however the less noticeable problem of *Salmonella* infection was also cited as a possible factor (Short *et al.* 1992).

The amount of background research available on a species intended for reintroduction also varies greatly. The Przewalski Horse or Takhi, *Equus ferus przewalskii*, last lived in the Gobi Altai of Mongolia but became extinct in the wild during the 1960s before detailed research was conducted on the species' ecological range and requirements (FAO 1986; Van Dierendonck & Wallis De Vries 1996). Although not ideal habitat for *E. f. przewalskii*, the Gobi Altai is now believed to be the only site remote enough from human pressures and predation where species could survive (Van Dierendonck & Wallis De Vries 1996). Prior to their demise, 13 individuals were taken into captivity forming the genetic basis for a captive breeding program with the eventual aim of reintroduction. The current population of 1200 plus animals is testimony to the success of captive breeding. However, although sufficient *E. f. przewalskii* now exist for reintroduction, little is known about their ecological requirements and problems with habitat choice and insufficient ethological knowledge have occurred with reintroduction trials (Van Dierendonck & Wallis De Vries 1996).

Under IUCN guidelines, a reintroduction must occur into an area that once formed part of the intended species' historical range and from which they have become extirpated (IUCN, 1998). However, habitat modification through changes in land use, the establishment of native or exotic species, or stochastic processes may have changed the area dramatically and even irreversibly since the species' extinction. Attempts to reintroduce butterflies into an English fenland habitat failed due to vegetation changes caused by the drainage of the local area, becoming unfavourable for the larval stage of the butterflies and hence preventing recruitment (Duffey 1977; Dempster and Hall 1980). Conversely, the Red Wolf, *Canis rufus*, was introduced into coastal prairie marshes rather than its original bottomland riverine forests in North Carolina because its historic range had been cleared of the dense vegetation cover required for the species (Stanley Price 1989).

Published accounts of former reintroductions, although rare for the number attempted, have led to selection criteria being proposed for assessing appropriate reintroduction sites (e.g. FAO 1986; Griffith *et al.* 1989; Kleiman 1989; Stanley Price 1989; Short *et al.*

1992; Kleiman *et al.* 1994; Van Dierendonck & Wallis De Vries 1996; Wolf *et al.* 1996; IUCN 1998). Further ecological considerations have been proposed by Lindenmayer (1994). This chapter details previous and original site selection criteria and then addresses how they have been used to select and prepare reintroduction sites for *Petrogale xanthopus celeris* and *P. x. xanthopus*.

2.2 Site selection criteria

2.2.1 Surveys in historic range

Published selection criteria state that a need to augment a species' wild population must first be established before proceeding with reintroduction for conservation purposes (IUCN 1987; Kleiman 1989; Stanley Price 1989; Kleiman *et al.* 1994; IUCN 1998). Detailed background research on historical accounts of the species and their former and current range are required. Comparing the former and current range of the species and establishing a list of possible release sites is often the first step in determining whether a species will be suitable for reintroduction. If no sites not currently occupied by conspecifics can be identified, then reintroduction should not proceed. Once a list of potential release sites has been established, each site should be surveyed in detail, and feasibility studies, including modeling, should be undertaken using the criteria listed below. If any one criterion is not met, the site should be abandoned.

2.2.2 Site proximity

An ideal reintroduction site is one that is in close proximity to both the origin of the wild captive-breeding stock and the site of the breeding facility for captive-bred stock. Animals for reintroduction should be genetically similar to the previously extinct or nearest wild counterparts at the chosen release site to ensure similar genetic adaptations to the local conditions. This also reduces the possibility of outbreeding depression or genetic swamping should the reintroduced and wild populations meet. Known dispersal distances for a species should be used to ensure that a chosen release site is far enough away from extant colonies; a reintroduction should never jeopardize an extant population through social disruption or disease transmission (Kleiman 1989; Stanley Price 1989; Short *et al.* 1992; Kleiman *et al.* 1994; Frankham 1995; IUCN 1998). Breeding facilities

should ideally be as close as possible to the reintroduction site so that release animals are adapted to local climatic conditions, and transit time is minimized, so minimizing stress and the possibility of capture myopathy (IUCN 1998; S. Conaghty pers. comm.).

2.2.3 Habitat quality

Many facets are involved in the ultimate quality of a reintroduction site (Lindenmayer 1994). As in the case of *E. f. przewalskii*, surveys of extant populations can be misleading (FAO 1986; Van Dierendonck & Wallis De Vries 1996). Potential reintroduction sites should be surveyed to determine their location in reference to the core or periphery of wild populations, the suitability of the topography and availability of shelter and nesting sites, predator and competitor densities, food availability and suitability, climatic suitability, water sources (if required), level of disturbance prior to and since the species' demise, and results of any habitat restoration (IUCN 1998). In the case of water, food and climate, year-round information is required, either through ground surveys or local knowledge, in order to determine if release animals are likely to have sufficient resources to survive troughs in resource availability. The presence of poisonous plants should be noted, especially if they could become a primary food source during troughs in resource availability. Survey results should be compared with habitat of wild populations. Detailed information on the species' biology is thus required in selecting suitable habitat. If such knowledge cannot be acquired through published research, the author recommends further background research.

The relative importance of each facet of habitat quality will be dependent on the species in question. Habitat generalists or larger mammal or bird species with the ability to disperse widely require a high quality ecosystem, while smaller or habitat-specific species, such as rock-wallabies, require reintroduction sites to be high quality local habitat. However, an ecosystem approach should also be taken for habitat-specific species.

Lim and Giles (1987) derived a habitat scoring system for *P. x. xanthopus* in New South Wales. Ranging between 0 and 13, one point was awarded for the presence of steep cliffs, outcrops, gullies, terraces, caves, rock-piles, and water (within 2 km of the nearest rock-pile). The presence of boulders and the rock surface texture were further classified and awarded 1 point for small boulders (<25 cm), 2 points for boulders between 25 and 100

cm, and 3 points for boulders greater than 1 m in diameter, while rough rock surface texture was awarded 1 point, intermediate 2 points, and smooth 3 points (Lim and Giles 1987). Aerial and ground surveys were then conducted throughout the possible range of *P. x. xanthopus*, there was a positive relationship between habitat score and the presence of the species. Extant colonies were found only in areas with scores of 12 or 13; sites with scores of 10 or 11 showed occasional use. Habitats with scores of nine or lower showed little sign of occupation by *P. x. xanthopus*. Although a useful local guide, the habitat scoring system does not take into account the great variety of habitats used by the species in Queensland, New South Wales and South Australia. For example, many *P. x. celeris* colonies occur on the dissected plateaux of the Adavale Basin, where Tertiary rocks have been eroded to expose softer, less-weathered Cretaceous sediments and rough-textured conglomerate-tillite rocks (Nedler 1984). This contrasts with the predominantly smooth sandstone in NSW and the Flinders Ranges, SA, where Lim (1987) undertook the majority of his research. Hence, the natural habitat of *P. x. celeris* would score less despite its relatively large populations.

2.2.4 Habitat size

The size of a potential reintroduction site should be sufficient to provide the resources needed for a species to exhibit its natural home range, dispersal and demography without environmental damage (Lindenmayer 1994). The introduction of Tammar wallabies, *Macropus eugenii*, to Greenly and Granite Islands in South Australia and Western Grey kangaroos, *M. fuliginosus*, to Woody Island, Western Australia all resulted in overgrazing, resulting in a reduction in the carrying capacity of the sites (Robinson 1980; Short *et al.* 1992). The size of the release site should also be large enough to prevent overlap with extant colonies. The establishment of several pooled populations or a meta-population is thought to be preferable to a single large population for both genetic and ecological reasons, specifically to decrease the chance that stochastic events such as flood or fire may eliminate all animals at once (Frankham 1995; Forbes & Boyd 1997; Margan *et al.* 1998). Larger areas of suitable habitat will normally be required for meta-populations to be formed.

Analysis of the home ranges of *P. xanthopus* has shown remarkably different results between the two subspecies. Lim (1987) reported a summer home range size of 211 ha

for males and 164 ha for females, compared to winter ranges of 169 ha for male and 134 ha for female *P. x. xanthopus* at Middle Gorge, South Australia. The collective home range, or total area used by the Middle Gorge *P. x. xanthopus* colony of 140 animals was estimated to be 1,000 ha (Lim 1987). This compares to an average home range of 20 to 40 ha for *P. x. celeris* at Idalia National Park, Queensland (Sharp 1994). Sharp (1994) suggested that the difference reflected the more marginal quality of the *P. x. xanthopus* habitat.

Despite extensive effort, dispersal ability of *P. xanthopus* remains largely unknown. In South Australia, Lim (1987) recorded inter-colony exploratory movements of 1.5 km between Middle and Buckaringa Gorges, however no animal from the Middle Gorge colony was found to enter Buckaringa Gorge due to it containing a separate colony. No Buckaringa Gorge *P. x. xanthopus* were captured for radio-collaring to determine if the reverse was occurring, or if there was any permanent movement between the colonies. In a study of dispersal patterns in *P. x. celeris*, Sharp (1997a) monitored 120 animals and found that only one juvenile male dispersed; a distance of 600 m. The results of the later study were consistent with a genetic study by Pope *et al.* (1996) showing significant differences in allele frequency at one microsatellite locus between colonies in close proximity (<10 km). *Petrogale xanthopus* has also been reported to travel from their daytime refuge up to 2 km to drink at Middle Gorge (Lim 1987) and 5 km in New South Wales (Lim 1987; Lim *et al.* 1992). A minimum exploratory movement of 1.2 km was also recorded by Sharp (1997a) for one male.

2.2.5 Habitat use and protection

The principal aim of any reintroduction, whether for conservation, scientific or aesthetic reasons, should be to establish a viable, self-sustaining population in an area of the species' former range. To ensure the long-term survival of a reintroduced population, land use, habitat protection, tenure and access must be assured (IUCN 1998). The release site and surrounding land should be used for a non-threatening purpose to the species involved. Shooting, mining, logging, camping or other human practices that may encroach on the release site should be avoided. Although camping itself may not be detrimental, factors such as illegal hunting, animal disturbance, habitat destruction,

pollution, possible disease transmission from humans or their pets, and the escape of exotic predators such as dogs or cats may.

Although national parks, sanctuaries or reserves make ideal release sites in terms of land use, protection and tenure, higher quality habitat with more limited human access may be available on private land. Greater control of exotic predators and competitors may also be practiced on private land, particularly if it is used for primary production.

A long-term commitment to habitat protection and access, whether formal in the form of a covenant or informal, should be achieved prior to any planned reintroduction, whether the land is private or publicly owned. Freehold ownership and a long historical commitment from the landowners is often preferable to leased or recently acquired land. However, the long-term intentions of private landowners should also be discussed, because if sold, new owners may not support previous agreements on land use, habitat protection or access.

To fulfill the requirements of post-release monitoring (IUCN 1998), there must be regular access to the release site. In my opinion, there are two forms of access that require attention. The first is legal access through the cooperation of the landowner. On private land, access can be revoked at any time, particularly if the practices of researchers are seen to be to the detriment of productivity or the traditions of the landowner; agreement on a monitoring protocol prior to release is essential. The second form of access is physical. Camping at the release site by the researchers is contra-indicated. Therefore transport between the release site and a base camp will be required. If the mode of transport is motorized, access can be at the mercy of stochastic events. For example, the Australian outback can be flooded for months. Although helicopters can be used to access sites, the high costs are likely to be prohibitive, severely limiting the extent of post-release monitoring. The release site should also allow access for monitoring (e.g. trapping, telemetry, focal or faecal sampling). If the habitat is so complex that effective monitoring is not possible, the fate of released animals cannot be determined, and the reintroduction could be seen as negligent (IUCN 1998).

2.2.6 Effects on the ecosystem

Reintroducing a species to their historical habitat should never be to the detriment of other native species or the ecosystem itself. Surveys at the release site should determine if an extinct species has been replaced by a potential competitor, particularly in the case of predators (IUCN 1998). It is possible that one endangered species may have replaced another over time, and a reintroduction may place both species at risk, jeopardizing biodiversity. Detailed knowledge of all species present at a potential release site is required, and the possible effects of the return of the extinct species may have on other species should be considered. This may be in terms of transmission of disease, competition for food or habitat, or direct predation. For example, reintroducing a predator could endanger a threatened prey species at the site, or a large herbivore could impose intense competition for limited food sources and dramatically upset a balanced even though altered ecosystem. In such cases, reintroduction should not proceed and alternative release sites investigated.

2.2.7 Timing of release

Long-term surveys, either of the site or of historical records, are required to determine the optimal timing for release (Stanley Price 1989). In Australia, annual rainfall records, combined with knowledge of the El Nino Southern Oscillation (ENSO) pattern, can be used to predict years when conditions are likely to be favourable for release. Ground surveys of food abundance and monthly rainfall records should then be used to determine the most favourable month for release. In Australia, this is normally after heavy rains following a drought or El Nino year. For optimal foraging, carnivores should be released later than herbivores to provide time for prey species to increase in numbers after the drought. Predicting the most opportune time for reintroduction is more critical if the animals are to be directly or 'hard' released to the site rather than first being acclimatised to the environment with supplementary food and water provided.

Ideally, released animals should breed just prior to or soon after release to allow for maximal rates of expansion of the population. However, females in late pregnancy should not be released, as stress on the mother could lead to abortion of the foetus. Knowledge of the breeding cycle of the animals is essential. For continuous breeders

such as *P. xanthopus* (Poole *et al.* 1985; Lim 1987; Robinson *et al.* 1994), reproductive status is more important, but for annual or bi-annual breeders, release should be planned to precede ovulation.

2.2.8 Fauna health

The health of the release animals is paramount. Reintroductions should release only the target species and not any disease they may harbour (Stanley Price 1989; Woodford and Kock 1991; Viggers *et al.* 1993; Woodford and Rossiter 1994; Cunningham 1996; Miller *et al.* 1999). Disease transmission between wild and released conspecifics should not occur if the reintroduction site meets IUCN (1998) criteria. Disease surveillance should be undertaken on wild counterparts near the release site, along with other fauna likely to come into contact with the release animals at the site in order to assess the presence of disease vectors and foci (Stanley Price 1989; Woodford and Kock 1991; Woodford and Rossiter 1994; Cunningham 1996; Conaghty and Schultz 1998). If necessary, released animals can sometimes be vaccinated against diseases known to be present in fauna at the release site. Conversely, captive animals found to be harbouring disease should be treated if possible or not released.

Many incidents of disease transmission have occurred through translocations, both from and to release animals (Woodford and Rossiter 1994). At least three cases of disease transmission have occurred in marsupials. *Setonix brachyurus* translocated from Rottne Island to Jandakot on the mainland were later found to be infected with *Salmonella* (Short *et al.* 1992), which was cited as a possible factor in the failure of reintroduction attempts. Fifty percent of Koalas, *Phascolartos cinereus*, translocated within Victoria from a tick-free area to one infested with the ixodid vector of tick paralysis rapidly succumbed to ticks (Woodford and Rossiter 1994). Common Brushtail possums, *Trichosurus vulpecula*, introduced to New Zealand from Tasmania have become vectors for bovine tuberculosis, aiding in the transmission of the disease throughout the New Zealand livestock industry (Hickling 1991).

Although numerous and varied, diseases in macropods are rarely of such severity as to markedly affect the health of the host (Speare *et al.* 1989), thus disease epidemics with high mortality are uncommon in free-ranging macropods (Conaghty and Schultz 1998).

Large populations of Euros, *Macropus robustus erubescens*, inhabit the same habitat as *P. xanthopus*, while Red Kangaroos, *M. rufus*, and Eastern Grey Kangaroos, *M. giganteus*, inhabit the surrounding plains. In Queensland, Swamp Wallabies, *Wallabia bicolor*, and Black-striped Wallabies, *M. dorsalis*, also occupy similar habitat to *P. xanthopus*. However most disease outbreaks are associated with environmental stressors such as drought, floods and overcrowding leading to nutritional stress, rather than contraction from sympatric species. As Speare *et al.* (1989) described, a drought may result in degraded habitat that promotes microorganisms (*Fusobacterium necrophorum*) that cause necrobacillosis (lumpy jaw) that if contracted, may lead to septicaemia and death. Thus, appropriate site selection and time of release will reduce the likelihood of diseases such as lumpy jaw causing reintroduction failure with macropods.

2.2.9 Limiting factors addressed

Factors known to have reduced reintroduction success include disease, competition from native and exotic species (including domestic livestock), over hunting or collection, habitat loss or degradation, unsuitability of release site, environmental pollutants and inappropriate management practices (Griffith *et al.* 1989; Stanley Price 1989; Kleiman *et al.* 1994; IUCN 1998). Such factors must be eliminated or controlled prior to release. Furthermore, they should be monitored and controlled indefinitely, and therefore it is essential that their costs be included in the total budget for the release. Restoration of degraded habitats may also be required before release to ensure sufficient resources. If predation, either by human or introduced predators, is identified as a limiting factor, research is required to determine the feasibility of its control (Short *et al.* 1992). In areas where the target species is hunted by humans for food, supplementary meat may need to be supplied to local inhabitants to alleviate hunting pressure.

Introduced predators have been implicated as the most limiting factor in the success of macropod reintroductions (Short *et al.* 1992). In Australia, long-term control of introduced Red Foxes, *Vulpes vulpes*, Feral Cats, *Felis catus*, and possibly Dingos, *Canis lupus dingo*, is essential for the success of any reintroduction of small to medium-sized fauna (Short *et al.* 1992; Priddel and Wheeler 1994; Sinclair *et al.* 1998). All three species are highly susceptible to '1080' (sodium monofluoroacetate) poison (McIlroy 1986). 1080 occurs naturally in plants in the genera *Gastrolobium* and *Oxylobium*, especially in Western

Australia. These plants are toxic to introduced herbivores, and introduced carnivores through secondary poisoning (Short *et al.* 1992), while most native fauna have evolved resistance to the poison. *Vulpes vulpes* and *C. l. dingø* are more likely to take poison baits through scavenging, but *F. catus* prefer live prey (Risbey *et al.* 1997; Risbey *et al.* 1999). Hence, for effective control of exotic predators a multidisciplinary approach is usually required and will involve baiting (both aerial and ground using a variety of baits), shooting and trapping.

As previously discussed, a number of factors have been implicated in the decline of *P. xanthopus*. Predation by *V. vulpes* (Copley 1983; Gordon *et al.* 1993; Sharp 1994; Hornsby 1997; Lapidge 2001) and to a lesser extent, Wedge-tailed Eagles, *Aquila audax* (Hornsby 1997; Sharp 1997b; Parker 2000; Bredhauer pers. comm.), *C. l. dingø* (Hornsby 1997; Sharp 1997a) and possibly *F. catus* is known to occur (Spencer 1991). Dietary competition from Feral Goats, *Capra hircus*, *M. r. erubescens*, Rabbits, *Oryctolagus cuniculus*, and domestic stock is also reported to occur, particularly in dry conditions, along with habitat competition from *M. r. erubescens* and *C. hircus* (Dawson and Ellis 1979; Lim *et al.* 1980; Copley 1983; Lim 1987; Lim *et al.* 1987; Henzell 1990; Lim *et al.* 1992; Sharp 1994). Habitat degradation through clearing of dispersal routes may also have played a role in the decline of *P. xanthopus*.

2.2.10 Community support

Ensuring community support for a reintroduction program prior to commencement is essential (Stanley Price 1989; Kleiman 1989; Kleiman *et al.* 1994; Miller *et al.* 1999). In some cases community members may have caused the species' local extinction through hunting or inappropriate land practices. Education, particularly through the media, information sessions, school education and establishing a presence in the community can all play key roles in soliciting support for the project. If a community has something to gain from the release, such as economic benefits of increased productivity through reduced stock predation, employment, eco-tourism, conservation education, traditional customs or a sense of community feeling, then the project is likely to be supported. The high costs and the resources used in reintroduction programs can create resentment amongst community members, particularly in poorer regions or where finances are being

diverted from other causes. For whatever reason, if genuine support cannot be obtained from the local community the site should be abandoned.

Reintroduction projects should also not cause any negative impact on the community, either economically or physically (Kleiman *et al.* 1994). The reintroduction of a large predator can do both by attacking livestock or humans. Reintroduction and associated site protection may also cause tourists to be diverted from the area or cause practices such as primary production or hunting to cease. Alternative employment or compensation is recommended in such cases (Kleiman *et al.* 1994).

2.2.11 Government support

Government support and legal or ethical approval are required at the outset of a reintroduction program (Kleiman *et al.* 1994). Such processes can be time consuming and should be factored in to the initial feasibility analysis and time budget. For an interstate or international project, approval and support needs to be obtained from all local, national and international organizations and may involve regulations and treaties such as CITES. Further approval or licensing is often required if the predators or competitors are to be eradicated from the release site through baiting, shooting or trapping. If a recovery team exists for the species, all experts or members, both government and non-government, should be involved in the initial decision-making processes and regularly informed of developments. Preferably, the species for reintroduction should be protected under federal and state laws.

2.3 Lambert Station in Queensland

2.3.1 Surveys in historic range

Although *P. x. celeris* is classified only as near threatened (IUCN 2000), the purpose of this project was to gain insight into ecological, physiological and genetic adaptations of reintroduced captive-bred animals, and not specifically for conservation purposes.

Petrogale xanthopus celeris have been bred at the Queensland Environmental Protection Agency, Charleville since September 1983 when three males and six females were collected from Lisburne Station. Previous releases of excess animals occurred into

Mariala National Park (Clancy and Close 1997) (Fig. 2.1); chosen because of the discovery of fresh wallaby pellets by P. McRae in 1986. Twenty-two excess animals have been released to two sites in four separate reintroductions between 1989 and 1994. No post-release monitoring of the animals was undertaken, and their fate is unknown. A survey of the area in 1997 yielded *P. x. celeris* faecal pellets (pers. obs.), and there have been recent sightings by park rangers (P. McRae, pers. comm., 2000).

Initial surveys for possible *P. x. celeris* reintroduction sites were conducted in November 1997. Lambert, Lisburne and Amaroo Pastoral Stations, along with Hell Hole National Park were surveyed for their distinctive faecal pellets, direct sightings and habitat suitability (Fig. 2.1). Populations were found on Lisburne and Amaroo Stations and in Hell Hole National Park. Further surveys were conducted on Lambert, Caranna, Lynbrydon and Listowel Valley Pastoral Stations in February and April 1998. Although thought to be extinct (N. Parsons pers. comm.), an extant colony was found on Lynbrydon Station (25°21'S, 145°15'E), and fresh faecal pellets were discovered on Listowel Valley Station (25°22'S, 145°12'E), 5 km from the Lynbrydon colony. Extensive searches of suitable residual tablelands on Lambert and Caranna Stations yielded no evidence of the species.

Petrogale x. celeris was reported to exist on Lambert Station during the 1960's by R. Bredhauer, the father of the current landowner (Lim *et al.* 1987). However, there were no signs during surveys in 1973 (Gordon *et al.* 1978) and 1984 (Lim *et al.* 1987; P. McRae pers. comm.). The last known *P. x. celeris* sighting on Lambert Station was by P. Bredhauer (current owner) in the early 1970's near sites 2 and 3 (Fig. 2.2) (P. Bredhauer pers. comm.). G. Bredhauer of Caranna Station reported seeing *P. x. celeris* on the property during his youth, however no date was given. No other historical records were found. Caranna Station was rejected as a possible release site due to current mining activities, the presence of significant numbers of *C. hircus*, and lack of predator control practices. Lambert Station was selected for the releases.

2.3.2 Site location

Lambert Pastoral Station is 140 km northwest of Charleville and 35 km southeast of Lisburne Station, the origin of the founding stock for the captive Charleville colony.

Figure 2.1 Location of Lambert Station reintroduction site for *Petrogale xanthopus celeris* in southwestern Queensland. Animals were bred at the Environmental Protection Agency captive breeding facility in Charleville, with founding stock originating from Lisburne Station. Amaroo, Caranna, Lynbrydon and Listowel Valley Stations, along with Hell Hole and Mariala National Parks were surveyed for possible release sites.

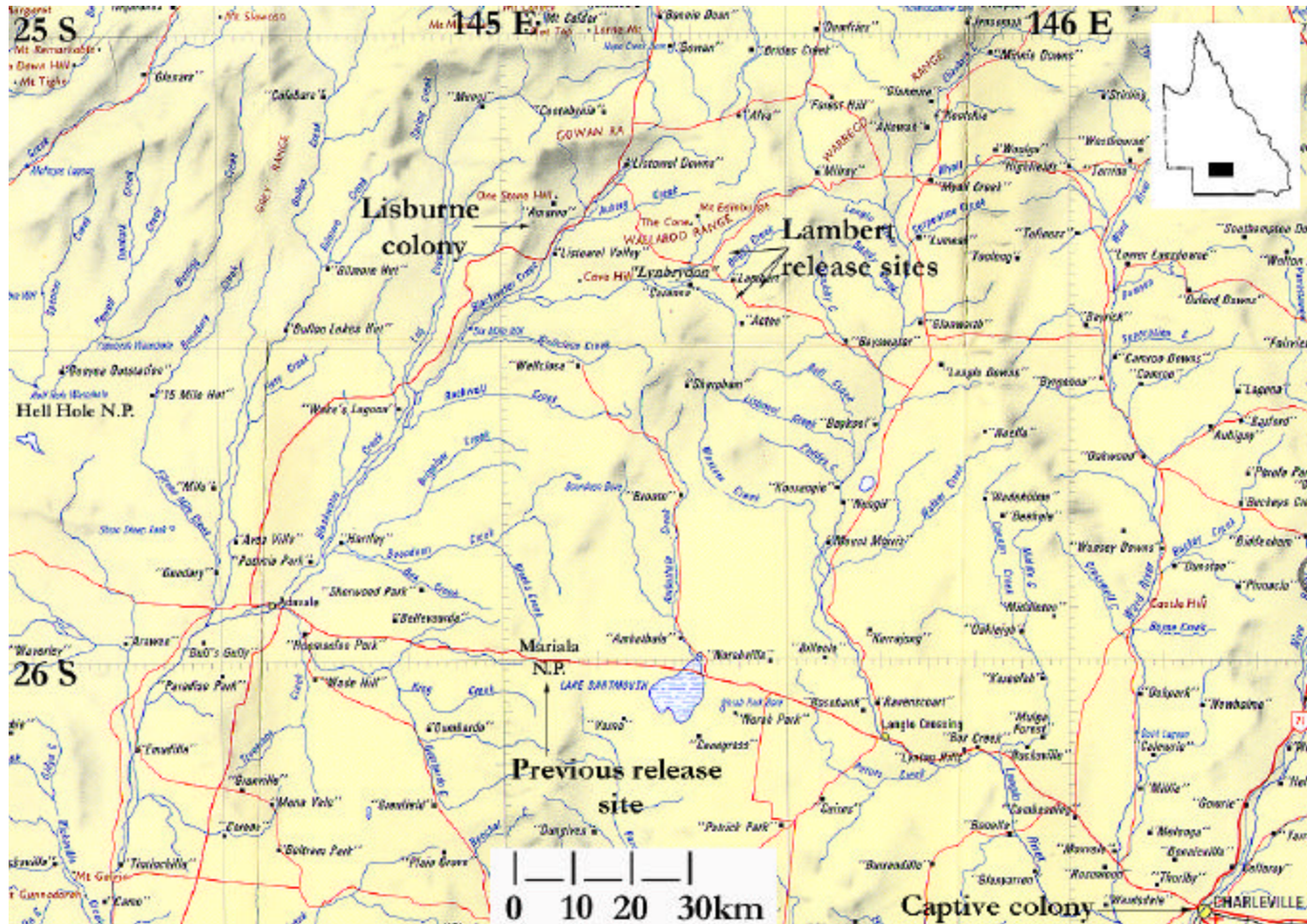
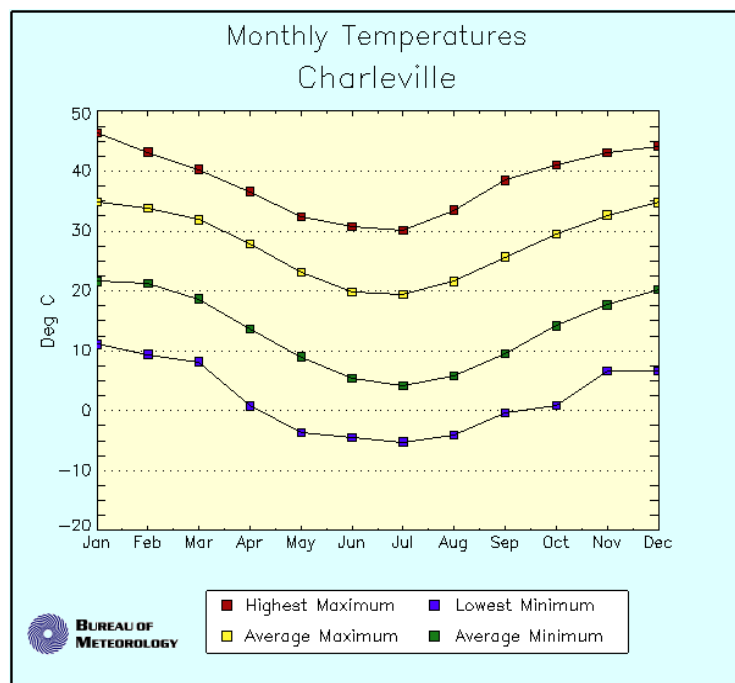
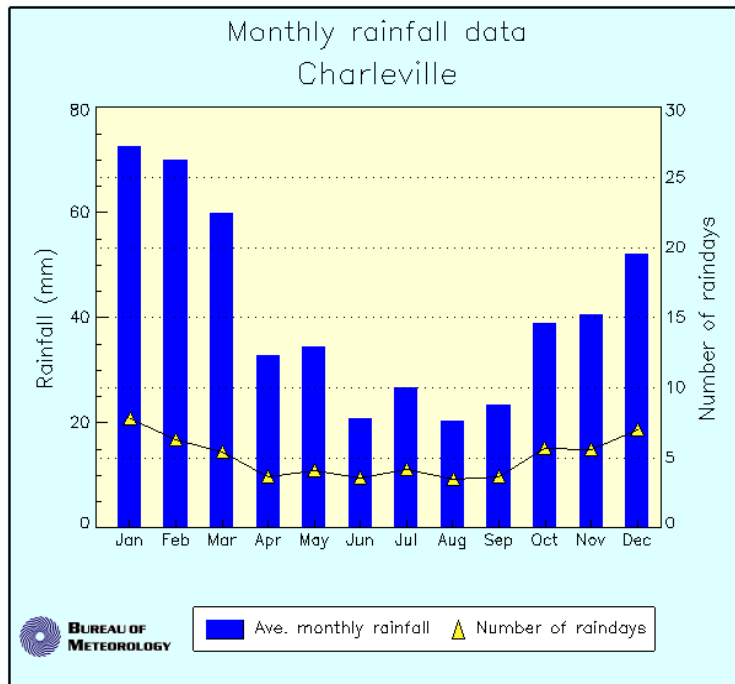


Figure 2.2 Mean monthly rainfall (n=82) and temperature (n=47) records for Charleville Post Office, southwestern Queensland.



Hence, the site is close to the source of the founding population for genetic reasons (Kleiman *et al.* 1994; IUCN 1998), and to the captive breeding facility for climatic adaptation and transport reasons. Charleville receives an annual rainfall of 498 mm (n=82) that falls predominantly through the summer monsoon season (Fig. 2.2), and is similar to that of Lambert Station (415 ± 152 (S.D.) mm, n=43). Temperature ranges for Lambert Station were not available, but were expected to be similar to those of Charleville (P. Bredhauer pers. comm.) (Fig. 2.2).

2.3.3 Habitat quality

Lambert Station lies at the eastern edge of the species' core range (Gordon *et al.* 1978; Lim *et al.* 1987; Gordon *et al.* 1993). Using the habitat scoring system of Lim and Giles (1987), suitable habitat on Lambert would be awarded one point each for the presence of steep vertical cliffs (between 1 and 10 m in high), outcrops, gullies, terraces, caves, rock-piles, and water (within 2 km of the nearest rock-pile, Fig. 8.1). A further three points would be awarded for boulders greater than 1 m in diameter and two points for the intermediate rock texture, resulting in a score of 12 out of 13 (Fig. 2.3). However, sites 2 and 3 (Fig. 2.4) would be more likely to score 11 due to the absence of distinct terraces.



Figure 2.3 North face of Site 1; typical *P. x. celeris* habitat on Lambert Station.

Three release sites were chosen (Fig. 2.4). Site 1 is separated from sites 2 and 3 by 6.49 and 6.84 km respectively, with site 2 and 3 being 1.83 km apart. Distances were measured between the closest outcrop complexes between sites using a GPS unit. The selection of three sites was based on a number of factors. First, it has been reported that several small populations will retain higher genetic diversity than a single large population (Frankham 1995; Forbes & Boyd 1997; Margan *et al.* 1998). Second, a stochastic event such as a fire would be less likely to result in all three colonies being affected. Third, although direct evidence of *P. x. celeris* dispersal is limited to 600 m (Sharp 1997a), the species is believed to be capable of longer dispersal (Lim 1987). The spacing of the release sites provided the opportunity to examine if wallabies disperse greater distances in non-contiguous habitats than the more contiguous habitat studied by Sharp (1997a) at Idalia National Park. Fourth, when initially surveyed, the *M. r. erubescens* population at Site 1 appeared significantly higher than those at of Sites 2 or 3. An initial aim of the project was to test whether *M. r. erubescens* had any effect on the establishment of released *P. x. celeris* through competition. However, numbers of *M. r. erubescens* equilibrated between the sites as the project progressed, negating testing of this effect.

Lambert Station contains low dissected residual tablelands of shallow, acid loamy lithosols with stone (conglomerate-tillite) and rubble cover. Steep cliffs combined with exposed rock outcrops are common, along with suitable gullies, terraces, caves and rock-piles for protection from the elements and from predators (Lim and Giles 1987; Sharp 1997a). Free water in dams is within 600 m of Sites 1 and 2 and 1.4 km of Site 3; these distances are substantially less than the distance *P. xanthopus* has been reported to travel to water (Lim 1987; Lim and Giles 1987). Scarps and tops of dissected tablelands are dominated by Bendee, *Acacia catenulate* and Mulga, *A. aneura*, open woodland with underlying Green Turkey Bush, *Eremophila gilesii*, Crimson Turkey Bush, *E. latrobei*, Twiggy Sida, *Sida intricata* and Velvet Potato-bush, *Solanum ellipticum*. The slope and base of Site 1 also contain Gidgee, *A. cambagi*, Wilga, *Geijera parviflora*, and Silver-leaved Ironwood, *Eucalyptus melanophloia*. The base of each site contains Silver-tails (or Silver Mulla Mulla), *Ptilotus obovatus*, and Showy Foxtail, *P. exaltatus*, Velvet Lantern-bush, *Abutilon calliphylum*, *S. intricata* and various chenopods including Ruby Saltbush, *Enchylaena tomentosa*, *Maireana*, *Bassia*, and *Rhagodia* species. Although traditionally Mitchell grass plains (*Astrelba* spp.), Buffel grass, *Cenchrus ciliaris*, was sown in 1964 to improve the pasture (P. Bredhauer pers. comm.). Much of the surrounding plains is now dominated

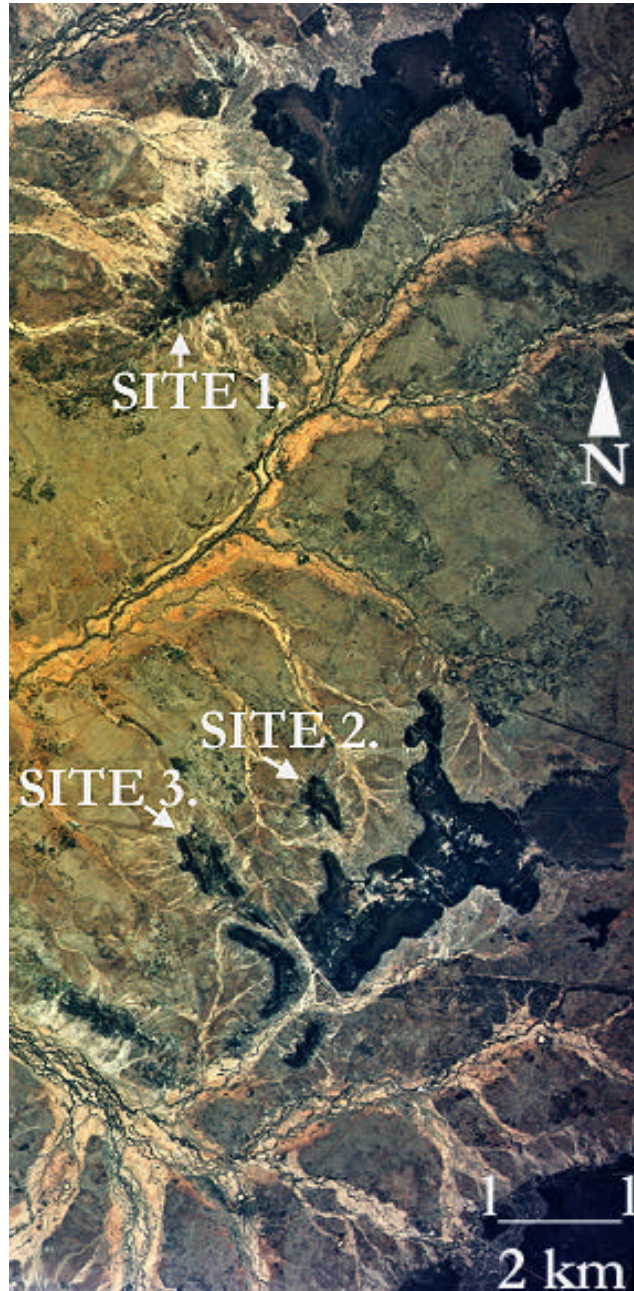
by *C. ciliaris* although it took nearly 10 years to properly establish. The property has been stick-raked to prevent erosion through water run-off and promote seed retention.

2.3.4 Habitat size

Lambert Station covers 17,000 ha, of which approximately 14% is rocky ranges, likely to be suitable *P. x. celeris* habitat. There are two main areas of suitable habitat, the range on which Site 1 is situated and the mesas and surrounding ranges on which Sites 2 and 3 are situated (Fig. 2.4). Site 1 is at the southwesterly tip (20 ha) of the range, which extends northeast for 8 km, and is on average 2 km wide. Thus 1,600 ha of suitable habitat exists south of the dingo barrier fence, most of which has been surveyed on foot or by vehicle. Lim (1987) reported a collective home range of 1,000 ha for a 140 animal *P. x. xanthopus* colony at Middle Gorge, South Australian. However, Sharp (1994) found *P. x. celeris* at Idalia National Park to occupy less than a fifth of the home range of *P. x. xanthopus*. On this basis, Site 1 should accommodate over 200 animals. Sites 2 and 3 are 40 ha semi-isolated mesas within 300 m of 800 ha of suitable habitat; this is the location of the 1970's *P. x. celeris* sightings on Lambert Station. This area should accommodate over 100 animals. Animals were released at the northern outcrop of each mesa (Fig. 2.3). Fifteen years of survey and monitoring of natural *P. x. xanthopus* colonies in South Australia has found only eight of 229 known colonies (3.5%) to contain more than 20 animals (Copley and Alexander 1997). Thus Lambert Station has sufficient habitat to accommodate many colonies of *P. x. celeris*.

The closest known *P. x. celeris* colony to Lambert Station is that 'rediscovered' on Lynbrydon Station, 17 km west of Site 1, during the course of the project. Although dispersal of up to 7 km has been reported for *Petrogale lateralis* (Eldridge and Kinnear 1999), the release sites on Lambert Station were considered to be far enough from natural colonies to avoid possible disease transmission, genetic swamping or social disruption.

Figure 2.4 Location of *Petrogale xanthopus celeris* release sites on Lambert Station, southwestern Queensland. Note non-contiguous nature of habitat and release sites.



2.3.5 Habitat use and protection

Lambert Station was formerly part of Listowel Downs Station before being sub-divided in 1953. The Bredhauer family purchased the 17,000 ha station freehold in 1961 and have managed the property since. The station is used for grazing 700 Hereford cattle and 5,000 Merino sheep. The low stocking rate (0.3 sheep and 0.04 cattle per ha; P.

Bredhauer pers. comm.), combined with a high abundance of *C. ciliaris*, attracts herbivores to the property, particularly macropods and *C. hircus* (P. Bredhauer pers. comm.). As a consequence, a macropod harvesting program is in operation, but is restricted to the plains between the ranges (S. Henshall pers. comm.). Feral Pigs, *Sus scrofa*, are also harvested. *Capra hircus* first appeared on Lambert Station in the early 1960's when 12 animals were sighted near Site 3. The population grew to an estimated 5,000 animals (P. Bredhauer pers. comm.) following a good season. A market was established for feral goat meat in the late 1970's, with mustering on Lambert Station commencing soon after (P. Bredhauer pers. comm.). *Capra hircus* have been mustered regularly since, with current numbers remain in the low 100's (P. Bredhauer pers. comm.). Lambert Station commenced aerial 1080 baiting at least once a year, often twice, in the late 1970's for exotic predators and *S. scrofa* with supplementary baiting as indicated. Fresh meat baits injected (*S. scrofa* baits) or sprayed (*V. vulpes* baits) with 1080 are distributed over the property, particularly along fence lines, at a minimum density of 1 bait per 7 ha (480kg of meat) (P. Bredhauer pers. comm.). S. Henshall, macropod harvester for Lambert and the surrounding pastoral stations of Acton, Caranna, Bayswater, Bayrick and Baykool (Fig. 2.1), remarked in early discussions that Lambert had the lowest visible *V. vulpes* density in the area.

A meeting was conducted with the landowners once Lambert Station was selected as a possible release site. Long-term protection and access were assured by the owners. Lambert Station is a well maintained property that undertakes annual track grading that provides good access to the release sites for post-release monitoring and for predator control. As the station is privately owned, human interference is limited, particularly on the hills. The station is also easily accessible by road from the towns of Charleville, Augathella and Blackall.

2.3.6 Effects on the ecosystem

Lambert Station contains abundant populations of *M. r. erubescens*, *M. rufus*, and *M. giganteus* (a combined population in excess of 20,000, P. Bredhauer pers. comm.), with smaller populations of *W. bicolor* and *M. dorsalis*. The added herbivore pressure in the form of 24 released *P. x. celeris* was deemed negligible. No evidence could be found to indicate that the niche of *P. x. celeris* had been filled by either a new or similarly

threatened species since their local extinction. Thus no negative ecological effects of the reintroduction could be identified; on the contrary, the additional exotic predator control program that would occur as part of the re-introduction should benefit all macropod species on the property (Banks *et al.* 2000). Each released animal was screened by S. Conaghty, veterinarian for the Adelaide Zoological Gardens, in order to minimize the risk of disease introduction (3.3.2.3).

2.3.7 Timing of release

Southwestern Queensland receives most of its rain in the summer monsoon season. Lambert Station received above-average rainfall (272 mm) in the months April to July, 1998, compared to the long-term average of approximately 115 mm in these months for the region (Fig. 2.2). August 1998 was chosen as the release date in order to take account of the high abundance of vegetation and the cool winter temperatures. *P. xanthopus* breeds continuously (Poole *et al.* 1985; Lim 1987; Robinson *et al.* 1994), so the release did not need to be timed for any reproductive factor.

Immediately before release, spotlight surveys were used to determine predator densities. Only one *V. vulpes* was sighted (and shot) in more than 100 km of surveys. It was deemed that predators were in sufficiently low numbers to allow the release to proceed (Sinclair *et al.* 1998), and the release took place on August 9, 1998.

2.3.8 Animal health

Diseases in free-ranging macropods are rarely severe enough to markedly affect the health of the host (Speare *et al.* 1989). In preparation for the reintroduction of *P. x. xanthopus* to Aroona Sanctuary, Conaghty and Schultz (1998) surveyed diseases in captive and wild *P. x. xanthopus* and *M. r. erubescens* and found little variation in microflora and parasite loads, with no pharmacological intervention deemed necessary in preparation for release. Trapping of wild *P. x. celeris* and other macropod species was beyond the scope of this study. However, *M. r. erubescens*, *M. rufus*, and *M. giganteus* shot as part of the harvesting program were examined for general body condition and signs of lumpy jaw. No shot macropods was found to show any visible sign of the disease, but an old *M. r. erubescens* skull was found at Site 3 with advanced mandible decay. Macropod Pox virus

was also detected in *M. r. erubescens* on Lambert Station by the author and S. Henshall during site selection surveys. Greater rates of infection were observed in the warmer months, possibly related to heat stress (S. Henshall pers. comm.; Speare *et al.* 1989). Although released *P. x. celeris* may be exposed to macropod herpesvirus (MHV), Wallal virus and *Toxoplasma gondii* from *F. catus*, vaccination is not possible. Furthermore, clinical disease has not been reported in free-ranging macropods for either virus, and wild *M. r. erubescens* has previously not been reported to contain *T. gondii* (Speare *et al.* 1989).

2.3.9 Limiting factors addressed

Petrogale. x. celeris is thought to have become extinct from Lambert Station through competition when large numbers of *C. hircus* in pre-mustering years stripped the hills of vegetation and drove *P. x. celeris* onto the surrounding plains to forage (P. Bredhauer, pers. comm.). *Capra hircus* kids are easy prey for *V. vulpes*, and hence *V. vulpes* numbers may have been artificially high (Lim 1987). The shooting and trapping *C. l. dingø* at the same time may also have allowed higher *V. vulpes* numbers (Gordon *et al.* 1993), which were not controlled until the late 1970s. Away from the safety of rocky outcrops, *P. x. celeris* is more vulnerable to *V. vulpes* predation. This scenario is supported by the population growth of *W. bicolor* since the control of *C. hircus* and *V. vulpes* commenced, and the recent 'discovery' of *M. dorsalis* near Sites 2 and 3 (pers. obs). Although *M. dorsalis* may have been present in the hills near Sites 2 and 3 all along, sightings have only occurred recently due to animals foraging on the open plain where they were previously were not observed. This is possibly due to an increased population size, resulting from reduced predation pressure, requiring greater foraging area.

Predation by *V. vulpes* and competition from *C. hircus* are considered the two threats to *P. xanthopus* (Dawson and Ellis 1979; Lim *et al.* 1980; Copley 1983; Lim 1987; Lim *et al.* 1987; Henzell 1990; Lim *et al.* 1992; Gordon *et al.* 1993; Sharp 1994; Hornsby 1997; Lapidge 2001). The owners of Lambert Station control both species as part of their management program. *C. hircus* are harvested on a regular basis because of the current high price paid for the animals (up to \$40 a head). *C. hircus* numbers were considered to be sufficiently low not to cause significant competition for *P. x. celeris*.

After the potential release sites were selected in February 1998, the author undertook an intensive ground and aerial baiting program for *V. vulpes* and *F. catus*. This was in addition to the shooting program undertaken by the author and S. Henshall (Lapidge and Henshall 2002; Appendix II). Ground baiting was conducted using 1080 *Fox-off* baits (Animal Control Technologies) in February, April, June and August 1998 prior to release, and September and November 1998, March 1999, April and October 2000 post-release. An average of 393 ± 42 baits were laid each time. Supplementary baits were laid whenever *V. vulpes* was sighted but not shot. Aerial baiting of the release hills and surrounding plains was conducted by the Queensland Environmental Protection Agency in December 1998, June 1999 and June 2000, the later two baitings to coincide with aerial boundary baiting undertaken by the station owners. Ground baiting was by two methods. Baits were buried to a depth of 5 cm at intervals of 100-200 m along all graded tracks around the release sites; *V. vulpes* make heavy use of such tracks. In addition, four concentric rings at a spacing of 200 m (i.e. base, 200m, 400m and 600m) were baited at intervals of 100-200 m around each release site. Bait locations were marked with *Dy-mark* water-based fluorescent paint and bait uptake was measured 3-10 days later to determine areas of *V. vulpes* activity. Ground baiting was not carried out across the whole station because of the susceptibility of working dogs, used for sheep mustering and pig catching, to 1080. Dogs were not used after bi-annual aerial baiting due to the wider broadcast of baits.

Felis catus are unlikely to take *Fox-off* baits, particularly if buried (Risbey *et al.* 1997; Risbey *et al.* 1999). Their destruction was consequently based on shooting. As macropod harvester for Lambert and surrounding stations, S. Henshall agreed to shoot *F. catus* and *V. vulpes* as part of his normal routine. Shooting of both species was further undertaken by the station owners, and by me during spotlight surveys to monitor predator and competitor numbers. Dragging of fresh macropod carcasses shot as part of the harvesting program was also used to lure predators into the open by creating a blood and scent trail. All *V. vulpes* (n=27) and *F. catus* (n=23) shot during fieldwork by any party were returned to Lambert Station for stomach content analysis to determine prey items consumed. *V. vulpes* (n=41) and *F. catus* (n=9) not examined were shot in my absence. Appendix I reports the results.

Despite large numbers of the natural competitor *M. r. erubescens* on Lambert Station, no control measures other than the harvesting program were undertaken. Similarly, large

numbers of the natural predator *A. audax* occur on Lambert Station, but their numbers were also not controlled; *P. x. celeris* has evolved in the presence of both *M. r. erubescens* and *A. audax*. Predation on *P. x. celeris* by *A. audax* is thought to be negligible (Lim *et al.* 1987; Hornsby 1997; Sharp 1997b; Parker 2000). Although present, *O. cuniculus* were in low numbers on Lambert Station, due to the area's soil types and high soil temperature and its unsuitability for warren formation and maintenance (Wilson *et al.* 1992). *O. cuniculus* were not controlled as they were deemed an easier food source for any remaining *V. vulpes* and *F. catus*, thus reducing predation pressure on *P. x. celeris*.

2.3.10 Community support

Although Lambert Station was surveyed in November 1997 and February 1998 as a potential release site, it was not until the end of the latter trip that a decision was made. During the February fieldtrip the landowners were consulted and agreed to provide assistance wherever possible through infrastructure, accommodation and predator control. S. Henshall, the resident macropod harvester, also agreed to shoot exotic predators whenever sighted, and to radio-track released animals weekly during my absence.

Publicity and additional support came from the Royal Zoological Society of South Australia, ABC Radio, both locally (4VL) and nationally (Radio National), and the *University of Sydney News*.

2.3.11 Government support

Early discussions with the Queensland Environmental Protection Agency (formerly Department of Environment) in November 1997 proved particularly fruitful. Although keen for a larger scale *P. x. celeris* release than they had previously attempted at Mariala National Park, they did not have the resources to undertake such a project and to ensure effective post-release monitoring. Vehicles, equipment and some funding were offered. Ethics approval was granted for the release in the form of a Scientific Purposes Permit (Permit No. W0/002083/98/SAA). *Petrogale xanthopus* is protected federally under the Endangered Species Protection Act 1992 and more recently the Environment Protection and Biodiversity Conservation Act 1999. Ethics approval was applied for and received

from the University of Sydney Animal Ethics Committee (ACEC Number: L04/4-98/3/2749). A permit for the use of 1080 poison was issued by the Queensland Department of Natural Resources, and a firearms license was obtained from the New South Wales Firearms Registry for the specific purpose of feral animal control.

2.4 Aroona Dam Sanctuary in South Australia

The reintroduction of *P. x. xanthopus* to Aroona Sanctuary was undertaken by the Royal Zoological Society of South Australia (RZSSA), NRG Flinders and the South Australian Department of Environment and Heritage (DEH) on September 26, 1996.

2.4.1 Surveys in historic range

A colony of *P. x. xanthopus* has been continuously maintained by the RZSSA since 1883 (Hornsby 1980). The founding stock for the colony is believed to have originated from Sliding Rock, 30 km southeast of Aroona Sanctuary. DEH was asked to suggest suitable reintroduction sites near Sliding Rock for release of captive-bred animals in order to test reintroduction practices.

Petrogale x. xanthopus was first seen by a European, Edward John Eyre, during his exploration of the Flinders Ranges in 1840. On August 21, 1840 he camped at the base of Mt Aroona, what is now Aroona Dam, and recorded seeing large numbers of rock-wallabies. A colony of *P. x. xanthopus* was known to occupy the site until 1982 (Barlow 1999). DEH indicated that the site was suitable for the reintroduction of *P. x. xanthopus*.

2.4.2 Site location

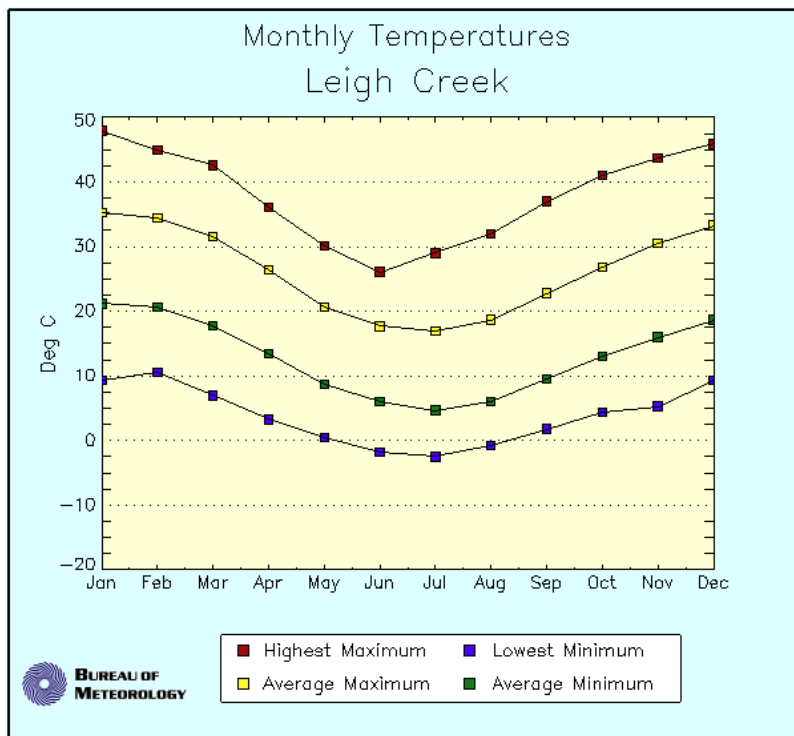
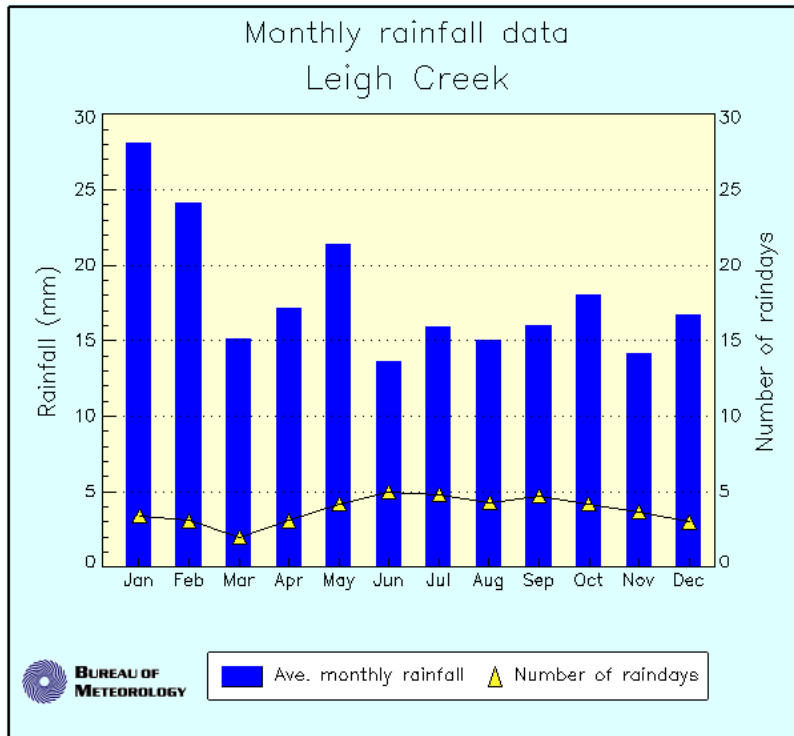
Mt Aroona is 500 km north of Adelaide and 30 km northwest of Sliding Rock (Fig. 2.5). Aroona Dam was built in 1955 to supply water to the Leigh Creek township and coal mine. The close proximity of the site to Sliding Rock, from which the RZSSA founding population was obtained, satisfies the selection criteria of Kleiman *et al.* (1994) and IUCN (1998). Leigh Creek receives an annual rainfall of 215 mm (n=30) throughout the year with no distinct season (Fig. 2.6). Rainfall for Monarto Zoological Park where the

animals for release were bred is not dissimilar (347 mm recorded 13 km away at Murray Bridge, n=114), although more seasonal and restricted to winter. Temperature ranges and seasonality for both sites are similar (Fig. 2.6) (Bureau of Meteorology 2001). The permanent water in the dam would also alleviate any chance of dehydration. Thus the site was deemed suitable climatically. Aroona Sanctuary can be directly accessed by road in five hours from Monarto Zoological Park; thus the likelihood of transport stress is small.

Figure 2.5 Location of Aroona Sanctuary reintroduction site for *Petrogale xanthopus xanthopus* in the northern Flinders Ranges, South Australia. Animals were bred at Monarto Zoological Park, 500km SSE of Aroona, with founding stock originating from Sliding Rock. Red Gorge and Wallaby Rock marked the two closest extant colonies of *P. x. xanthopus*.



Figure 2.6 Mean monthly (n=30) rainfall and temperature records for Leigh Creek Airport, South Australia.



2.4.3 Habitat quality

The sanctuary is at the western edge of the species current range in the Flinders Ranges (Copley 1983; Lim *et al.* 1987; Copley and Alexander 1997). The largest know extant colony of *P. x. xanthopus* is at 'Wallaby Rock', Depot Springs, 32 km east-north-east of Mt Aroona. A smaller and possibly transient population occurs at Red Gorge, North Moolooloo Station, 24.5 km due east of the sanctuary. Although this colony was reported to be extinct on a number of occasions (Copley 1983; Lim *et al.* 1987), surveys in 1996-97 (Lapidge 2000), and more recently in October 2000, located both fresh faecal pellets and animals. All other extant colonies are further east of the sanctuary, placing it on the periphery of the species' current range. Using the habitat scoring system of Lim and Giles (1987), Mt Aroona would be awarded one point each for the presence of steep cliffs, outcrops, gullies, rock-piles, caves, and water. A further two points would be awarded for boulders greater than 25 cm but less than 1 m in diameter and three points for the smooth quartzite rock texture, resulting in a score of 11 out of 13. However, the scoring system of Lim and Giles (1987) was developed in New South Wales and does not take into account the lack of terraces at many northern Flinders Ranges sites, in contrast to New South Wales and Queensland.



Figure 2.7 Mt Aroona; typical *P. x. xanthopus* habitat in Aroona Sanctuary.

The Flinders Ranges consist of folded sedimentary rock aged between 500-1000 million years old. The present landscape reflects the folding and faulting thought to have occurred hundreds of millions of years ago, yet continues (Corbett 1980). The steep quartzite Aroona Range is floristically characterized by Spinifex, *Triodia irritans*, hummock grassland with a Blackoak, *Casuarina pauper*, low open woodland with *Solanum ellipticum*, *Ptilotus obovatus*, *Enchylaena tomentosa*, Silver Senna, *Senna artemisioides artemisioides*, and Bullock Bush, *Alectryon oleifolius coriacea*, low trees. The more vertical slopes contain abundant Rock Sida, *Sida petrophila*, Desert Lantern-bush, *Abutilon leucopetalum*, and Flinders Ranges Corkwood, *Hakea endnieana*, while gentler slopes contain more abundant Rock Emubush, *Eremophila freelingii*, and *S. a. artemisioides*. After rain, Ward's Weed, *Carrichtera annua*, and Wild Hops, *Acetosa vesicaria*, become prevalent. Tree Tobacco, *Nicotiana glauca*, and Wild Lettuce, *Lactuca saligna*, are common around the waters edge.

The release site is shown in Fig. 2.6. The site at the base of Mt. Aroona was chosen for its rocky terrain and caves nearby, its suitability for radio tracking from surrounding peaks and its inaccessibility to tourists. Although more complex terrain exists in the sanctuary in the form of vertical cliffs, these were deemed too dangerous for inexperienced captive-bred wallabies.

2.4.4 Habitat size

Aroona Sanctuary covers 4,400 ha, with the Aroona range encompassing approximately 900 ha, most of which is suitable for *P. x. xanthopus* (Fig. 2.8). Lim (1987) reported that the collective home range of 140 wallabies was 1,000 ha. The presence of abundant water should further increase the carrying capacity of the sanctuary (Lim *et al.* 1987). Hence the sanctuary has sufficient continuous habitat to support a population of 140 animals or more. By comparison, the Middle-Bucharinga Gorge complex used by Lim (1987) is c. 600 ha in area.

Red Gorge has the closest extant *P. x. xanthopus* colony at 24.5km (Fig. 2.4), with open plains between the two sites. This distance was considered sufficient to prevent dispersal and intermixing of wild and released animals.

Figure 2.8 Location of *Petrogale xanthopus xanthopus* release site in Aroona Sanctuary, northern Flinders Ranges, South Australia.



2.4.5 Habitat use and protection

Aroona Dam was constructed between 1952-55 to supply water for the coal mining operations of The Electricity Trust of South Australia (ETSA). The area surrounding the dam initially formed part of Myrtle Springs Station and was used primarily for sheep grazing (Mincham 1983). ETSA initiated feral animal control around the dam in the 1980s (Barlow 1999) and applied for sanctuary status under South Australian legislation; this was granted in 1995. An electric fence was constructed on the northeastern boundary of the sanctuary to exclude sheep and goats, and exclosures were erected to monitor land rehabilitation within the sanctuary.

Aroona Sanctuary includes the rocky Aroona range, clay plains and hill slopes, wetland and sandy desert. Camping is permitted in three designated areas within the sanctuary, but rarity of facilities means that few people utilize the camping grounds. A permanent

caretaker lives in the sanctuary and NRG Flinders staff regularly visit the dam for water sampling. All-weather access can be gained to the release site for post-release monitoring.

2.4.6 Effects on the ecosystem

Large populations of *M. r. erubescens* inhabit the Aroona Range, with *M. rufus* and *M. giganteus* occurring on the surrounding plains. Surveys indicated no new or more abundant native species had occurred since the extinction of *P. x. xanthopus* 13 years prior. The release of 12 wallabies was deemed to have little ecological impact on the abundant *M. r. erubescens* if disease was not transmitted.

2.4.7 Timing of release

The *P. x. xanthopus* were released on September 26, 1998. The area had received 90 mm of rainfall between July and early September 1998, nearly double the long-term average of 47 mm for the period (Fig. 2.5). Rainfall at Aroona Sanctuary is irregular, with no distinct season, so the release occurred as soon as good rains had fallen in the cooler winter months, after exotic predators had been eradicated (S. Conaghty pers. comm.).

2.4.8 Animal health

The Veterinary Department of the RZSSA undertook disease surveillance of captive *P. x. xanthopus* at Adelaide Zoological Gardens and Monarto Zoological Park and wild *P. x. xanthopus*, *M. r. erubescens* and *F. catus* in and around Aroona Sanctuary prior to release (Conaghty and Schultz 1998). All seriological tests for MHV (n=38), Wallal virus (n=38) and Toxoplasma (n=54) in captive *P. x. xanthopus* and three wild animals were negative. In contrast, *M. r. erubescens* tested at Aroona Sanctuary exhibited 100% sero-conversion for MHV (n=2), 66% for Wallal virus (n=3) and 0% for Toxoplasma (n=1). Four of five *F. catus* sampled showed antibodies to *Toxoplasma gondii*. Gastrointestinal bacteriology and parasitology were examined, with 100% of wild *P. x. xanthopus* (n=6) and *M. r. erubescens* (n=3) testing positive for *Salmonella spp* and 30% (n=10) and 60% (n=5) for *Eimeria spp*. respectively. Thirty-six percent of captive *P. x. xanthopus* tested positive for *Salmonella infantis* (n=11) and 73% for *Eimeria spp*. (n=41). Skin scrapings and hair samples of captive *P. x. xanthopus* revealed *Aspergillus versicolor* and *Heterodoxus ampullatus* (louse)

present at low levels. Both are known to be common in wild *P. x. xanthopus* (Conaghty and Schultz 1998). Thus the captive and wild *P. x. xanthopus* showed similar results. Because the consequences to the microbiological balance and immunity of *P. x. xanthopus* to eradication, no treatment against any of these organisms were used on the pre-release animals.

Although released *P. x. xanthopus* were exposed to HMV, Wallal virus and *Toxoplasma gondii* at Aroona Sanctuary, vaccination was are not possible. Poxvirus was also detected in *M. r. erubescens* (n=1) at the sanctuary. This may spread to released *P. x. xanthopus*, but it is rarely of any consequence to free-ranging macropods (Speare *et al.* 1989).

2.4.9 Limiting factors addressed

The reason(s) for the decline of *P. x. xanthopus* at Aroona Sanctuary is not known, but a combination of *V. vulpes* predation, large numbers of *C. hircus* and drought have been implicated (B. Odermatt pers. comm.). Although the terrain of Mt Aroona offers good protection from *V. vulpes* predation, the availability of suitable food plants is limited, particularly in dry times. Drought attracts *C. hircus* to the permanent water supply of Aroona Dam and reduces food supply, both through added grazing pressure and lack of regeneration. *P. x. xanthopus* must forage on the nearby plains with their increased predation risk.

Although undertaken previously to a lesser degree, broad-scale feral animal control commenced in July 1995 after Aroona Sanctuary was identified as a possible release site (Barlow 1999). An intensive 1080 ground-baiting program for *V. vulpes* began involving RZSSA and ETSA staff, along with all pastoral stations neighbouring the sanctuary. J. Crutchett, Property Manager for Monarto Zoological Park, coordinated the program. Baits were initially applied at the recommended density of 4-5 *Fox-off* baits per km² every month. Poisoned chicken heads, bullock liver and fish were also used to attract cats. Surveys indicated little change in *V. vulpes* numbers in the sanctuary, so baiting was increased to 34 baits/km², with 1080-injected fresh meat, in May and June 1996 before being reduced to 7 baits/km² then stabilized at 3 baits/km² on a three-monthly schedule. This baiting regime has continued since release. A 10 km 1080 buffer zone was created around Aroona Sanctuary, which has since been expanded to 35 km, through the free

supply of *Fox-off* baits to pastoral stations on the proviso that they provided the labour. Until recently, aerial baiting has not been allowed in South Australia. A benefit to the pastoralists of the baiting program has been increased lamb yields (S. Conaghty pers. comm.).

Three-monthly spotlight surveys were undertaken throughout the sanctuary and surrounding area along set transects to estimate density of exotic species and *M. r. erubescens*, and the effect of baiting. Any *V. vulpes*, *F. catus* and *O. cynivalis* sighted were shot. *Capra hircus* and feral *Ovis aries* were regularly mustered and destroyed. The outbreak of Rabbit Calicivirus Disease in 1995 greatly reduced the *O. cynivalis* population in the Flinders Ranges. Large-scale warren destruction was undertaken by NRG Flinders staff. Large numbers of *A. audax* occur around Aroona Sanctuary, but numbers of this natural predator have not been controlled.

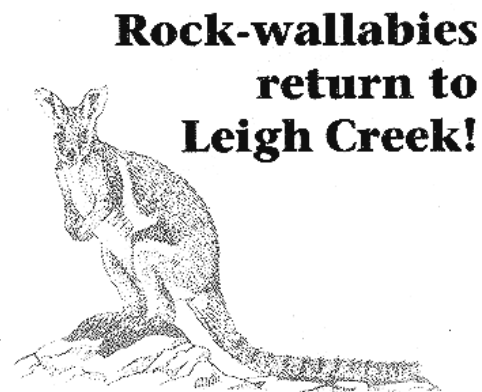
2.4.10 Community support

Due to the close proximity of Leigh Creek to Aroona Sanctuary (4 km), community support for the reintroduction was considered essential. Public meetings were held in the town and at the local school by the RZSSA to inform residents and school children about the project. Numerous interviews were conducted on the local community radio station as well as the local ABC station. Residents were informed of baiting in the sanctuary and surrounding area through a pamphlet delivered to every home and tourist warning them to keep domestic pets under control (Fig. 2.9). As part of their curriculum, the year 9 class at Leigh Creek Area School compiled and edited a periodic newsletter, *Rock Wallaby News*. Five issues have so far been produced and delivered locally, as well as to many Australian and international zoological institutions. Leigh Creek Area School received a special grant from the Education Department of South Australia to erect a three-station automated tracking system (triangulation) at the sanctuary, with data to be used for educational and research purposes.

Through the support of the local pastoral community for the 35 km fox-free buffer zone around Aroona Sanctuary, an area approximating 5,000 km² is now largely exotic predator free. In comparison, the area of Flinders Ranges National Park, where exotic species have been effectively controlled through Operation Bounceback, is

approximately 1,200 km² (Arkell 1999). Consequently, the Aroona Dam reintroduction has potentially benefitted many native species occurring in the zone, not just *P. x. xanthopus*. The pastoral community has also been instrumental in creating the Natural Heritage Trust-funded Aroona Catchment Biodiversity Enhancement Project in 1998. Furthermore, with funding from NRG Flinders and the RZSSA, an extensive flora and fauna survey of the northern Flinders Ranges was undertaken by the South Australian Biological Survey Unit (Brandle 1998).

Figure 2.9 Pamphlet distributed to residents of Leigh Creek and tourists informing them of the pending reintroduction.



As part of its ongoing commitment to the environment, ETSA has teamed up with the Adelaide Zoo and the Department of Environment & Natural Resources to bring back the threatened Yellow-footed Rock-wallaby to the Leigh Creek area.

Yellow-footed Rock-wallabies were originally found throughout the area but were wiped out by competition with goats and rabbits and predation by foxes.

To prepare for the release of the Zoo bred rock-wallabies in Spring 1996, the Aroona Dam Sanctuary will be cleared of these pest species. Please be aware of the 1080 poison baiting campaign in the conservation area. Signs will be posted in all areas under baiting.

For more information contact Beat Odermatt at ETSA on (086) 754366 or Erik Dahl in the Hawker DENR Office on (086) 484244.

ETSA Corporation



Support and funding for the reintroduction of *P. x. xanthopus* was received from NRG Flinders, DEH, the Zoological Parks Boards of New South Wales and Victoria, the Natural Heritage Trust, Los Angeles Zoo, Nissan Australia and Transceiver Services. NRG Flinders further supplied a house and logistical support for post-release

monitoring, as well as financial support for the research conducted described in this thesis.

2.4.11 Government support

The reintroduction of *P. x. xanthopus* to Aroona Sanctuary occurred with logistical support from DEH (Ref: 18/96). The monitoring research reported here was ethically approved by the same body (Ref: 3/98). *Petrogale x. xanthopus* was initially protected under the 1912 South Australian Animals Protection Act, then federally under the Endangered Species Protection Act 1992 and more recently the Environment Protection and Biodiversity Conservation Act 1999.

2.5 Discussion

Both the Queensland and South Australian reintroductions of *P. xanthopus* have adhered to site selection guidelines previously proposed by Kleiman (1989), Stanley Price (1989), Short *et al.* (1992), Kleiman *et al.* (1994) and IUCN 1998. Principally, both releases have been into an area of the species' former range from which they have become locally extinct, and the reintroduction sites are in close proximity to the original wild source of the founding stock for the captive breeding colonies. This was less of an issue in South Australia due to founding members being from various locations throughout the Flinders Ranges. Detailed surveys were undertaken prior to each release to determine the suitability of each site for reintroduction, as well as the feasibility of post-release monitoring. In each case the quality and size of the release site were suitable, assurances of long-term protection were obtained, limiting factors causing the species' initial decline were addressed, and support from both the community and government was obtained.

Table 2.1 summarizes and compares the two reintroduction sites using the selection criteria discussed. Although different in topography, Aroona consisting of high sandstone ridges and deep valleys and Lambert of low dissected tablelands of conglomerate rock, there are many similarities between the two sites. Both ranges peak at about 400 m above sea level. Although the Aroona range is dominated by *T. irritans* hummock grassland and Lambert by *A. catenulate* woodland, many of the same shrub

species and food plants occur at the two sites, including *Eremophila*, *Ptilotus*, *Sida*, *Solanum*, *Abutilon* and *Enchylaena* species. Both sites share a common temperature range throughout the year, with temperatures below 0°C in winter to near 50°C in summer. Possibly the biggest difference between the two sites is in terms of annual rainfall, with Lambert Station receiving summer monsoonal rains of nearly double the quantity of the sporadic Aroona Sanctuary rainfall.

Table 2.1 Comparison of Lambert Station and Aroona Sanctuary for site selection criteria.

Site selection criteria	Lambert <i>Queensland</i>	Aroona <i>South Australia</i>
1. Surveys in historic range	Adavale basin	Nth. Flinders
2. Site proximity to founding stock	35 km	30 km
3. Habitat quality	11-12/13	12/13
4. Habitat size	2,400 ha	900 ha
5. Habitat use and protection	Grazing, freehold	Sanctuary
6. Effect on ecosystem	Negligible	Negligible
7. Timing of release	August 1998	September 1996
8. Animal health	Few diseases	Few diseases
9. Limiting factors addressed	Yes	Yes
10. Community support	Land owners	Leigh Creek, NRG
11. Government support	Qld EPA	DEH

FOUNDER SELECTION AND PREPARATION

3.1 Introduction

Intentional releases of animals using wild-sourced stock are reported to have a higher success rate than those based on captive-bred animals (Griffith *et al.* 1989; Wolf *et al.* 1996; Miller *et al.* 1999; Fischer and Lindenmayer 2000). Griffith *et al.* (1989) reported a success rate of 75% (n=163) for projects using wild stock compared to 38% (n=34) for captive stock. Similar findings were reported by Wolf *et al.* (1996) of 71% (n=142) and 50% (n=24) for wild and captive stock respectively. With wild stock, success depended on whether the population density was high, medium or low at the time of translocation (77%, 78% and 37% success respectively), and whether it was increasing, stable or declining (83%, 63% and 44% success respectively) (Griffith *et al.* 1989). Similar criteria have not been examined with captive populations. In an analysis of 116 reintroductions (as defined by IUCN 1998 and in Ch 1) by Fisher and Lindenmayer (2000), 45 used wild stock, 52 used captive stock, and the source of 19 was unknown. A success rate of 31% was noted for projects using wild versus 13% for captive stock. Thus, intentional releases of animals should ideally use wild-born stock.

For many species such as the Przewalski Horse, *Equus ferus przewalskii* (FAO 1986) or Californian Condor, *Gymnogyps californianus* (Cohn 1993) it is not possible to use wild-caught stock because they are extinct in the wild. A greater risk to the species may also be involved using wild-caught stock, as the removal of wild animals may be to the demographic, social or genetic detriment of the source population (Kleiman *et al.* 1994; IUCN 1998). In contrast to captive populations, rarely can a whole wild population be

sampled demographically and genetically to determine the impact of removal of individuals.

Since conservation reintroductions are not normally instigated until the species is close to extinction, many projects rely on captive-bred animals for the founding population. Such has been the case in Australia with Eastern Barred Bandicoots, *Perameles gunnii* (Backhouse *et al.* 1994), Rufous Hare-wallabies, *Lagorchestes hirsutus* (Gibson *et al.* 1994), and Greater Bilbies, *Macrotis lagotis* (Southgate 1994). Moreover, Kleiman *et al.* (1994) suggested that the growth in scientific management of captive-breeding programs in zoos since 1980 has meant reintroduction of captive stock has become more feasible. Thus, although greater reintroduction success may be achieved through the use of wild stock, reintroduction procedures and animal adaptation upon reintroduction should be studied with captive-bred animals to improve the lower success rate so far achieved for captive endangered species, without the risk of jeopardizing remaining wild populations.

The purpose of the South Australian *P. x. xanthopus* release was to test reintroduction techniques and release manipulations for captive-bred rock-wallabies (Ferris and Macdonald 1995; Eldridge 1997b). The reintroduction of *P. x. celeris* in Queensland was undertaken to gain insight into how captive-bred animals adapt to the wild. Thus, the use of captive-bred animals was integral to both reintroductions. Chapter 3 introduces founder selection and preparation criteria, then addresses how these were used to select and prepare captive *P. x. celeris* and *P. x. xanthopus* for release to the wild.

3.2 Founder selection criteria

3.2.1 Founder selection

3.2.1.1 Knowledge of species

Knowledge of the species refers to both information on the reintroduction species' biology and information on previous reintroduction attempts with the species or genus. The former should include previous research on the species' ecological and physiological requirements, demography, behaviour, reproduction and genetics in the wild. Such information is critical to develop a reintroduction protocol for habitat selection, including its required size (taking into account home ranges and dispersal) and resources

(food, water and shelter), the size and structure of social groupings to be released, and if enough captive-animals are available to replicate this, and the potential predators and competitors of the release species and whether they need to be or can be controlled. If sufficient numbers of a captive-bred species are not present, it should not be released (Griffith *et al.* 1989; Kleiman 1989; Stanley Price 1989; Chivers 1991; Kleiman *et al.* 1994).

Reintroduction projects should always take into account the results of previous release attempts of the same or similar species, as well as have contact with persons having relevant expertise (IUCN 1998). Reasons for the success or failure of former attempts should be incorporated into the planning for later projects, so that reasons for success are repeated and mistakes are not. The lack of published literature on earlier reintroductions and reasons for their success or failure can often hinder this process (Short *et al.* 1992; Kleiman *et al.* 1994; IUCN 1998)

3.2.1.2 Self-sustaining captive population

Reintroduction of captive-bred animals should never jeopardize the remaining captive population, either genetically or demographically, and animals selected for release should be surplus to the future needs of captive propagation of the species. However, reintroduction should not be used as a way of disposing of surplus stock or as a form of captive-population management (Konstant and Mittermeier 1982; Kleiman 1989; Kleiman *et al.* 1994; IUCN 1998). Selection of release animals for particular traits such as age, sex or reproductive status may change the total captive population demographically in the short-term, but not necessarily to its long-term detriment. If genetic variation and an appropriate demography are maintained in the remaining stock then the population should suffer no long-term effects. However, for this to occur the captive population must be of a sufficient size, with a known sex and age ratio, and with baseline genetics of the entire population known.

It has been shown that, for wild-sourced stock, animals taken from large and increasing populations are more likely to form the basis of a successful translocation (Griffith *et al.* 1989). The same principle can theoretically be applied to captive populations; the use of large and increasing captive populations should result in more animals suitable for reintroduction, with less likelihood that the loss of the release animals will be detrimental

to the captive population. Furthermore, the growth of the captive population may indicate the reproductive viability of the population for reintroduction and provide a basis for modelling population growth post-release.

3.2.1.3 Animal management and suitability

For reintroductions of captive animals, the origin of the founding stock, the time they have spent in captivity (particularly generational time), and management practices used should be known (Stanley Price 1989; IUCN 1998). Whether the population has been managed actively (predetermined matings using a studbook) or passively (natural mate selection), particularly in the light of population genetic results, and whether wild animals or captive animals from other facilities have been added to the population to increase genetic diversity, should also be considered.

Captive animals for reintroduction should be from a healthy and rapidly breeding population that exhibits behavioural traits as close as possible to those of their wild counterparts. Although the adequacy in the wild of more detailed behaviours such as hunting or predator avoidance can be hard to assess in captivity (McLean *et al.* 1996; Miller *et al.* 1999), social interactions, feeding behaviour, dietary selection, reproduction, general health, locomotion and genetics can be readily monitored. Ideally, captive animals for reintroduction should have been exposed to conspecifics, appropriate habitat, natural climatic variations, natural foods, appropriate circadian rhythms, sympatric species and natural and exotic threats throughout their development. The more a captive population has been exposed to local conditions and variables, the more its natural instincts are likely to have been retained, and the less time and resources that will be required for pre-release training (Kleiman 1989; Stanley Price 1989; Biggins *et al.* 1998; Vargas and Anderson 1999).

The location of a captive population will often determine its suitability for reintroduction, both economically and climatically. Ideally, captive populations for reintroduction should be within a reasonable distance of both the researcher's base and the reintroduction site in order to minimise transport costs. Although many reintroduction projects acquire stock from international breeding programs, such as the Arabian Oryx, *Oryx leucoryx* (Stanley Price 1989), Golden-lion Tamarins, *Leontopithecus*

rosalia (Kleiman 1989) and *E. przewalskii* (Van Dierendonck & Wallis De Vries 1996), this contributes significantly to the cost of the project and may make it economically unfeasible. A greater disease and climatic risk can also be involved with animals obtained from overseas.

Climatic conditions that the animals have been held under, including enclosure facilities, should be considered for each population of the captive species. For example, a cold-climate species managed in a warm-climate zoological institution for many generations may have naturally selected for traits more suited to the warm climate, such as a thinner coat or smaller fat deposits. Facilities such as heated night quarters could further exacerbate adaptation to captivity. To counter this disadvantage, captive-bred *E. f. przewalskii* from warm climates such as southern North America and Australia undergo up to 10 years acclimatisation with supplementary provisions prior to reintroduction to the Mongolian steppe with its extreme winter temperatures (FAO 1996; Van Dierendonck & Wallis De Vries 1996). Thus, captive animals for reintroduction should ideally come from climatically similar areas to the release site.

Captive populations can be managed as a single large population, several smaller isolated populations, or small sub-populations with periodic translocation of animals. For reasons of genetic adaptation to captivity and resulting behaviour, the latter technique is recommended (Margan *et al.* 1998). Where numerous sub-populations are present for founder selection, animals should be chosen from the closest and most climatically similar facility to the release site so long as other parameters (e.g. genetics, health, size) permit.

3.2.1.4 Husbandry

To efficiently collect baseline data and select founders, and monitor animals post-release, individual animals must be frequently recaptured and handled for examination. Minimal stress procedures for capturing or anaesthetising and handling animals should be developed prior to commencing animal husbandry in order to minimise incidents of injury and capture myopathy, particularly when critically endangered species are involved. Different methods of capture, such as nets, anaesthetising through darting or races, will result in different levels of stress and injury in animals and this is likely to be species

specific. Methods that can be used to isolate individual animals for capture may be preferable to mass capture methods in terms of stress management. For species such as macropods who often 'throw' or drop their pouch-young upon capture, procedures should be developed before handling to combat or rectify the problem. In my opinion, the more an animal is handled (within reason) prior to release, the more readily it will tolerate recapture and handling post-release, thus facilitating post-release monitoring. However, care must be taken that animals, particularly predators and birds, do not imprint on humans. Furthermore, if capture methods used for post-release monitoring differ from those used in captivity, animals should be exposed to the new methods to facilitate post-release monitoring.

Many macropod species are prone to capture myopathy (Kakulas 1961, 1963a, 1963b; Munday 1972; Shepherd 1982; Munday 1988; Speare *et al.* 1989). The aetiology of capture myopathy in macropods is reported to be multifactorial, with both physical and psychological stress involved (Kakulas 1961, 1963a, 1963b; Shepherd 1982; Munday 1988), and vitamin E deficiency being a possible cause (Munday 1988; Rucker and Morris 1997). Myopathy has been reported in the Quokka, *Setonix brachyurus* (Kakulas 1961, 1963a, 1963b), Red Kangaroo, *Macropus rufus* (Shepherd 1982), Agile Wallaby, *M. agilis* (Speare *et al.* 1989), Red-necked or Bennett's Wallaby, *M. rufogriseus*, and Tasmanian Pademelon, *Thylogale billardierii* (Munday 1972), *L. hirsutus* (Cole *et al.* 1994) and once in *P. x. xanthopus* (S. Conaghty pers. comm.). Thus special care needs to be exercised during reintroduction of macropods, including supplementation with vitamin E.

3.2.1.5 Genetic suitability

Considerations for genetic suitability of a captive population for reintroduction are three-fold. First, as discussed in 2.2.2, the population should be genetically similar to conspecifics near the release site to ensure genetic adaptation to local conditions. This will help to alleviate the risk of outbreeding depression and genetic swamping being detrimental to either population should they meet (Kleiman 1989; Stanley Price 1989; Kleiman *et al.* 1994; Frankham 1995; IUCN 1998). Second, the captive population should be genetically diverse, with high levels of heterozygosity and low levels of relatedness between individuals, to minimize inbreeding. Inbreeding has been linked to reduced survival and ability to adapt to new environmental conditions, lower fecundity, lower

growth rates, developmental instability and defects, and increased susceptibility to disease (Ralls *et al.* 1988; Kleiman 1989; Stanley Price 1989; Haig *et al.* 1990; Leberg 1990; Stanley Price 1991; Frankham 1994; Kleiman *et al.* 1994; Frankham 1995; Lacy 1997; IUCN 1998; Miller *et al.* 1999). Third, the removal of animals for release should not be to the genetic detriment of the captive population. Thus, the genetic diversity of the captive population should be maintained after individuals have been removed for reintroduction (Kleiman 1989; Stanley Price 1989; Kleiman *et al.* 1994; IUCN 1998). Ideally a studbook for the captive animals will provide the primary, though not exclusive, information on which to base animal selection. However, the genetic background of each individual should be known for reintroduction, or should be determined at the outset.

Rapid and detrimental genetic adaptation to captivity of endangered species has been reported (Frankham 1994, 1995; Snyder *et al.* 1996). Frankham (1994, 1995) has suggested that minimizing time a population is in captivity, introducing wild genes, minimizing selection by reducing crowding, and minimizing effective heritability by equalizing family or enclosure populations will minimize genetic adaptation to captivity. However, the introduction of wild genes can be to the detriment of a captive population, both genetically and in terms of population health, particularly if the animal is from a different area to the initial stock or is harbouring disease foreign to the captive population (Margan *et al.* 1998). Although the introduction of a genetically distinct animal will increase heterozygosity, the resulting outbreeding may diminish genetic qualities required for local adaptation upon reintroduction to their historic range.

Despite the popularity and importance of preserving genetic diversity in populations for reintroduction, few practical examples are provided in the literature. In a study on Guam Rails (*Rallus owstoni*), Haig *et al.* (1990) tested six options for preserving genetic diversity:

1. randomly choose adults for breeding,
2. choose the most fecund captive breeders,
3. use allozyme data to choose parents that will produce the most genetically diverse chicks,
4. choose pairs to equalize founder contribution in the population,
5. choose pairs to maximize allelic diversity, and
6. choose pairs to maximize founder genome equivalents.

Results indicated that the last three options produced the most genetically diverse populations for release, with option 5 of maximizing allelic diversity resulting in the lowest loss (30.2%) of unique alleles, similar to option 6 (30.5%), but significantly lower than option 4 (35.2%) and option 3 (55%), based on an average of 10,000 simulations (Haig *et al.* 1990). Furthermore, options 3, 5 and 6 were found to increase heterozygosity in founders when compared to the captive population (Haig *et al.* 1990). Leberg (1990) also suggested establishing populations with as many unrelated individuals as possible. However, for many populations used in reintroductions relationships amongst individuals are unknown. In non-pedigreed colonies, relationships among individuals can be established using polymorphic nuclear genetic markers. Microsatellites are an ideal choice as they show strict Mendelian inheritance, are often highly polymorphic and enable the rapid screening of large numbers of animals for many loci (Bruford *et al.* 1996; Sunnucks 2000).

A trade-off exists between the time a population for reintroduction has spent in captivity and its possible adaptation to the captive environment. A captive population needs to prove that it is self-sustaining before its members are deemed suitable for reintroduction. If a captive population shows no evidence of increasing, due to either a demographical, physical, behavioural, morphological or genetic problem with breeding or survival, or to a lack of time in captivity, then population growth post-release will be uncertain, excluding the population as suitable for reintroduction. Furthermore, the shorter the time a population has been in a captive-breeding program, the fewer animals will be available for release, the more limited will be the amount of selection of founders for specific traits (sex, age, reproductive status, genetics) and the greater will be the risk of genetic and/or demographic damage to the captive colony. Alternatively, the greater the generational time a population spends in captivity, the more likelihood of adaptation to the captive environment, the more time and resources that will be required to prepare animals for the wild, and the smaller the likelihood of success (Griffith *et al.* 1989; Wolf *et al.* 1996; Miller *et al.* 1999; Fischer and Lindenmayer 2000). Thus, captive populations for release should be held long enough to produce sufficient animals for release (species dependent), to prove themselves reproductively, and to allow establishment of baseline data on growth, health, behaviour and genetics for later comparison if released.

3.2.1.6 Establish baseline data

If a captive population is deemed suitable for reintroduction, accurate and diverse baseline data on each animal should be established to allow for careful selection of founders for specific traits, and to provide the ability to investigate animal adaptation to the wild post-release (Stanley Price 1989; Kleiman 1989; Miller *et al.* 1999). Detailed records should be kept on genetics, fecundity, survival, development, health including disease, physical condition, blood parameters, ecto- and endo-parasite loads, population cohesion, diet, dental condition and behaviour (Stanley Price 1989; Woodford and Kock 1991; Viggers *et al.* 1993; Woodford and Rossiter 1994; Cunningham 1996; Miller *et al.* 1999; IUCN 1998). As well as establishing baseline data, initial investigations may discover effects of inbreeding if low survival and fecundity, high disease susceptibility, and retarded growth and development are found to occur. Furthermore, initial screening may exclude a large proportion of a captive population from release. For example, half (n=15) of the captive Bearded Vulture, *Gypaetus barbatus aureus*, population selected for reintroduction was unsuitable for release in central Europe because of physical damage, abnormal behaviour or imprinting on humans (Anderegg *et al.* 1983). Due to the unknown consequences to immunity and microbiological balance, vaccinations are not always recommended for some species (Conaghty and Schultz 1998), and their prior use in a captive population should be noted.

Although diseases in macropods are rarely of the severity as to markedly affect the health of the host, they are more prevalent and severe in overcrowded, nutritionally stressed or heat-stressed animals, and thus occur more often in captivity (Speare *et al.* 1998). The high prevalence of a disease such as necrobacillosis (lumpy jaw) in a macropod population will often mean that the population is unsuitable for reintroduction. Thus, if only one captive population of the species is available for release, reintroduction should not proceed.

There are three stages where accurate baseline data are essential. First it should be used to select appropriate animals for release and arrange them into new social groups or pairings using baseline data on genetics, age, sex, health and behaviour. Second, pre group-formation baseline data should be compared to post group-formation baseline data to determine the suitability of newly formed release groups. Evidence of increased

mortality, reduced fecundity, large weight loss, poor condition or aggressive behaviour mitigates against success. Although 'ideal' release pairings or groups can be determined using baseline data, adverse social interactions between individual animals may result in the need for new release groups to be formed. Third, post group-formation baseline data should be compared to post-release monitoring data to determine any changes in survival, fecundity, health, behaviour and genetics at regular intervals post-release.

Despite the calls for ethics, welfare and hypothesis testing in reintroduction biology for over 20 years (Krebs 1978; McNab 1983; Griffith *et al.* 1989; Kleiman 1989; Armstrong *et al.* 1994b; Kleiman *et al.* 1994; Serena and Williams 1994; Soderquist 1994; Soderquist and Serena 1994; Sarrazin and Barbault 1996; Waples and Stagoll 1997; IUCN 1998; Seddon 1999; Fischer and Lindenmayer 2000) the affect of introduction to the wild on captive-bred animals is unknown. Insight cannot be gained without diverse and accurate baseline data on every animal released.

3.2.1.7 Founder selection

The single most important factor in any reintroduction attempt is selecting the appropriate number, age, sex and genetic composition of the founding population (Stanley Price 1989), but no specific criteria exist. As a general rule, as many 'suitable' animals as possible should be released (Griffith *et al.* 1989; Wolf *et al.* 1996; Fischer and Lindenmayer 2000), although Griffith *et al.* (1989) also reported that the increase in success associated with releasing more animals quickly becomes asymptotic for birds at 80 to 120 individuals and for larger mammals at 20 to 40 individuals in total throughout the project. In analysis of 116 previous reintroductions, Fischer and Lindenmayer (2000) found no correlation between the number of individuals released and success, as indicated by the establishment of a self-sustaining population. Releases of 1-10, 11-20, 41-60, and 201+ animals in total resulted in similar success rates of 27%, 20%, 23% and 36% respectively, while releases of 21-40 and 61-100 suffered the lowest success rates of 11% and 12% respectively. Similarly, the reintroduction of 52 Brush-tailed Bettongs (*Bettongia penicillata*) was successful, while c. 700 Quokkas (*Setonix brachyurus*) was not in Western Australia (Short *et al.* 1992).

The effective number (N_e) of founders genetically is normally a fraction of the total number of animals released due to differences in reproductive success among individuals caused by factors such as dominance and mortality (Leberg 1990; Frankham 1995). In terms of genetic conservation in a founding population, it is the author's opinion that it is more beneficial to release a small number of genetically diverse animals than a high number of inbred animals. The size and carrying capacity of the release site must also be taken into account when determining the number of animals used to form a founding population. Minimum carrying capacity should be estimated during periods of low resource availability. If the founding population forms half the current carrying capacity of the release site, subsequent periods of low resource availability (e.g. during drought, flood or snow) may decimate the population. Moreover, animals released into an area with a low carrying capacity may not breed.

Natural population ratios of the specific species for reintroduction should be mimicked in selecting founders (Stanley Price 1989; Burgman *et al.* 1994), while allowing the population or colony 'room to grow'. A trade-off exists between releasing young animals that are reported to adapt more readily to the wild, and releasing older reproductively mature animals that will expand the reintroduced population more quickly but suffer higher mortality (Borner 1985; Kleiman *et al.* 1986). Immature animals may not survive long enough to breed, hence negating the aim of population growth. Short *et al.* (1992) also reported a higher survival of male macropods upon reintroduction, thus more females should be released, particularly if a natural colony has a female bias such as with *Petrogale* species (Lim 1987). The social groups formed by founders will also determine success. For example, in a study on wild otters (*Lutra lutra*) in Scotland, one male normally shared its range with two breeding females, consequently before reintroduction to England three unrelated cubs were joined for eight months in solitude. Post-release monitoring showed that the groups remained together and quickly bred (Green *et al.* 1984). A general model for the effects of founder age-class distribution on extinction risk has been created (Burgman *et al.* 1994), suggesting adults are more valuable than sub-adults in terms of reducing extinction risk for Leadbeater's Possum (*Gymnobelideus leadbeateri*).

A further consideration is whether reintroduction should occur as a single large population, possibly composed of several social groups, or as several small isolated

populations or several small populations allowing dispersal in between (i.e. a meta-population). The combination of ecological (Griffith *et al.* 1989; Wolf *et al.* 1996) and genetic (Frankham 1994; Margan *et al.* 1998) evidence predicts a meta-population approach will minimize the probability of failure of all populations from stochastic events, disease or other variables (Griffith *et al.* 1989; Wolf *et al.* 1996), while maintaining long-term genetic diversity and reproductive fitness as long as dispersal occurs between sub-populations (Margan *et al.* 1998). It has further been suggested that meta-populations offer a way of increasing exposure and public interaction and lowering costs of reintroduction projects (Craig 1994).

3.2.1.8 Animal welfare

The welfare of animals for reintroduction is paramount (IUCN 1998). Each animal released should have every likelihood of survival (Waples and Stagoll 1997). Not all individuals will have an equal capacity to survive, and animals possessing traits likely to reduce survival, such as injury, disease, deformity, or abnormal behaviour should be excluded from release (Stanley Price and Fairclough 1997). In a recent example, Canadian Lynx (*Lynx canadensis*) were reintroduced to Colorado despite their main prey item, Snowshoe Hares (*Lepus americanus*), being in short supply; the result was the starvation of all animals and public uproar (Kloor 1999). Furthermore, not all species are suitable for reintroduction or translocation (Stanley Price 1989; Pietsch 1994; Stanley Price and Fairclough 1997). Attempts to translocate Common Brushtail (*Trichosurus vulpecula*) and Common Ringtail (*Pseudocheirus peregrinus*) Possums have resulted in 70% to 98% mortality upon relocation through predation by *V. vulpes* due to naivety, stress and conspecifics being aggressive defenders of territory (Pietsch 1994). Translocation is thus not a viable management tool for possums, though reintroduction might. Reintroduction has only been recommended for one third of Australian vertebrate taxa considered in nationally endorsed Action Plans (Kennedy 1992; Serena and Williams 1994; Maxwell *et al.* 1996). Stanley Price (1989) listed nine criteria that make a species 'reintroducible':

1. generalists of extreme environments,
2. species tolerant of habitat change or of wide range of habitat conditions,
3. species with cohesive groups,
4. large animals,
5. explorer species,

6. scavengers,
7. species with sanctuaries in habitat,
8. nocturnal species, and
9. species whose behaviour can be manipulated.

To this list can be added:

10. species that breed early and have large clutches (Griffith *et al.* 1989),
11. herbivores (Griffith *et al.* 1989),
12. game species (Griffith *et al.* 1989),
13. mammals (Griffith *et al.* 1989; Wolf *et al.* 1996),
14. species well studied in the wild (IUCN 1998),
15. genetically diverse species (Frankham 1995),
16. species not genetically adapted to captivity (Frankham 1994),
17. species less disposed to inbreeding and its associated effects (author), and
18. flagship species to facilitate education (Kleiman 1989; Dietz *et al.* 1994).

For the purposes of this project or other experimental reintroductions, I would add:

19. species that have historically declined, but are not critically endangered,
20. charismatic species to facilitate funding and support,
21. species that breed well in captivity, and have a large captive population,
22. robust species to facilitate capture, handling and post-release monitoring,
23. habitat-specific species to facilitate post-release monitoring, and, in Australia,
24. species with high immunity to 1080 poison to facilitate exotic predator control.

Although most criteria relate (directly or indirectly) to the suitability of a species for reintroduction, criterion 23 opposes criteria 1 and 2 and relates directly to experimental reintroductions of less threatened species. Few species would fit all criteria, and some criteria naturally contradict each other, such as large animals (Stanley Price 1989) and species that breed early and have large clutches (Griffith *et al.* 1989). However, as a general rule, the more criteria that a potential species for reintroduction fit, the increased likelihood of animal survival, and the more likely the reintroduction will succeed. Alternatively, the less a species fits the criteria, the stronger the welfare case against reintroduction.

3.2.2 Founder preparation

3.2.2.1 Modelling release

The IUCN (1998) recommend modelling populations for release under various sets of conditions and undertaking Population and Habitat Viability Analysis (PHVA) prior to reintroduction. The principal advantage of modelling in reintroduction biology is to assess likely costs and benefits up front, in terms of dollars and animals or minimising extinction risk, thus identifying the potential success and viability of the project. Modelling can be particularly advantageous when critically endangered species are concerned, for example, the Bridled Naitail Wallaby, *Onychogalea fraenata* (McCallum 1994), Greater Bilby, *Macrotis lagotis* (Southgate and Possingham 1995), and Eastern Barred Bandicoots, *Perameles gunnii* (Reading *et al.* 1996), in identifying and preventing reintroductions that are unlikely to succeed and hence further endanger the species.

Modelling a reintroduction can be as simple as determining habitat size and potential carrying capacity of the release site, to detailed mathematical models of populations such as VORTEX (Lacy 1993) and ALEX (Possingham and Davies 1995; Southgate and Possingham 1995). Ecological models predominantly examine PHVA (Burgman *et al.* 1994; Lindenmayer 1994; McCarthy 1994; Southgate 1994; McCallum 1994; McCallum *et al.* 1995; Possingham and Davies 1995; Southgate and Possingham 1995; Reading *et al.* 1996; Sinclair *et al.* 1998), including impacts of predation and age-class distribution. Modelling of reintroductions has come to prominence in the last decade, partly because of the high previous failure rate of reintroductions (Griffith *et al.* 1989; Wolf *et al.* 1996; Fischer and Lindenmayer 2000). The aim of modelling is to try to identify and reduce potentially unsuccessful projects before they commence.

3.2.2.2 Pre-release training

Kleiman (1989) identified six forms of pre-release training, including predator avoidance, hunting and foraging, finding or constructing shelters or nests, locomotion, orientation and navigation, and social interactions. Training of captive-bred animals can either be indirect through provisioning or manipulation of the environment, or direct through physical training or the use of wild conspecifics. Stanley Price and Fairclough (1997)

recently published an IUCN position statement on behavioural constraints to release to be used in conjunction with IUCN (1998).

The principal reason for the high failure rate of macropod reintroductions has been predation by exotic predators (Short *et al.* 1992). Thus predator avoidance training would seem critical for reintroductions involving macropods. A great deal of resources were allocated to training critically endangered *L. hirsutus* prior to reintroduction to the Tanami Desert, Northern Territory (McLean *et al.* 1994; Underwood 1995; McLean *et al.* 1996). Despite becoming more cautious of predators after training, animals rapidly reverted to predator naivety once training ceased, and reintroduced populations were destroyed by *F. catus* and *V. vulpes*. Due to no survival of 79 *L. hirsutus* released into the Tanami Desert, reintroduction attempts have been discontinued and the species has been introduced to predator-free islands (Langford and Burbidge 2001). No successful cases of predator avoidance training in reintroduced mammals could be found in the literature.

Numerous captive-bred predators have been trained to hunt in captivity by first exposing them to carcasses, then to live prey. If sufficient prey exist at the release site many large carnivores such as wolf, cheetah and wild dogs show an innate ability to hunt and kill (Stanley Price 1989). However, smaller carnivores such as Black-footed Ferrets, *Mustela nigripes*, have been shown to greatly benefit from exposure to live prey, principally hamsters (*Mesocricetus auratus*) and prairie dogs (*Cynomys* spp.), during development in captivity (Biggins *et al.* 1998; Vargas and Anderson 1998; Vargas and Anderson 1999). Re-introduced *M. nigripes* exposed to white-tailed prairie dogs (*C. ludovicianus*) in captivity had a 19% greater chance of survival one-month post-release and 18% nine months post-release (Biggins *et al.* 1998). Similarly, Kleiman *et al.* (1986) reported that post-release survival of *L. rosalia* was correlated with feeding skill in captivity, where animals were given food puzzle boxes to increase foraging ability. However, captive-bred *P. x. xanthopus* adapted to an appropriate diet for the area and similar to their wild counterparts within three months, despite no direct training (Lapidge 2000).

Locomotion training may consist of teaching birds to fly or teaching a mammal to utilize a complex environment, such as rainforest or cliff face. Ideally, captive-bred animals should be raised in enclosures large enough to allow the development of flight, or with appropriate habitat in order to develop complex spatial orientation. Despite training in

spatial orientation, released *L. rosalia* were found not to be able to negotiate the more structurally complex rainforest (Kleiman *et al.* 1986).

Primates have been the main group of animals to undergo nest building training (Box 1991). Chimpanzees, *Pan troglodytes*, were provided with freshly cut branches in captivity and encouraged to build nests, which they did upon release (Box 1991).

The importance of social training or founder cohesion will depend on the sociality of the species (Armstrong *et al.* 1994a). For solitary animals, such as Brush-tailed Bettongs, *Bettongia penicillata*, the requirement for training is likely to be minimal (Delroy *et al.* 1986), while for primates with complex social structures training of captive-bred animals in social etiquette can require a great deal of time and resources (Box 1991). In a case where dominant *E. f. przewalskii* males from different breeding facilities were introduced for the first time at the reintroduction site, one male killed the other and the foals he had sired (Pereladova *et al.* 1999). Similarly, rock-wallabies live in a hierarchy where challenging younger males may receive serious injury from older males (Lim 1987), and thus the structure of formed colonies should be tested for founder or colony cohesion. Although no specific evidence exists, younger males in a cohesive colony are more likely to remain with the colony rather than dispersing upon release.

Pre-release training in some form has been used in 36% of mammal and 48% of bird reintroductions (Beck *et al.* 1994). However, only 50% of successful intentional releases of animals used training, leading Griffith *et al.* (1989) to report that survival of trained and untrained animals did not differ. Beck *et al.* (1991) suggested the use of post-release training for sedentary species, such as active presentation of natural foods and inducing locomotion of released *L. rosalia*, but only 12% of all reintroductions or 8% of successful reintroductions have used any form of post-release training (Beck *et al.* 1994).

An essential training exercise neglected by reintroduction practitioners is post-release recapture training. Theoretically, animals accustomed to traps in captivity are likely to be less wary of them in the wild and more inclined to enter traps, as long as no adverse consequences result, facilitating post-release monitoring. Although post-release monitoring is considered mandatory in reintroduction guidelines (Kleiman 1989; Stanley Price 1989; Kleiman *et al.* 1994; IUCN 1998), no reports of recapture training could be found in the literature.

3.2.2.3 Veterinary preparation

Detailed final veterinary examinations should be undertaken immediately before release to determine the health of each animal and to ensure that they are free from exotic diseases not present at the release site (Stanley Price 1989; Woodford and Kock 1991; Viggers *et al.* 1993; Woodford and Rossiter 1994; Cunningham 1996; Miller *et al.* 1999; IUCN 1998). Baseline data from the initial screening (3.2.1.6) should be compared to ensure no new diseases are present. This is particularly important for animals from non-local sources, as new diseases or parasites may have been contracted during shipment. A period of quarantine is advised for such animals (Stanley Price 1989; Woodford and Kock 1991; Viggers *et al.* 1993; Woodford and Rossiter 1994; Cunningham 1996; Miller *et al.* 1999; IUCN 1998). Although not always recommended (Conaghty and Schultz 1998), vaccinations may be required and should be instigated with sufficient time for animals to develop immunity. For example, reintroduced *O. leucoryx* had to be certifiably free from brucellosis, leptospirosis, tuberculosis, bluetongue, mange, anthrax, rinderpest and foot and mouth, and be vaccinated against anthrax, foot and mouth, clostridial diseases and pasteurellosis, which took several months (Stanley Price 1989).

For marsupials, intervention may be required to remove pouch-young for hand-raising to avoid certain loss in transport or at release, so that females are not in peak lactation at the time of release, and so the time before the first generation of wild-born young emerge is minimized (Conaghty and Schultz 1989).

3.2.2.4 Radio-collaring and tagging

IUCN (1998) stated that post-release monitoring is required for all (or a sample of) released individuals. To effectively monitor released animals, each individual should be permanently marked, either through tagging, transponders or tattoos, and have a radio-transmitter attached. Telemetry, particularly with mortality sensing transmitters, provides an efficient way of monitoring survival and dispersal. However, consideration must be given to appropriate methods of marking individuals and attaching radio transmitters. Not all individuals will accept transmitter attachment, resulting in a violent reaction and somersaults as occasionally occurs with *P. xanthopus* (Lim 1987). Incorrectly fitted radio-collars or other transmitter devices can also result in death. In a recent introduction of

four *M. lagotis* to Thistle Island, South Australia, three animals caught their forearms under the radio-collars, preventing locomotion and foraging. They consequently died of starvation. Later attempts have used transmitters attached to the base of the tail without incident (van Weenen pers. comm.). Tagging, tattooing and transmitter attachment should thus be undertaken prior to release to allow sufficient time to monitor infection or rejection. As a general rule, the smaller the release animal, the smaller the transmitter package (mainly battery) that can be used, the shorter the transmission life of the unit and the shorter the range of telemetry reception. These factors must be considered for post-release monitoring and in selection of a release site that will facilitate telemetry reception.

3.2.2.5 Transport

Transport plans should be developed that will minimize stress on individuals, including the time in transit (IUCN 1998). The mode of transport will be dependent on the distance between the captive facility and the release site; consideration should be given to whether a slightly longer road or sea transit would be likely to be less stressful than air transport. The possibility of disease transmission should also be kept to a minimum. Appropriate housing should be provided during transit, preferably in a device the animal is familiar with. Consideration should also be given to whether the animal would benefit from being anaesthetised during transit.

3.2.2.6 Release type

The type of release is generally described as 'soft' or 'hard'. According to Griffith *et al.* (1989), soft release refers to the provisioning of food and shelter at the release site for a variable amount of time post-release, while hard release refers to no provisioning. A release may also be immediate or direct versus delayed or indirect (Griffith *et al.* 1989). Other authors (Moore and Smith 1991) have defined a soft release as one that involves an acclimatisation period at the release site in holding facilities, along with provisioning. A hard release has also been defined as one that does not involve any pre-release training, acclimatisation, provisioning, predator control or post-release monitoring (Fischer and Lindenmayer 2000). Thus there are no clear or general definitions of the two types of release. Some evidence exists that animals in a soft release are more likely to develop site

fidelity and less likely to disperse (Stanley Price 1989). Irrespective of release strategy, no consistent association has been found between reintroduction or translocation success and release type (Griffith *et al.* 1989).

3.2.2.7 Intervention

No guidelines exist on intervention in reintroduction attempts (IUCN 1998). Individual projects should include contingency plans for intervention, for reasons of poor health, high mortality, or supplementary feeding, as rash decisions can often be made in the 'heat of the moment'. Many reintroductions of captive-born animals will suffer initial high mortality (Stanley Price 1989; Beck *et al.* 1991; Short *et al.* 1992). In my opinion, the level of mortality at which intervention occurs should be the product of the number of animals released, the likelihood of continued mortality, the conservation status of the species concerned, and whether the reintroduction is for conservational, aesthetic or experimental purposes. Reintroduction practitioners that allow the demise of a large proportion of an endangered species could be seen as negligent. The ultimate aim of any reintroduction is the establishment of a viable self-sustaining population, and if this seems unlikely to occur, then intervention and the return of remaining animals to captivity may be necessary. Such has been the case recently with the 20-year old Arabian Oryx (*O. leucoryx*) reintroduction in Oman (Gorman 1999).

3.3 *Petrogale xanthopus celeris* in Queensland

3.3.1 Founder selection

3.3.1.1 Knowledge of species

Petrogale xanthopus celeris is sexually dimorphic, with males (~ 8kg) attaining larger size than females (~6kg) (Lim 1987). The two sub-species of *P. xanthopus* have been found to be genetically distinct (0.72% average sequence divergence), and independent management has been recommended (Pope *et al.* 1996; Eldridge 1997c). Eldridge (1997c) estimated that the subspecies separated some 180 000 years ago.

Petrogale xanthopus lives in colonies ranging from six adults in a group or 'family' unit at a density of <0.62 km⁻², up to 120 animals comprising several groups at a density of 29 km⁻².

² (Lim 1987; Sharp 1994). However, most colonies (>96%) in South Australia are composed of a single group of less than 20 animals (Copley and Alexander 1997). Colonies are centred on suitable habitat such as cliffs or rock piles (Lim 1987), with availability of food resources and the number of shelter sites reported to determine their size (Sharp 1994). A colony typically consists of one or two larger adult males, several adult females and their offspring (Lim 1987; Sharp 1994). A number of other, normally younger, males and possibly females may occur nearby as young males typically disperse from the main group. All animals in a group have overlapping home ranges (Lim 1987). Habitat requirements and home ranges of *P. xanthopus* have been reported in Chapter 2.

Petrogale xanthopus has been described as the least social species of the genus *Petrogale* (Lim 1987). Although existing in colonies, *P. xanthopus* is regarded as a solitary animal though generally amicable to conspecifics, sympatric macropods and exotic herbivores (Nicholls 1972; Lim 1987). In the wild the species is predominantly nocturnal throughout the warmer months to periodically active throughout the day and night during the colder months (Nicholls 1972; Lim 1987). Lim (1987) reported no apparent maintenance of a dominance hierarchy by aggressive encounters, and that adults do not form strong bonds with each other except for a brief time during mating. Greatest interaction was reported for a female and her pouch young (Nicholls 1972; Lim 1987).

Petrogale xanthopus is a continuous breeder, with births occurring in all months of the year in both captive and wild populations (Poole *et al.* 1985; Lim 1987; Robinson *et al.* 1994). Although reproduction is correlated with rainfall in wild populations, births have been recorded during severe drought (Lim 1987; Robinson *et al.* 1994). The species exhibits embryonic diapause and is reported to have a two-phase reproductive strategy whereby young females produce offspring in a 1:1 sex ratio and older females in a 2:1 male to female ratio. Thus joeys from young females are essentially for recruitment into the natal colony and joeys from older females more for dispersal and to establish a male-exchange system with other colonies (Lim 1987; Robinson *et al.* 1994). The length of the oestrous cycle in the species is 32-37 days, gestation 31-33 days, pouch life 190-201 days, and interim pouch life before permanent exit 7-10 days (Poole *et al.* 1985; Hornsby 1978). The 'at foot' stage, when the joey follows its mother, is reported to be different for *P. xanthopus* compared to other macropods in that joeys are often left in the rock outcrops for protracted periods while the mother feeds and drinks on surrounding scree slopes

and plain, only returning to suckle their young. Although the transfer mechanism is not understood, the species is reported to transfer water from the mouth of mother to at-foot joey and as such is the only mammal to do so (Lim *et al.* 1987). This finding has been supported by aboriginal observations (Lim *et al.* 1987). Sexual maturity in the species varies greatly between 321 and 765 days, with an average of 18 months (Poole *et al.* 1985). Studies on wild *P. x. xanthopus* colonies have shown less than 10% of wild animals survive past six years of age, although individuals as old as 11 years have been recorded (Lim 1987; Robinson *et al.* 1994).

Petrogale xanthopus is classed as an intermediate browser-grazer (Sanson 1978). Although opportunistic in its dietary preferences (Sanson 1989), the species shows partial selectivity for grasses and herbaceous species of improved nutritional quality following rain (Dawson and Ellis 1979; Copley and Robinson 1983). In drier areas or under drought conditions the species shows greater selectivity for browse and plants with stellate trichomes (Copley and Robinson 1983; Allen 2001; Lapidge 2000). Lim (1987) reported that *P. x. xanthopus* requires free water for its survival in summer and was regularly observed travelling to water or drinking. However, Nicholls (1972) recorded only one drinking incident during summer observations of a wild *P. x. xanthopus* colony in the Flinders Ranges, South Australia, that being from a rock pool following heavy rain. Measurements of water turnover for the species were undertaken at Middle Gorge, South Australia, and showed two distinct patterns. In a dry summer, individuals had a water turnover rate of 53 ml/kg/day, while in a wet winter it increased to 147 ml/kg/day (Lim *et al.* 1987; Green 1989). Through comparison with measurements of plant water content, it was concluded that summer water turnover was in excess of that available from the vegetation and thus animals must be drinking free water (Lim *et al.* 1987). The average *ad libitum* intake of water by captive *P. x. celeris* was measured at 98.2 ml/kg/day (Murphy 1985). The field metabolic rate of *P. x. xanthopus* is reported as 2,209 kJ/day or 248 kJ/kg/day. The water turnover rate, field metabolic rate and inferred food intake rate of *P. x. celeris* are described in Chapter 9.

The reintroduction of *P. x. xanthopus* to Aroona Sanctuary, South Australia, in September 1996 was the first for the species. It was judged successful, based on the establishment of 80% (8/10) of the released animals one year post-release, their successful adaptation in diet (Lapidge 2000) and evidence of breeding (unmarked joeys sighted by the author in

December 1997). Therefore, similar release methodology was adopted for the later reintroduction of *P. x. celeris*. Furthermore, the project coordinator of the reintroduction for the pre-release to release stage (S. Conaghty) was the consulting veterinarian for the reintroduction of *P. x. celeris* to Lambert Station, Queensland.

The only other reintroduction of a *Petrogale* species is that of the Brush-tailed Rock-wallaby, *P. penicillata*, to Wombeyan Caves, New South Wales in 1980/81, which lasted 10 years. Although reported by Short *et al.* (1992), the New South Wales Department of Tourism was the instigator. No detailed post-release monitoring was undertaken of the 10 released animals, although there was evidence of recruitment, and the cause of their decline is unknown, though *V. vulpes* and *F. catus* predation was implicated. The Black-footed Rock-wallaby, *P. lateralis pearsoni*, was introduced to South Pearson, Thistle and Wedge Islands, South Australia in 1960, 1974-75, and 1975 respectively (Short *et al.* 1992; Copley 1994). The two latter releases failed, reportedly due to insufficient suitable rock habitat (Short *et al.* 1992). The initial release has succeeded, possibly due to more suitable habitat and the absence of exotic predators on the island (Short *et al.* 1992). Introductions of *P. penicillata* have occurred on Kawau, Motutapu and Rangitoto Islands, New Zealand and Oahu Island, Hawaii (Maynes 1989). Despite low founder populations ($n=2$ on Oahu), rapid establishment occurred on each island (250 *P. penicillata* on Oahu in 1981) (Maynes 1989; Veitch 1994), probably due to the lack of predators and availability of suitable habitat. From these results for *Petrogale* species it is clear that the number of founding animals is less important than the quality of habitat and absence of exotic predators.

3.3.1.2 Self-sustaining captive population

The captive population of *P. x. celeris* was established at the Charleville compound of the Queensland Environmental Protection Agency in September 1986 from three males (8.0, 5.0 and 4.5 kg), six females (7.5, 6.5, 5.0, 4.5, 3.0 and 1.6 kg) and two pouch young of unrecorded sex. To date, this is the only captive population of *P. x. celeris*. Few records on genetics, population growth, health or matings have been kept, but there has been no addition of wild animals to the colony since its formation. Between 1989 and 1994, 22 captive-bred animals (4 male: 8 female: 12 unrecorded) were released to two sites in Mariala National Park (Fig. 2.1) in four separate reintroductions.

The captive *P. x. celeris* colony was first surveyed as part of the current project in April 1998 and subsequently in June and August prior to release. The population was contained in three enclosures, each 0.1 ha in size (Fig. 3.1) and containing an average of 15 animals. All animals were captured individually using hanging nets, then sexed, weighed, measured (head, ear, forearm, tibia, hind foot, tail and pouch depth/scrotum size), ear numbered, ear tissue sample obtained for genetic analysis, bled for haematology and biochemistry analysis, examined for ecto-parasites and alopecia, and marked with temporary animal dye to avoid recapture. Animal ages were determined using Poole *et al.* (1985), Bach (1998) and mass versus age records of *P. x. xanthopus* from SPARKS (Adelaide Zoological Gardens), as no growth curves have been determined for *P. x. celeris*. At the completion of fieldwork (February 2001), growth curves were developed by the author (Chapter 7.3) and all animals re-aged. Forty-one animals (not including pouch young) in a sex ratio of 20 males to 21 females and varying in age from birth to >5 years were recorded. On the basis that the population was started with 9 adults or sub-adults in 1983, was reduced by 22 animals through previous releases, and currently contained 41 adults or sub-adults, a minimum of 54 births have been recorded in 15 years. However, this figure includes no deaths and is therefore a gross underestimate. Although requiring baseline genetics, the population was deemed of suitable size, demography, reproductive ability (86% of sexually-mature females carried pouch young) and health for reintroduction.

Figure 3.1 One of three *P. x. celeris* enclosures at the Charleville EPA compound, each 0.1 ha in size and housing 15 animals.



3.3.1.3 Animal management and suitability

The founding population was sourced from Lisburne Station (Fig. 2.1) 160 km northwest of Charleville in September 1983. Assuming a generation time of 18 months (Lim 1987), the population had been in captivity for ten generations. The population had been passively managed, with natural mate-choice and periodic intermixing among the three enclosures, all of which are joined by runs. Animals had rarely been handled prior to the currently project, and so the animals were fearful of humans. Animals were fed a commercial macropod pellet diet (Ridley Agriproducts) and lucerne, with occasional supplements of cut Mulga (*Acacia aneura*) browse. Various *Acacia* and *Eucalyptus* species were present in the enclosures, along with Showy Groundsel (*Senecio magnificus*) after rain. The animals readily consumed native vegetation. A drinking trough provided free-water. Rock piles in each enclosure provided shelter and habitat for climbing, thus maintaining spatial orientation, and no night quarters were provided. Although the enclosures protected the animals from exotic predators, periodic exposure to Brown Snakes (*Pseudonaja textilis*) occurred. Evidence of social unrest was evident in each enclosure in the form of alopecia (Fig. 3.2). Alopecia, or fur loss resulting in bald patches, was the result of fur pulling during male fights for dominance. This most likely occurred because of the high stocking rate and unnatural sex ratios in each enclosure. Although the only *P. x. celeris* colony existing, its close proximity to the release site ensured similarity in climate and a reasonably short time in transit for release. In summary, passive management practices and the provision of appropriate food and habitat had ensured natural behaviour was retained in the captive-animals, thus making them suitable for reintroduction.

3.3.1.4 Husbandry

The initial capture method of using a padded loop on a pole was abandoned after the death of F3. All subsequent captures used a hanging net strung between the enclosure fence and a suitable tree, a procedure used by the RZSSA (S. Conaghty pers. comm.). Selected individuals were herded into the net using people as shields, with a person waiting to grasp the animal by the base of the tail as soon as it entered the net. Animals were placed in hessian bags and processing commenced after the animal's heart rate had slowed. No injuries occurred with this technique, and its selectivity allowed for low

overall physical exertion and stress in the colony, indicated by reduction in creatinine concentrations throughout sampling (Table 3.4) (Finco 1997; Conaghty and Schultz 1998).

Blood samples (4 ml) were obtained from a lateral tail vein using a 21g x 1" needle (Terumo) for adults and 23g x 1¼" needle for juveniles (Terumo) and a 5 ml Luer syringe (Terumo). From the 4 ml blood sample, 0.5 ml was placed into a 1 ml EDTA vial (Sarstedt) and refrigerated for later haematological analysis; the remaining 3.5 ml placed into a 5 ml Lithium Heparin vial (Sarstedt), centrifuged at high speed for 10 minutes with plasma extracted, from which 0.5 ml was transferred to a vial and frozen for biochemical analysis, and 1ml to an Eppendorf tube and frozen for Vitamin E analysis. All animals were administered intramuscularly with 0.2 ml Vitamin-E-Selenium (Vitamin-E-Selen: 150 IU Vitamin E and 0.5 mg selenium/ml: Hoechst Roussel Vet) to prevent capture myopathy whilst handling (S. Conaghty pers. comm.). Whole blood was analysed on a Sysmex K4500 (TOA Medical Electronics Co. Ltd., Japan) and plasma biochemistry on a Discrete Biochemical Analyser (Cobas Mira, F.Hoffmann-La Roche & Co., Switzerland) at the University of Sydney's Department of Veterinary Anatomy and Pathology. Vitamin E concentration was analysed using reverse-phase High Performance Liquid Chromatography (Shimadzu, Japan) at Royal Prince Alfred Hospital's Department of Clinical Biochemistry.

Many female *P. x. aeleris* dropped their pouch young upon capture, particularly if they were more than 13 weeks of age. Thrown young were collected from beneath the drop net or from within the hessian bag, processed separately, kept warm in cloth until the processing of the dam had ceased and, then replaced in the pouch. The pouch was then sutured closed using dissolvable (24h) Ethincon surgical catgut sutures (Chromic 2/0 G113). The dam was then gently released without inversion. Local anaesthetic (Lignomav: 20mg/ml Lignocaine hydrochloride: Mavlab) was sometimes injected into the lip of the pouch to facilitate stretching to accommodate the return of larger joeys. The suturing technique was adopted after discussions with *P. xanthopus* veterinarians, with the author being trained in the technique by D. Schultz (Adelaide Zoo). Despite 23 pouch young being thrown upon capture, none was thrown after the dam had been released and no losses occurred. Sutures were normally chewed by the dam after the enclosure was vacated, which reopened the pouch. Taping the pouch closed using

masking tape was tested once, but the pouch young was thrown after the dam's release (later sutured) and the method was abandoned.

Although *P. x. xanthopus* is predisposed to capture myopathy (S. Conaghty pers.comm.), it remains rare in the species and no incidents were recorded in the captive *P. x. celeris* population during or after handling.

3.3.1.5 Genetic suitability

Due to the close proximity (35 km) of the release site (Lambert Station) to the wild population from which the founding stock was sourced (Lisburne Station) (Fig. 2.1), the captive population was deemed genetically similar to wild conspecifics in the area. A genetic study undertaken by Pope *et al.* (1996) using mitochondrial DNA and microsatellite loci from 11 animals in the captive population (7 males: 4 females) showed levels of heterozygosity and nucleotide diversity exceeding those of two large wild *P. x. celeris* colonies (0.709 ± 0.099 compared to 0.200 ± 0.154 and 0.417 ± 0.191). Pope *et al.* (1996) concluded that reintroduction of captive animals would be unlikely to fail through a lack of genetic diversity, thus indicating the colony's genetic suitability.

Baseline genetic diversity was not available for the entire captive population, consequently ear biopsies (4mm diameter) were collected from each of the 41 captive animals during the initial April 1998 survey and stored in 100% ethanol. Total genomic DNA was isolated from each biopsy using the salting out procedure of Sunnucks and Hales (1996). Six microsatellite loci were used: Y76, Y148, Y151, Y170, Pa55 and Pa595, two more than Pope *et al.* (1996). The first four were cloned from *P. x. celeris* (Pope *et al.* 1996), while the latter two were cloned from *P. assimilis* (Spencer *et al.* 1995). Conditions for amplification of loci were as previously described (Spencer *et al.* 1995; Pope *et al.* 1996). Genotypic data for each locus were analysed using BIOSYS-1 (Swoffard and Selander 1981) to obtain estimates of average number of alleles per locus (i.e. allelic diversity = AD) and average heterozygosity (H) (Nei 1978). Conformance to Hardy-Weinberg equilibrium was tested by the Markov chain method of exact probability using GENEPOP v.3 (Raymond & Rousset 1995). Estimates of genetic relatedness (R) between each pair of individuals were made using KINSHIP 1.2 (Queller & Goodnight 1989). All six microsatellite loci were polymorphic in the captive *P. x. celeris* population,

with a total of 26 alleles (2 - 7 alleles per locus) detected (Table 3.1). At each locus the population was in Hardy-Weinberg equilibrium, with $P>0.3$ in all cases.

Average heterozygosity in the current study (0.656 ± 0.045 for 40 individuals) was similar to Pope *et al.* (1996) (0.709 ± 0.099 for 11 individuals), indicating uniform levels of heterozygosity throughout the colony. Although established 15 years earlier with only nine founders, the captive *P. x. celeris* colony was shown to have greater allelic diversity and heterozygosity than the two largest known wild colonies in Idalia National Park (Pope *et al.*, 1996). This finding is opposite to that of Leberg (1990), Frankham (1994, 1995) and Lacy (1997) who report loss of genetic diversity in captive populations. As the captive colony was allowed to breed freely, this result possibly indicates the species' ability to avoid inbreeding.

Table 3.1. Observed allele frequencies, number of alleles per locus (AD) and average heterozygosity (H_e) at 6 microsatellite loci in the captive *Petrogale xanthopus celeris* population. For each locus, allele size (in base pairs) is shown in bold. Mean values (\pm standard errors) are indicated at the base of the table.

Locus	n	Allele Frequencies						AD	H_e
Y76	40	161	167	171	185				
		0.025	0.250	0.200	0.525			4	0.629
Y148	40	193	197	199	201				
		0.350	0.213	0.425	0.013			4	0.660
Y151	40	171	173	175	177	179			
		0.150	0.225	0.250	0.050	0.325		5	0.766
Y170	40	130	132	134	136				
		0.087	0.175	0.125	0.613			4	0.578
Pa55	40	150	152						
		0.488	0.512					2	0.506
Pa595	40	287	291	323	329	359	363	375	
		0.375	0.162	0.112	0.050	0.100	0.112	0.087	7
									0.797
								4.3±0.7	0.656±0.045

3.3.1.6 Baseline data

Baseline data on fecundity, survival, growth rates, behaviour, development, injury, deformity, body mass and condition, haematology, biochemistry, dental health and parasites were established for captive *P. x. celeris*. Eighty-six percent of females greater than 18 months of age, and some as young as 12 months, carried pouch young of various

ages in captivity. The sex ratio of 17 pouch young recorded was close to unity (8 :9). First conception was recorded in five females as 16 ± 2 month and 3.8 ± 0.4 kg. Four incidents of pouch young mortality occurred during the five-month baseline data collection period (24%). The deaths of seven adults (17%) occurred through the same period. One female (F3) died possibly from a sub-dural haemorrhage caused by being caught in a loop net or being bled from the jugular vein. The animal was necropsied at Sydney University by M. Krockenburger. Both methods were abandoned after the death. One older male (M38) dislocated its right hip after handling and was promptly euthanased at the Charleville Veterinary Clinic, with the reproductive tract being removed and sent to D. Taggart (Monash University, Melbourne). Two animals (M37 and F26) died from 'sand cholic', the former necropsied by M. Krockenburger and the later by S. Conaghty, possibly resulting from sand injection during digging for food items in the fine red sand of the enclosures. Both animals died between baseline data collection periods. The three remaining deaths (F1, M20 and M23) occurred on May 19, 20 and 21 respectively between the April and June baseline surveys, after being fed *A. aneura* branches by the local keeper on May 17. Animals had previously been fed only leaves and it is not known whether the animals died for ingestion of the branches or other causes as the carcasses were disposed of before they could be collected for necropsy.

Figure 3.2 Alopecia in a male (M28) *Petrogale xanthopus celeris* caused through fighting.



Growth rates of head, ear, forearm, pouch depth/testes size, tibia, hindfoot and tail were recorded for all animals. However, sufficient data were not obtainable to form accurate growth curves for the species from the captive data alone, mainly due to the short period of time. Prior to colony formation, 33% of males showed evidence of intraspecific aggression in the form of alopecia (Fig. 3.2), indicating social unrest in enclosures, possibly due to overstocking. No incidents of abnormal behaviour were recorded; all animals exhibited appropriate flight responses and natural circadian rhythms. No incidents of irregular development or deformity were recorded. Body mass, haematology (Table 3.2) and biochemistry (Table 3.3) were monitored both pre- and post colony, or release group, formation to determine the affect of handling and new social groups. Natural body mass fluctuations occurred pre-colony formation and haematological and biochemical values were comparable to captive *P. x. xanthopus* (Conaghty and Schultz 1998). Before release, 44% (4/9) of males and 27% (4/15) of females either carried lice or had done so in the previous six months. Although not identified, lice were possibly *Heterodoxus ampullatus*, a species commonly associated with both captive and wild *P. xanthopus* (Conaghty and Schultz 1998). No evidence of necrobacillosis (Lumpy Jaw) was found in any captive animal. The captive *P. x. æleris* colony was thus deemed suitable for reintroduction due to high fecundity and survival (excluding deaths due to management practices), appropriate behaviour and development, suitable haematology and biochemistry and no apparent disease.

3.3.1.7 Founder selection

Twenty-six of the 41 animals (11 : 15) were deemed of suitable age (1-5 years), mass (> 4 kg for radio collaring) and health for release. Three colonies of eight animals (3 : 5) were formed. Each consisted of a large older male, two younger males and five reproductive females of varying age in order to mimic a wild colony and, at the same time, utilise the maximum number of suitable release animals available. The small colony size should also allow recruitment without dispersal, and was well below the potential carrying capacity of each release site (2.3.4). Colony assignment was initially based on matching age and mass among the three colonies, while taking into account haematology and biochemistry. Average relatedness values and allelic diversity were subsequently calculated for each colony. Individual animals were then transferred between colonies to minimise average colony relatedness and maximise allelic diversity based on the

recommendations of Haig *et al.* (1990), while maintaining the age, sex and mass ratio of the colony (Table 3.4). Newly formed colonies were housed separately and monitored to ensure social cohesion for two months prior to release. A 22% reduction in signs of intraspecific aggression was recorded based on the presence of alopecia (see 3.3.1.3). Non-selected animals, being too old or too small (<4 kg), were temporarily housed together in the largest enclosure (Fig 3.1): four adult and five juvenile males, and one adult and seven juvenile females; thus a sex ratio of close to 1:1 was maintained. This population was separated into male and females in May 1999, and individual animal records initiated in an attempt to control breeding.

The allelic diversity of the base population was well represented within each release group, with all alleles having frequencies > 0.1 (22/26) (Table 3.4), and a colony average of $91 \pm 5\%$ allelic diversity (24/26 alleles) being retained. The four rare alleles (frequencies < 0.1, Table 3.1) were not represented in each release group since they were only present in a limited number of individuals. However, they were retained in the captive population, thus the removal of the 24 release animals was not to the genetic detriment of the captive population.

Release groups were constructed to minimise relatedness, and the average relatedness between individuals within each colony was close to zero (Table 3.5). Within each release group some pairs of individuals had relatively high R values (i.e. $R > 0.5$) (Table 3.5), but this does not necessarily indicate a close relationship (e.g. full sibs) as these animals shared common (i.e. high frequency) alleles at most loci (Table 3.4). This is illustrated by male 31 and female 13 from group 1 ($R = 0.524$, Table 3.5), where these individuals share at least one common allele at all six loci (Table 3.4).

3.2.1.8 Animal welfare

Previous reintroductions and introductions of *Petrogale* species indicted their aptness for reintroduction providing suitable habitat was present but *V. vulpes* was not (Maynes 1989; Short *et al.* 1992; Copley 1994; Veitch 1994). Individual *P. x. celeris* selected for release were all healthy adults or sub-adults that showed no behavioural, morphological or physiological evidence that would exclude them from release. While all animals were reproductive, younger to middle-aged animals were selected to facilitate adaptation to the

Table 3.2 Pre- and Post-colony formation haematology for captive *Petrogale xanthopus celeris*. Pre-colony or release group formation haematology values were obtained from captive animals before they were rearranged into colonies for release (post-colony formation). Key to abbreviations: WBC- white blood cells, RBC- red blood cells, HGB- haemoglobin, PCV- packed cell volume, MCV- mean corpuscular volume, MCH- mean corpuscular haemoglobin, MCHC- mean corpuscular haemoglobin concentration, and PLT- platelets. Results for captive *P. x. xanthopus* from Adelaide Zoo courtesy of Conaghty and Schultz (1998). Results for wild Allied Rock-wallabies, *P. assimilis*, are from Spencer and Speare (1992). *Significant difference between pre-and post-colony formation values (ANOVA $P < 0.05$).

			MASS	AGE	WBC	RBC	HGB	PCV	MCV	MCH	MCHC	PLT
			Kg	Months	$\times 10^9 \text{ L}^{-1}$	$\times 10^{12} \text{ L}^{-1}$	g L^{-1}	LL^{-1}	fL	pg	g L^{-1}	$\times 10^9 \text{ L}^{-1}$
Colony 1	Pre-colony 1	N=7	5.30±0.7	22±3	8.7±0.9	5.7±0.5	176±15	0.519±0.045	91.5±3.2	31.1±1.2	340±4	157±110
	Pre-colony 2	N=7	5.34±0.5	24±3	7.9±1.4	5.45±0.44	168±9	0.488±0.033	90.5±4.3	30.9±1.0	342±8	140±54
	Post-colony	N=8	5.19±0.7	26±3	6.1±2.4	4.72±0.68*	150±19*	0.448±0.065	94.9±3.4*	31.8±1.8	335±15	170±107
Colony 2	Pre-colony 1	N=6	5.14±2.01	22±14	6.3±1.8	5.2±0.8	165±23	0.484±0.067	93.4±5.0	31.8±2.0	341±12	123±73
	Pre-colony 2	N=7	4.81±1.72	24±14	7.3±2.5	5.35±0.26	165±10	0.478±0.030	89.34±4.6	30.9±1.4	346±9	144±120
	Post-colony	N=8	4.91±1.61	26±14	6.0±2.0	4.75±0.41*	152±16	0.462±0.040	97.6±5.6*	31.9±2.0	328±24	158±73
Colony 3	Pre-colony 1	N=8	5.34±1.04	23±7	6.9±0.6	5.7±0.6	182±16	0.527±0.058	92.0±4.0	31.9±0.9	334±35	127±41
	Pre-colony 2	N=7	5.13±0.8	25±7	5.4±0.8	5.45±0.40	173±11	0.499±0.032	91.8±3.7	31.7±0.9	346±10	95±52
	Post-colony	N=8	5.29±1.0	27±7	6.3±1.1	5.09±0.49	163±16	0.488±0.054	95.7±3.4	31.9±0.9	334±11	182±60*
Adelaide Zoo	<i>P. x. xanthopus</i>	N=72			6.5±1.5	6.1±0.3	176±9	0.545±0.032	88.3±3.7	28.6±1.1	324±10	120±39
Wild	<i>Petrogale P. assimilis</i>	N=>293			9.0±0.1	5.0±0.2	135±4	0.412±0.012	82.3±1.4	27.0±0.6	32.8±0.1	147±66

Table 3.3 Pre- and Post-colony formation biochemistry for captive *Petrogale xanthopus celeris*. Pre-colony or release group formation haematology values were obtained from captive animals before they were rearranged into colonies for release (post-colony formation). Results for captive *P. x. xanthopus* from Adelaide Zoo courtesy of Conaghty and Schultz (1998). Vitamin E analysis was not undertaken pre-colony formation due to its late inclusion owing to earlier prohibitive costs. *Significant difference between pre-and post-colony formation values (ANOVA $P < 0.05$).

			MASS	AGE	Protein	Albumin	Triglycerides	Creatinine	Urea	Globulins	Vitamin E
			Kg	Months	g L ⁻¹	g L ⁻¹	mmol L ⁻¹	imol L ⁻¹	mmol L ⁻¹	g L ⁻¹	imol L ⁻¹
Colony 1	Pre-colony 1	N=7	5.30±0.7	22±3	99.4±16.5	63.8±12.3	1.37±0.51	151±29	9.32±2.08	35.6±7.4	-
	Pre-colony 2	N=7	5.34±0.5	24±3	87.1±20.7	47.4±9.5	0.86±0.70	133±8	10.4±1.77	39.6±11.7	-
	Post-colony	N=8	5.19±0.7	26±3	79.9±19.4	44.1±7.7	1.11±0.52	119±17	10.77±2.11	35.9±13.6	6±1 (n=3)
Colony 2	Pre-colony 1	N=6	5.14±2.01	22±14	120.7±28.1	80.4±18.0	1.23±0.69	165±37	10.71±2.43	40.3±13.1	-
	Pre-colony 2	N=7	4.81±1.72	24±14	98.9±21.1	59.9±16.0	0.64±0.20	144±21	10.56±1.73	39.0±14.4	-
	Post-colony	N=8	4.91±1.61	26±14	66.8±25.0*	39.9±12.4	0.75±0.21	109±38*	10.0±2.20	30.9±15.2	10 (n=1)
Colony 3	Pre-colony 1	N=8	5.34±1.04	23±7	97.1±24.3	61.3±16.5	1.00±0.63	140±21	8.16±1.91	35.8±10.0	-
	Pre-colony 2	N=7	5.13±0.8	25±7	79.1±7.2	45.6±3.0	0.71±0.26	127±15	10.59±1.39	33.6±5.6	-
	Post-colony	N=8	5.29±1.0	27±7	77.3±26.4	48.4±8.4	0.91±0.62	138±34	10.8±1.80	31.1±16.5	10±3 (n=4)
Adelaide Zoo	<i>P. x. xanthopus</i>	N=79			67.1±6.4	52.2±5.2	-	122±22	-	15.3±4.4	2.1±2.5 (n=69)

Table 3.4 Colony composition, age structure, mass, condition scores and individual genotypes (at 6 microsatellite loci) for reintroduced *Petrogale xanthopus celeris*. Allele sizes are given in base pairs. Condition is an arbitrary score referring to body condition and independently assigned by S. Conaghty, veterinarian (See 7.2.3). A score of 5 refers to faultless condition, while 1 is extremely poor.

Colony ID	Individual ID	Sex	Age (mth)	Mass (kg)	Condition (out of 5)	Individual genotypes at 6 microsatellite loci					
						Y76	Y148	Y151	Y170	Pa55	Pa595
Colony 1	9		30	6.75	4	171,171	193,197	175,177	130,134	150,152	375,363
	12		24	5.00	3	167,185	197,199	171,173	132,136	150,150	291,291
	13		27	4.50	2	167,185	193,199	177,179	136,136	150,152	287,323
	25		29	4.95	3	167,171	193,193	179,179	134,136	150,150	291,327
	30		24	4.80	3.75	161,185	199,199	175,179	132,136	150,152	287,291
	31		23	5.75	3	167,185	193,199	175,179	134,136	150,152	287,323
	35		27	4.95	4	167,161	197,199	179,179	134,136	150,150	359,323
	43		25	4.80	4	185,185	193,199	171,175	136,136	150,152	287,291
Mean ± S.D.			26 ± 2	5.2 ± 0.7	3.1 ± 0.6						
Colony 2	2		51	8.25	4	167,185	199,199	173,173	136,136	150,152	287,359
	6		38	5.60	2.5	171,185	193,199	173,179	132,136	150,152	287,327
	15		48	6.50	3.75	185,185	193,197	175,179	136,136	152,152	359,363
	19		15	4.00	3.5	185,185	193,197	173,179	136,136	150,152	291,323
	24		14	4.20	3.5	167,171	193,199	171,179	136,136	150,150	287,287
	29		16	4.00	2.5	171,185	193,193	173,175	132,134	150,150	287,375
	32		15	4.00	3	185,185	193,199	179,179	136,136	152,152	359,323
	41		23	4.00	2.5	185,185	197,199	173,175	136,136	152,152	287,363
Mean ± S.D.			28 ± 15	5.1 ± 1.5	3.1 ± 0.6						
Colony 3	5		31	5.50	3	167,185	197,199	173,173	136,136	150,152	287,359
	14		33	6.00	3	171,171	193,197	175,179	134,136	150,152	323,363
	17		28	5.00	2.5	167,185	193,199	173,179	134,136	152,152	287,327
	22		38	7.00	3.5	167,171	193,199	175,177	130,134	150,150	287,323
	27		15	4.00	3.5	167,185	197,199	171,175	130,136	150,152	287,375
	28		24	5.00	2.5	167,185	193,193	173,179	136,136	152,152	291,375
	34		22	5.60	3	167,185	193,199	171,179	136,136	150,152	287,359
	36		22	4.50	3.5	171,185	197,199	171,173	132,136	150,152	287,291
Mean ± S.D.			27 ± 7	5.3 ± 0.9	3.1 ± 0.4						

Table 3.5 Relatedness estimates between individual *Petrogale xanthopus celeris* within each release colony (M - male; F - female).

Colony ID	Individual ID	Relatedness between individuals within groups							Group Average
<u>Colony 1</u>		M9	M12	F13	F25	F30	M31	F35	
	M9	-							
	M12	- 0.397	-						
	F13	- 0.473	-0.204	-					
	F25	0.088	0.069	0.110	-				
	F30	-0.542	0.151	0.045	- 0.226	-			
	M31	- 0.313	- 0.357	0.524	0.160	0.114	-		
	F35	- 0.219	0.102	0.156	0.409	0.060	0.216	-	
	F43	- 0.477	0.187	0.283	- 0.180	0.327	0.172	-0.436	- 0.023 ± 0.295
<u>Colony 2</u>		M2	F6	F15	M19	M24	F29	F32	
	M2	-							
	F6	0.191	-						
	F15	- 0.021	- 0.122	-					
	M19	0.103	0.043	0.499	-				
	M24	0.212	0.207	- 0.355	- 0.144	-			
	F29	- 0.253	0.207	- 0.355	- 0.144	0.109	-		
	F32	0.167	0.146	0.695	0.543	- 0.141	-0.464	-	
	F41	0.423	0.002	0.660	0.352	- 0.274	-0.394	0.369	0.081 ± 0.322
<u>Colony 3</u>		F5	F14	F17	M22	F27	M28	M34	
	F5	-							
	F14	- 0.418	-						
	F17	0.215	- 0.261	-					
	M22	- 0.355	0.204	- 0.354	-				
	F27	0.114	- 0.424	- 0.236	0.000	-			
	M28	0.142	-0.103	0.465	- 0.551	- 0.204	-		
	M34	0.326	- 0.360	0.226	- 0.288	0.101	0.284	-	
	F36	0.283	- 0.261	- 0.260	- 0.366	0.023	- 0.221	- 0.155	- 0.087 ± 0.279

wild (Stanely Price 1989). The species was known to historically occur at the release site, with suitable, relatively undisturbed habitat still present. Extensive exotic predator control had been undertaken for six months before the release and zero counts of *V. vulpes* and *F. catus* were recorded immediately prior to release. Above-average rainfall in the months before reintroduction ensured high floristic abundance, diversity and water content in plants. Thus the likelihood of survival was high, and consequently the reintroduction was deemed ethical.

In reference to criteria listed in 3.2.1.8, *P. x. celeris* occurs in an arid-zone rocky environment; lives in cohesive groups (although generally solitary); is a large mammal by marsupial standards; is an explorer species (as will be evident in 6.3); and is predominantly nocturnal. It also breeds at an early age (12 months); is a generalist herbivore; has been well studied in the wild (e.g. Nicholls 1972; Lim 1987; Lim *et al.* 1987; Sharp 1997a); is genetically diverse (Eldridge 1997c); is not genetically adapted to captivity; and avoids inbreeding (Pope *et al.* 1996). For the purposes of this project, the species has historically declined but is not endangered (IUCN 2000); attracts funding and support (Chapter 4); breeds well in captivity; is robust and not highly prone to myopathy; is habitat specific; and is highly unlikely to excavate and consume 1080 poison baits due to its herbivorous diet. Thus *P. x. celeris* is an ideal candidate species for experimental reintroduction, possibly more so than any other Australian marsupial.

3.3.2 Founder preparation

3.3.2.1 Modelling release

Because of the absence of post-release monitoring in the past, no suitable data were available to model the reintroduction of *P. x. celeris*. The suitability and carrying capacity of the release sites was however determined (2.3.4), and limiting factors of the reintroduced population were addressed (2.3.9). Because the species is not endangered, the project was unlikely to cause long-term demographic or genetic detriment to the captive population, and the potential knowledge to be gained from the release, modelling was not deemed essential.

3.3.2.2 Pre-release training

Of the six forms of pre-release training identified by Kleiman (1989), only one was considered necessary for *P. x. celeris*. Predator avoidance training was not attempted because of its previously high failure rate and the extensive resources required (McLean *et al.* 1996; Langford and Burbidge 2001). Lapidge (2000) indicated that captive bred *P. x. xanthopus* readily adopted an appropriate diet on release without training and thus diet training was unnecessary. However, some vegetation (*Abutilon*, *Ptilotus*, *Acacia* and *Eremophila* spp.) was collected from Lambert Station and offered in the release colony enclosures before release; it was quickly consumed. Spatial orientation or locomotion training was not required because of the presence of suitable rock piles in the enclosures. Navigation for migration and nest building training is not applicable to the species. Thus social training was the only form undertaken.

Constructed release colonies were housed separate from other colonies. Pre-colony formation (the state of the population prior to manipulation) and post-colony formation baseline data indicated a reduction in intraspecific aggression among males (33% to 11%). The coat of one male (M28, Fig. 3.2) had alopecia, but the animal was not replaced. Significant decreases in haematological values were detected using single-factor ANOVA ($P < 0.05$) in pre- and post-colony formation red blood cell (RBC) counts for Groups 1 and 2 and in haemoglobin for Group 1, while significant increases in mean corpuscular volume (MCV) for Groups 1 and 2 and platelets for Group 3 were recorded. The reduction in RBC count was probably due to a reduction in dietary iron intake (D. Griffin pers. comm.). Most animals were housed in enclosures containing rock piles pre-colony formation and were observed to drink water after rain from pools formed in the sedimentary rocks which are high in iron (Nedler 1984). Water runoff from rocks would also likely increase iron levels in surrounding vegetation. Groups 1 and 2 were housed in enclosures without rock piles for two months before release (not usually used for *P. x. celeris*), while Group 3 and the remaining animals were housed in enclosures with rock piles. Although lower RBC counts can indicate stress, lower creatinine values recorded do not (Table 3.3), and no significant mass changes occurred in either group. Furthermore, RBC counts were similar to wild *P. assimilis* (Spencer and Speare 1992). The increased MCV values are the direct result of lower RBC values; MCV equals packed cell volume divided by the RBC count. Changes in haemoglobin and platelets are also

directly related to changes in RBC count (Feldman *et al.* 2000). The only significant changes in biochemical values were for protein and creatinine in Group 2 (Table 3.3). The low protein level is likely due to Group 2 having four juveniles under the age of 16 months (Table 3.4), which naturally have lower protein levels (Kaneko 1997), thus reducing the average protein concentration. The reduction in creatinine is possibly due to a reduction in physical exertion (Finco 1997) during capture, as animals became accustomed to the technique. A general reduction occurred in all groups, but only one was significant (Table 3.3). Thus none of the changes in blood values was deemed detrimental, and constructed release groups were not changed.

Animals were captured using hanging nets in captivity, but this technique was not suitable for post-release monitoring. To ensure a reasonable recapture rate post-release, treadle-operated box traps to be used in the field were wired open and placed in each of the release colony enclosures four months before release. Traps were baited with food and water and replenished regularly. Animals became accustomed to the traps with many using them as shelter (Fig. 3.3). A habit of bending the trap door down through jumping on it developed in all colonies. Consequently traps were inverted (Fig. 3.3) to avoid further damage. Traps were not removed until after the release, at which time they were transported to the site and placed with the same colony that had previously occupied them in captivity, so that scents would be familiar. The habit of bending doors down continued in the wild.

Figure 3.3 Trap-trained and radio-collared *Petrogale xanthopus celeris* before release.



3.3.2.3 Veterinary preparation

S. Conaghty, Adelaide Zoological Gardens veterinarian, was employed to undertake independent veterinary screening and preparation of animals for release. All release animals were anaesthetised using Isoflurane (Fig. 3.4) by open mask and examined for mass gain/loss, body and coat condition, ecto-parasites, eyes, ears, teeth and gum margins, foot pads, abdominal palpation and chest auscultation. Blood samples were taken for haematological and biochemical analysis. Fresh faecal samples were collected and examined for *Eimeria*, strongyles and oocysts. Skin scrapings were collected and examined for mites.

No significant changes in body mass were recorded in any group (ANOVA $P < 0.05$). Body condition of individual release animals is shown in Table 3.4. Pre-colony condition scoring was not undertaken. Ecto-parasites were observed on 29% (7/24) of animals before release, but they are known to be common on captive and wild macropods (Speare *et al.* 1994; Conaghty and Schultz 1998). The eyes of one animal (F13) showed a pinpoint cataract in the right anterior lens, but this was deemed not to disadvantage the animal. Three animals (F5, M12 and F32) showed evidence of pinna deformity, most likely through previous tagging (Pope *et al.* 1996). Ears were cleaned and animals deemed suitable for release. Teeth and gum margins of all animals were good with no evidence of lumpy jaw. Foot pads were good, with only F29 showing evidence of wear. Abdominal palpation and chest auscultation examination indicated no problems besides bloated abdomens in F6 and F35, and slight crackling in the left lung of F6. However, both animals were considered releaseable. *Eimeria spp.* were found in 29% (2/7), strongyles in 43% (3/7) and oocysts in 57% (4/7) of animals sampled. However, all are common in both captive and wild *P. xanthopus* (Conaghty and Schultz 1998). All skin scrapings collected (n=24) returned negative results for mites. Thus no diseases were encountered in the captive *P. x. celeris* population that did not occur at the release site. Vaccinations against possible disease at the release site, such as Lumpy Jaw, Poxvirus, Herpesvirus, Wallal virus or *Toxoplasma gondii* were not available.

Reproductive status of each female was examined, with 73% (11/15) carrying pouch young. Two large joeys from F30 and F43 were thrown and hand-raised, five smaller joeys (1 :4 , 110±9 days old, dams 5, 6, 14, 17 and 29) were thrown and euthanased

with Lethobarb as they were too young to hand-raise. Four females (F13, F15, F27 and F36) were released with small foetuses in pouch, three less than 21 days of age and one 56 days (in F36). Trapping five months after release indicated the younger foetuses had survived but the older female foetus was aborted (e.g. not carried full term) at the time of release.

Figure 3.4 Anaesthetised *Petrogale xanthopus celeris* for final veterinary screening and radio-collaring.



3.3.2.4 Radio-collaring and tagging

All captive *P. x. celeris* had been previously ear numbered during the first captive survey in April 1998. Radio-collars (Titley Electronics Microlite two-stage collars, 70g mass, mortality sensing, 18 month battery life) were fitted in August 1998, one week before release, under general anaesthesia (Fig. 3.4). The model of the collar had previously been tested at Adelaide Zoological Gardens for six months and successfully used with reintroduced *P. x. xanthopus*. Large central ear holes were made using a leather punch while under anaesthesia to allow time to heal prior to tagging. This technique is reported to result in lower rates of post-tagging infection (S. Conaghty pers. comm.). All release animals were ear-tagged with 3 cm (diameter) microprism retroreflective ear-tags (Allflex tags, modified by Reflex Promotions) immediately before being loaded into transport boxes for release. The same ear-tags had previously been used successfully with the smaller-eared *P. penicillata* (C. Rummery pers. comm.). Males were tagged in their right ear with a single colour and females were tagged in the left ear with a double or split colour. Each animal at each site carried a different coloured ear tag, which made them instantly

recognizable at approximately 50 m during daylight hours and 100 m at night using spotlights, or further with the aid of binoculars (refer to title page for picture). Air holes in the transport boxes were made smaller than the tags to prevent injury. No ear-tags were lost throughout the project.

3.3.2.5 Transport

Animals were transported to the Lambert Station release site in a two-hour drive. Each animal was confined to a 50 x 50 x 50 cm cardboard box with 20 small air holes. Boxes were thickly lined with shredded paper to provide cushioning. Four Toyota Landcruiser station-wagons containing six animals/boxes in each were used to transport the 24 animals and eight staff to the release site. A minimum of eight staff was required to carry the eight animals/boxes to the base of each release site, thus avoiding delays in release. Collection of animals and associated ear tagging commenced at 15.00 h on the day of release (August 9, 1998) and finished at 17.00 h. Transport to the site was between 17.00 h and 19.00 h. Animals were released at Site 3 at 19.00 h, Site 2 at 19.30 h and Site 1 at 20.00 h (Fig. 2.3), with sun set occurring at 18.30 h. Thus, animals were in transit for 3 - 4 hours on average and anaesthesia was not deemed necessary. No animals showed evidence of stress upon release.

3.3.2.6 Release type

Direct release with no provisioning was used. The short distance the animals had travelled and the similarity in climate between Charleville and Lambert supported no acclimatization. Provisioning at the release sites was not used for two reasons. First, *P. xanthopus* has been found to readily adopt an appropriate diet in the wild upon release (Lapidge 2000). Second, the source or area of provisioning (feed hopper, water trough) can become a target for predators.

Once at each release site, all animals were carried to the foot of the hill below the most suitable habitat and rock outcrop. Boxes were arranged in an arc and opened simultaneously. All animals hopped off into the rock piles at the base of the respective hills, although some stopped to forage on Silver-tails (*Ptilotus obovatus*) on the way. Once

all animals were on the hill the site was vacated until the following day to allow them to explore undisturbed.

3.3.2.7 Intervention

On the basis of the experimental nature of the reintroduction, the non-endangered status of *P. x. celeris*, a suitable large captive population remaining, and the healthy status of the animals before release, the decision was made that intervention would not occur after reintroduction as long as the fate of most animals released was known and factors causing mortality were continually addressed, such as by intensifying predator control measures.

3.4 *Petrogale xanthopus xanthopus* in South Australia

3.4.1 Founder selection

Founder selection and preparation for the reintroduction of *P. x. xanthopus* to Aroona Sanctuary was undertaken by staff of the Royal Zoological Society of South Australia (RZSSA) at Adelaide and Monarto Zoological Parks. Not all details are available to the author and the following information relates to published reports (Conaghty and Schultz 1998; Barlow 1999).

3.4.1.1 Knowledge of species

Previous research on *P. xanthopus* has been documented in 3.3.1.1. The only published known difference between the sub-species is in home range, with that of *P. x. xanthopus* being dramatically larger (2.2.4) possibly due to their more arid habitat in South Australia (Lim 1987; Sharp 1994).

3.4.1.2 Self-sustaining captive population

The captive population of *P. x. xanthopus* was established at Adelaide Zoological Gardens in 1883 from 14 animals from Sliding Rock, 23 km east of Beltana in the northern Flinders Ranges (Fig. 2.4) (Hornsby 1980). A further four *P. x. xanthopus* were added in

1893, eight by 1900, 11 by 1915 and four between 1938 and 1975. Later donations were from various colonies throughout the Flinders Ranges. Between 1963 and 1971, 52 *P. x. xanthopus* were sent to other institutions including Adelaide University, Cleland Conservation Park, Penola Station Fauna Sanctuary and private collectors in South Australia, Taronga Zoo and the Australian Reptile Park in New South Wales, Melbourne Zoo and Healsville Sanctuary in Victoria and the Commonwealth Industrial and Scientific Research Organization (CSIRO), Division of Wildlife Research in Canberra. After a peak of 60 animals in 1967, the Adelaide Zoo colony was reduced to a single pair by 1975 through dispositions and high mortality (Hornsby 1980), possibly indicating a genetic bottleneck. Since 1975, numerous wild *P. x. xanthopus* particularly males, from various colonies throughout the Flinders Ranges have been periodically captured and brought into captivity for genetic purposes (S. Barlow pers. comm.). Stock records for the species began in 1942 and are currently maintained through the SPARKS (Single Population Analysis and Records Keeping System) database. By 1998 the captive *P. x. xanthopus* population numbered 136 animals in 11 Australian institutions (Conaghty and Schultz 1998), of which 23 (8 : 15) were kept at Adelaide Zoo and 72 (46 : 26) at the RZSSA's open range zoo, Monarto Zoological Park (MZP). Annual surveys of captive *P. x. xanthopus* held by the RZSSA are now undertaken, with information being updated on SPARKS.

3.2.1.3 Animal management and suitability

Due to the collapse of the captive *P. x. xanthopus* population in 1975 and its later resurgence through the use of wild stock, the population had effectively been in captivity for only 20 years (or 13 generations) at the commencement of the reintroduction project in 1995. *P. x. xanthopus* at MZP were selected as the most appropriate release stock because they were housed in large natural enclosures, received minimal supplementary food (macropod pellets: Ridley Agriproducts), were off-exhibit and therefore unaccustomed to humans, were exposed to natural predators, particularly Wedge-tailed Eagles (*Aquila audax*), and because the size of the population would maximise selection of suitable release animals (Barlow 1999). The species at MZP is actively managed, with breeding males selected using SPARKS for harem colonies. Excess males are housed in bachelor colonies. Despite being 500 km south, the climate at MZP is similar to that at the release

site, and no captive population occurs closer. Thus the MZP *P. x. xanthopus* population was the most suitable for reintroduction.

3.4.1.4 Husbandry

Husbandry methods used at MZP were also used by the author in Queensland (See 3.3.1.4 for details).

3.4.1.5 Genetic suitability

The source of the original founding population of *P. x. xanthopus* held by the RZSSA was Sliding Rock, 30 km from the Aroona Sanctuary release site. Later additions to the colony originated from various locations throughout the Flinders Ranges, however most were from the central ranges or further north. Consequently the population should be genetically similar to other colonies in the area and suitable for reintroduction. Although not determined, periodic additions of wild *P. x. xanthopus* to the RZSSA colony should have increased genetic diversity (Frankham 1994, 1995). The removal of 10 animals from the combined *P. x. xanthopus* colony of 95 animals held by the RZSSA was deemed unlikely to be to the genetic detriment of the species, and detailed baseline genetics were not established to the author's knowledge. However, the existence of a studbook allowed for the selection of animals with diverse parentage, therefore avoiding removal of specific genetic strains.

3.4.1.6 Baseline data

Extensive baseline data were established by the RZSSA Veterinary Department before the release of *P. x. xanthopus* during 1995 and 1996. Animals were aged during annual January surveys of the captive population, thus an animal born at any time in that year was assigned a birth date of January of that year (Table 3.8). All captive animals aged between two and five years were anaesthetised with Isoflurane by open mask, and blood, skin and faecal samples collected and examined for haematology, biochemistry and microorganisms to establish a baseline for captive *P. x. xanthopus* (Tables 3.6 and 3.7).

Particular attention was paid to tooth/gum margins because of the common incidence of Lumpy Jaw in captive macropods (Conaghty and Schultz 1998). No inflammatory or organ problems were detected in blood profiles (Table 3.6). Wild *P. x. xanthopus* previously brought into captivity, and wild Euros (*Macropus robustus erubescens*), a sympatric species of *P. xanthopus* were also examined for pathogenic microorganisms (Table 3.7). Body masses, condition scores, pelage and ectoparasite burdens were used as indicators of general health. Fecundity and mortality records were obtained from, and updated to, SPARKS during initial surveys.

Limited sampling (n=4) of wild macropods for vitamin E (alpha-tocopherol) concentration indicated a higher mean (9.27 mg/L) than for captive *P. x. xanthopus* (2.06 mg/L), thus 0.02-0.025 ml/kg of vitamin-E/selenium was administered intramuscularly whilst handling. Physiological stress of capturing and handling was monitored through the lymphocyte: neutrophil ratio, with no evidence of a stress leucogram in the majority of *P. x. xanthopus*. Individuals occasionally showed marked elevations in creatinine but no overt cases of myopathy resulted (Conaghty and Schultz 1998).

3.4.1.7 Founder selection

Two males and eight females were chosen for the initial reintroduction group (Table 3.8). Two groups of potential release animals were selected and placed under minimal surveillance in a naturalistic enclosure six months prior to release, with no supplementary food, and exposed to natural predators (*A. audax*). All animals selected had previously reproduced and were between the ages of two and three years. Veterinary screening one month before release indicated few health differences between potential release animals, thus females were selected on previous reproductive and rearing success. The two males were selected on the basis of not fathering current diapause embryos, in order to maximise levels of genetic diversity. A further two females underwent the same selection process and were released a year after the initial group to replace two animals that died in the first 12 months.

Table 3.6 Clinical pathology profile for normal captive *Petrogale xanthopus* held by the Royal Zoological Society of South Australia. Includes samples with slight haemolysis and elimination of outlying values. Compiled on MEDARKS and ISIS by S. Conaghty.

Clinical pathology	Units	Mean \pm S.D.	Range	(N)
RBC	*10 ¹² /L	6.132 \pm 0.345	5.27 - 7.01	73
Hb	GM/L	176 \pm 8.8	157 - 196	71
PCV	%	54.46 \pm 3.24	48.3 - 62.7	71
MCH	*10 ⁹ /L	28.57 \pm 1.08	23.54 - 30.97	73
MCHC	GM/L	324 \pm 9.5	301 - 349	73
MCV	fL	88.29 \pm 3.72	71.75 - 96.46	73
WBC	*10 ⁹ /L	6.506 \pm 1.543	3.7 - 9.9	72
Neutrophils	*10 ⁹ /L	2.119 \pm 1.394	0.225 - 6.106	73
Bands	*10 ⁹ /L	0.0 \pm 0.0	0.0 - 0.0	72
Lymphocytes	*10 ⁹ /L	4.115 \pm 1.541	1.08 - 7.332	72
Monocytes	*10 ⁹ /L	0.043 \pm 0.066	0.0 - 0.34	71
Eosinophils	*10 ⁹ /L	0.183 \pm 0.161	0.00 - 0.552	72
Basophils	*10 ⁹ /L	0.009 \pm 0.023	0.0 - 0.08	71
NRBC	/100	1.329 \pm 2.357	0.0 - 12	70
Platelets	*10 ¹² /L	0.12 \pm 0.039	0.035 - 0.228	65
Glucose	mmol/L	5.3 \pm 3.02	0.0 - 12	79
BUN	mmol/L	11.9 \pm 1.86	7.78 - 16.6	79
Creatinine	umol/L	122 \pm 21.7	80 - 180	79
Calcium	mmol/L	2.77 \pm 0.208	2.16 - 3.2	78
Phosphorus	mmol/L	1.84 \pm 0.542	1.03 - 3.91	77
Sodium	mmol/L	148.3 \pm 4.4	136 - 157	78
Potassium	mmol/L	6.291 \pm 1.444	3.7 - 14.2	75
Na/K ratio		24.01 \pm 5.61	7.7 - 39.5	77
Chloride	mmol/L	98.36 \pm 2.92	90 - 107	78
Bicarbonate	mmol/L	11.92 \pm 6.72	3 - 30	79
Cholesterol	mmol/L	3.55 \pm 1.03	2.2 - 4.99	8
Anion Gap		44.74 \pm 9.43	20 - 60	78
AST	mmol/L	109.5 \pm 36.9	40 - 256	78
ALT	mmol/L	56.75 \pm 14.99	31 - 75	8
Total Bilirubin	mmol/L	0.376 \pm 0.633	0.0 - 2	77
ALP	mmol/L	550.3 \pm 212.6	298 - 890	8
LD	mmol/L	675.8 \pm 239.4	274 - 1403	77
CK	mmol/L	1808 \pm 1922	351 - 13970	106
GGT	U/L	21.62 \pm 11.76	0.0 - 52	79
Total Protein (C)	GM/L	67.1 \pm 6.37	54 - 82	78
Albumin (C)	GM/L	52.2 \pm 5.16	37 - 66	79
Globulin (C)	GM/L	15.3 \pm 4.35	8 - 28	79
Total Protein (E)	GM/L	68.86 \pm 1.078	41.4 - 84	13
A/G ratio		0.87 \pm 0.12	0.66 - 1.07	13
Albumin (E)	GM/L	32 \pm 5.7	17 - 41.3	13
Alpha 1 (E)	GM/L	7.11 \pm 1.78	4.34 - 9.63	13
Alpha 2 (E)	GM/L	5.67 \pm 2.23	2.38 - 9.03	13
Beta (E)	GM/L	16.2 \pm 5.2	9.86 - 27.3	13
Gamma (E)	GM/L	8.3 \pm 3.09	5.13 - 14.6	13
Vitamin E	MG/L	2.06 \pm 2.523	0.1 - 8.43	69
Selenium	umol/L	0.876 \pm 0.5	0.32 - 2.73	45
Zinc	umol/L	11.92 \pm 1.89	9 - 16	13
Copper	umol/L	3.9 \pm 1.87	2 - 12	36

Table 3.7 Selection of microorganisms found in captive and wild *Petrogale xanthopus xanthopus* and wild Euros, *Macropus robustus erubescens*, prior to the reintroduction of *P. x. xanthopus* to Aroona Sanctuary, South Australia (Conaghty and Schultz 1998).

MICROORGANISM	CAPTIVE	YFRW	WILD YFRW	WILD EURO
Strongyles	74	(39)	100 (10)	100 (5)
<i>Eimeria</i> spp.	73	(41)	30 (10)	60 (5)
<i>Salmonella</i> sp.	0	(6)	100 (6)	100 (3)
<i>S. infantis</i>	36	(11)		
<i>S. birkenhead</i>			100 (1)	
<i>Enterococcus</i> sp.	73	(15)		
<i>Pasteurella</i> sp.			33 (3)	
<i>E. coli</i> (>1 type)	100	(17)	100 (12)	100 (5)
Haemolytic <i>E. coli</i>	38	(8)		
Non-haem. <i>E. coli</i>	91	(11)		
<i>Serratia liquifaciens</i>	100	(4)	100 (2)	
<i>Proteus</i> sp.	100	(5)	100 (1)	
<i>P. mirabilis</i>	100	(11)		
<i>Bacillus</i> sp.	100	(12)	100 (11)	100 (5)
<i>Corynebacterium</i> sp.	100	(4)		
<i>Klebsiella ozaenae</i>	100	(11)	100 (2)	100 (2)
<i>K. Aeroginosa</i>			33 (3)	
<i>Pseudomonas fluorescens</i>	100	(7)		
<i>Streptococcus acidominus</i>	100	(7)		
& haemolytic <i>Streptococcus</i> sp.	100	(1)	73 (11)	100 (5)
Fungi	0	(2)	0 (3)	0
<i>Penicillium</i> sp.			100 (6)	100 (3)
<i>Candida</i> sp. (not <i>C. albicans</i>)	100	(2)	100 (1)	
<i>Aspergillus</i> sp.			100 (2)	
<i>A. niger</i>			100 (6)	100 (3)
<i>Mucor</i> sp.	100	(7)		
<i>Geotrichum</i> sp.	100	(4)		
Macropod Herpesvirus	0	(38)	0 (3)	100 (2)
Wallal virus	0	(38)	0 (3)	66 (3)
<i>Toxoplasma gondii</i>	0	(54)	0 (3)	0

Table 3.8 Founding population of *Petrogale xanthopus xanthopus* at Aroona Sanctuary, South Australia, obtained from SPARKS. * Held back from main release due to cut on nose.

Animal	Sex	Studbook No.	~Birth date	~Age (m)	Mass (kg)	Released
1		1247	Jan-94	32	-	26-Sep-96
2		1250	Jan-94	32	-	26-Sep-96
3		1259	Jan-94	32	5.80	26-Sep-96
4		1282	Nov-93	34	6.00	26-Sep-96
5		1279	Jul-94	26	6.00	26-Sep-96
6		1237	Jan-94	32	6.50	26-Sep-96
7		1246	Jan-94	32	5.50	26-Sep-96
8		1285	Jan-95	20	5.20	26-Sep-96
9		1255	Jan-94	32	5.65	23-Oct-96*
10		1239	Jan-94	32	-	26-Sep-96
11		1223	Aug-94	25	-	22-Sep-97
12		1248	Jan-94	32	-	22-Sep-97
Mean	2.10			30± 4	5.8± 0.4	

3.4.1.8 Animal welfare

As with the release of *P. x. celeris*, the reintroduction of *P. x. xanthopus* occurred after extensive screening of each animal's health, preparation for reintroduction, above average rainfalls at the release site and the control or removal of limiting factors. Consequently the release was undertaken with the animal's welfare of paramount importance and individuals had the highest possible chance of survival.

3.4.2 Founder preparation

3.4.2.1 Modelling release

The reintroduction of *P. x. xanthopus* was not modeled, for the same reasons as those given for the later *P. x. celeris* release (3.3.2.1). Aroona Sanctuary was considered of sufficient size and carrying capacity to allow the population to expand greatly, and evidence exists that the species was once prolific in the area (Eyre 1841). Eradication of exotic vertebrates in the vicinity of Aroona Sanctuary (2.4.9) further increased the likelihood of reintroduction success (Short *et al.* 1992).

3.4.2.2 Pre-release training

Indirect dietary training was used with captive *P. x. xanthopus* through the reduction and removal of supplementary food in the months prior to release, which encouraged the animals to forage on grasses and forbs growing in the enclosure (Lapidge 2000). Indirect social and predator avoidance training occurred through natural exposure to *A. audax*. The affect of social training on colony cohesion was determined through blood profiles during final veterinary preparation one month before release. Specific profiles of release animals were not available for statistical testing and so it is not known intraspecific aggression occurred. Trap training was not conducted as no plans had been made to recapture the animals.

3.4.2.3 Veterinary preparation

P. x. xanthopus selected for release underwent veterinary examination one month prior to release, at which time animals were radio-collared and in-pouch joeys were removed and hand-reared or euthenased for reasons given in 3.2.2.3. There was a high correlation between organisms inhabiting captive and wild *P. x. xanthopus* (Table 3.7). Conaghty and Schultz (1998) decided not to eradicate any organisms in the release animals because of the unknown consequences to the immunity and microbiological balance of *P. x. xanthopus*. As previously stated (3.3.2.3), vaccinations are not available for most macropod diseases.

3.4.2.4 Radio-collaring and tagging

All captive *P. x. xanthopus* held by the RZSSA are microchipped, with individual records kept on SPARKS. The same model radio-collars (Titley Electronics) used for *P. x. celeris* were tested successfully for six months at Adelaide Zoological Gardens. Collars used on *P. x. xanthopus* differed slightly in that each had a uniquely coloured band and thus ear tagging was not used. Later collars fitted during the trapping project were all black, and ear-tagging was undertaken by the author to allow for more rapid and less ambiguous identification of animals in traps.

3.4.2.5 Transport

The 10 animals selected were crated on the morning of release after a final veterinary examination. One female (F9) had facial bruising and was kept back and released a month later. The nine remaining animals were housed individually in large pet-paks, loaded into a vehicle and driven six hours north to the Aroona Sanctuary release site. Animals were ferried over the dam using a small dingy and released at the base of Mt Aroona (Fig. 2.6) at dusk on September 26, 1996. The same procedure was undertaken for the later release of females 9, 11 and 12. All animals were in transit for 8-10 hours without anaesthesia. No animals showed evidence of stress upon arrival.

3.4.2.6 Release type

All animals were released directly to the wild and without provisioning. Released animals had been feeding on natural foods for some months before release with no mass loss. Fresh water was available from the dam, and therefore provisioning with water was unnecessary. Although specific reasons could not be found, it is the author's belief that direct release was chosen for the same reasons listed in 3.3.2.6.

3.4.2.7 Intervention

A veterinarian experienced in *P. x. xanthopus* pathology was located at the release site for 40 days post-release to determine the reason for any mortality. Whether a decision on intervention guidelines was made prior to release is unknown. As this was also a trial reintroduction, it is felt they were not for similar reasons stated in 3.3.2.7.

3.5 Discussion

Ultimately, the aim of founder selection and preparation is to select the combination of animals that will best survive and reproduce upon reintroduction, and to exclude those that may not. For reasons detailed in 3.3.1.8, *P. xanthopus* makes an ideal experimental reintroduction candidate species. It is the most appropriate *Petrogale* species because it is less endangered than the Prosperine Rock-wallaby (*P. persephone*), Brush-tailed Rock-

wallaby (*P. pencillata*) or Black-footed Rock-wallaby (*P. lateralis*) (Maxwell *et al.* 1996), it is the most widely studied in the wild (e.g. Copley 1983; Lim 1987; Lim *et al.* 1987; Copley 1990; Lim *et al.* 1992; Sharp 1994; Sharp 1997a), and it has the largest self-sustaining captive populations (c.170 animals in 15 institutions prior to reintroduction) (Slater 2001). The detailed selection and preparation criteria outlined in this chapter will hopefully provide a basis for the impending conservation reintroduction of *P. pencillata* into Victoria.

Although predation has been found to be the major cause of failure in macropod reintroductions in the past (Short *et al.* 1992), and calls have been made to incorporate training into reintroduction programs (Griffin *et al.* 2000), no training was attempted for either *P. x. celeris* or *P. x. xanthopus*. Although previous attempts to train macropods have shown promise, animals rapidly forgot their training, and high mortality occurred upon release (Gibson *et al.* 1994; Underwood 1995). Therefore, in the present study, resources were devoted to exotic predator control at the site of release. No successful cases of predator avoidance training in reintroduced mammals could be found in the literature. There are probably two principal reasons. First, no training can prepare a captive-bred animal for the stealth that exotic predators use while hunting. Second, wild-born animals that have evolved with exotic predators for over 6,000 years in the case of the *Canis lupus dingi*, 200 years with *F. catus* and 130 years with *V. vulpes* have still been wiped out by exotic predators (Kinneer *et al.* 1984 and 1988; Flannery 1994; Sharp 1994; Hornsby 1997). This is partly due to the unnaturally high densities of predators that have been maintained by unnaturally high populations of exotic prey species such as the rabbits and cattle.

The major difference between reintroduced *P. x. celeris* in Queensland and *P. x. xanthopus* in South Australia was in the diversity of the age and size of the animals. The average age of *P. x. celeris* was 27 months (range 14 – 51) and average mass was 5.20 kg (range 4.00 - 8.25 kg) (Table 3.4). This compares to an average of 30 months (20 - 34 months) and an average mass of 5.8 kg (5.20 - 6.50 kg) for *P. x. xanthopus*. The RZSSA selected *P. x. xanthopus* between two and three years of age as the animals most likely to survive, form individual home ranges and breed (Barlow 1999). This was achievable due to the high number of animals in the RZSSA colony. In Queensland however, more natural colonies were formed in terms of age and mass. The survival and adaptation of the different

release animals, and the identification of an optimum release age for the species are discussed in Chapters 6 and 7. Table 3.9 summarizes and compares founder selection and preparation criteria used by the author and RZSSA for the reintroductions of the two sub-species. Many techniques used are similar, mainly because the author adopted successful techniques developed by the RZSSA for the species.

Table 3.9 Comparison of founder selection and preparation criteria used for *Petrogale xanthopus celeris* in Queensland and *Petrogale xanthopus xanthopus* in South Australia.

Founders	<i>P. x. celeris</i> Queensland	<i>P. x. xanthopus</i> South Australia
Selection criteria		
1. Knowledge of species (published authors)	Sharp, Lim, Pope	Lim, Copley, Robinson
2. Self-sustaining captive population	41 in 1 institution	136 in 11 institutions
3. Animal management and suitability	Indirect, self- Yes	Direct, studbook-Yes
4. Husbandry	Same for both species- low stress	
5. Genetic suitability	High diversity	Diverse founders
6. Establish baseline data	All captive animals surveyed	
7. Founder selection	3 colonies 3 :5	1 colony of 2 :10
8. Animal welfare	Optimal animals & conditions	
Preparation criteria		
1. Modeling release	Neither directly	
2. Pre-release training	Indirect trap, social, diet	Indirect social & diet
3. Veterinarian preparation	Extensive for both, no vaccinations	
4. Radio-collaring and tagging	Collared & ear tagged	Collared & microchip
5. Transport	Road- 2 hours	Road- 6 hours
6. Release type	Direct and without supplementation	
7. Intervention	No- experimental release	No- trial release

RESOURCES

4.1 Introduction

Mainland macropod reintroductions have a success rate of only 11% (Short *et al.* 1992). The principal reason for this is that wildlife managers have failed to control exotic species, use modern technology or manage remote projects from their research bases (Short *et al.* 1992). In essence, they have failed to secure adequate resources for the project, either for site or animal selection and preparation or post-release monitoring, and hence under IUCN (1998) guidelines such projects should not have proceeded. A critical review of resources available for the life of the project is thus essential if reintroduction biologists are to learn from their mistakes and to advance this immature field of wildlife conservation to a point where endangered species can be reintroduced with some confidence.

A controlled release of a species to the wild provides the ideal opportunity to learn about the animal's ecological requirements, as well as to test ecological models under natural conditions (Krebs 1978; Griffith *et al.* 1989; Kleiman 1989; Stanley Price 1989; May 1991; Short *et al.* 1992; Armstrong *et al.* 1994b; Kleiman *et al.* 1994; Serena and Williams 1994; Southgate 1994; Sarrazin and Barbault 1996; Van Dierendonck & Wallis De Vries 1996; Wolf *et al.* 1996; Sinclair *et al.* 1998; Miller *et al.* 1999; Fischer and Lindermayer 2000; Lapidge 2000). High mortality is not necessarily a failure, as long as the reason/s for it have been identified and addressed in subsequent releases (Armstrong *et al.* 1994; Serena and Williams 1994; Sarrazin and Barbault 1996; Miller *et al.* 1999). Resources, including finances, staff, expertise, time and equipment are required to collect baseline data, monitor the animals post-release and to analyse and publish the results. Resources should thus be factored into initial project outlines, and secured prior to any work commencing on the site or the animals. In my opinion, if resource criteria cannot be met the

reintroduction project should be aborted before another opportunity to advance this field is lost.

4.2 Resources criteria

4.2.1 Funding

Long-term funding is significantly correlated with success of a reintroduction (Beck *et al.* 1994). Reintroduction is an expensive and often non-cost-effective method of wildlife conservation (MacKinnon and MacKinnon 1991; Lindburg 1992; Sunquist 1993). Planning for a reintroduction should consider finances at the outset. Funding is required for:

1. staff Salary, insurance and consultants;
2. education, including staff (training) and community (advertising);
3. animal husbandry, food, veterinary care, disease surveillance, collection of baseline data, pathology;
4. genetic screening;
5. feasibility studies with surveys in historic range;
6. habitat purchase, protection, fencing and security;
7. habitat restoration through re-vegetation and/or feral animal control;
8. equipment including computers, telemetry or GPS transmitters and receivers, traps; and
9. fieldwork supplies, transport and infrastructure

all prior to the reintroduction. Furthermore, post-release monitoring and associated fieldwork costs will significantly increase after reintroduction, although this may be partly offset by reduced husbandry and habitat acquisition costs.

Few reintroduction projects have detailed published accounts of the costs involved. Those that have include USD\$17M total, or USD\$80,000 per Sea Otter, *Enhydra lutris*, rehabilitated and reintroduced after the Exxon Valdez oil spill (Estes 1998), USD\$6.7M over eight years for two Gray Wolf, *Canis lupus*, reintroductions in Yellowstone National Park and central Idaho (Bangs and Fritts 1996), USD\$10M for the Black-footed Ferret, *Mustela nigripes*, captive-breeding and reintroduction program (Dietz *et al.* 1994), USD\$25M total or USD\$1M per annum (Cohn 1993) for the Californian Condor,

Gymnogyps californianus, reintroduction program, and USD\$1.4M on a single failed Canadian Lynx, *Lynx canadensis*, reintroduction in Colorado (Kloor 1999). Kleiman *et al.* (1991) provided a detailed account of all costs associated with the reintroduction of Golden-lion Tamarins, *Leontopithecus rosalia*, in Brazil. Over a seven-year period, an estimated US\$1,083,005 was spent, not including in-kind support, averaging USD\$22,563 per surviving reintroduced *L. rosalia* (48/97 or 49%). Although many of these costs seem high, and often cause public outcry (Kloor 1999), when compared to the profits of business and industry that have caused the species' initial decline they are insignificant. For example, the first-quarter net profit for Exxon (United States) in 2000 was USD\$3.35 billion (Reuters 2000), and although no comparative figures can be obtained, the profits gained from land sales and agriculture that caused habitat destruction/land clearing and extermination of *C. lupus*, *M. nigripes*, *G. californianus* and *L. rosalia* probably far outweigh the costs of recovery programs.

4.2.2 Staff

Experienced and dedicated staff, or intellectual resources (Miller *et al.* 1999) are required for every aspect of a reintroduction project. It has been suggested that due to the long-lived and often stressful nature of reintroduction programs, staff changes are common (Snyder *et al.* 1996, Miller *et al.* 1999). This is counter-productive to projects as it inevitably results in a loss of expertise. Consequently the structure of the reintroduction committee should be organized with specific roles and responsibilities of each member defined to avoid conflicts over decision-making.

If experienced staff are not available, consultants should initially be used until such time regular staff have gained the knowledge and experience required to proceed with the project. Although some positions can be multi-faceted, most reintroductions require a project manager, financial controller, veterinarian, geneticist, animal keepers, sponsorship officer, education officer and field researchers. Ideally, all positions should be maintained throughout the project for consistency. Considerable time can be spent educating staff, particularly if the local community is involved, and this should be factored into time budgets (FAO 1986; Stanley Price 1989).

4.2.3 Expertise

Under IUCN (1998) guidelines (4a.ii), reintroduction projects should consult with persons having relevant expertise with previous reintroductions of the same or similar species. Factors relating to reintroduction success have been published by Griffith *et al* (1989), Short *et al* (1992), Wolf *et al* (1996), Miller *et al* (1999), Fischer and Linder Mayer (2000) and Short and Turner (2000) amongst others. However, specific knowledge on the species or genus for reintroduction is always preferable (Sarrazin and Barbault 1996).

Petrogale xanthopus has been relatively well studied in the wild, although few results have been published. Acknowledged ecological *P. xanthopus* experts include L. Lim (Ecological Consultant) (Lim *et al.* 1980; Lim 1987; Lim and Giles 1987; Lim *et al.* 1987; Lim *et al.* 1992); T. Robinson (Robinson *et al.* 1994), P. Copley (Copley 1983; Copley and Robinson 1983; Copley 1990; Copley and Alexander 1997), P. Alexander (Alexander and Copley 1999), M. Lethbridge* (South Australian Department of Environment and Heritage)(*and University of South Australia); G. Gordon (Gordon *et al.* 1978; Gordon *et al.* 1993), P. McRae (Queensland Environmental Protection Agency); A. Sharp (University of Queensland/ New South Wales National Parks and Wildlife Service) (Sharp 1994; Sharp 1997a), T. Dawson, B. Ellis (University of New South Wales) (Dawson and Ellis 1979) and I. Hume and C. Allen (University of Sydney), the last four focusing primarily on dietary research. R. Close (University of Western Sydney) (Close 1997), M. Eldridge (Macquarie University) (Eldridge 1997a, b, c; Eldridge and Close 1997; Eldridge and Kinnear 1999) and L. Pope (formerly University of Queensland) (Pope *et al.* 1996) are acknowledged *P. xanthopus* geneticists, and D. Schultz and S. Conaghty (Royal Zoological Society of South Australia) (Conaghty and Schultz 1998) are acknowledged *P. xanthopus* veterinarians.

No previous attempt had been made to reintroduce *P. x. xanthopus* prior to the RZSSA project. However, *P. penicillata* has been previously reintroduced, along with numerous introductions of *P. penicillata* and *P. lateralis* (see 3.2.1.1), however no expert could be identified due to the haphazard nature of the releases and the lack of post-release monitoring.

4.2.4 Time

Reintroduction is a time-consuming process, involving pre-release planning, preparation and monitoring, release, post-release monitoring and data collection, data analysis and publication. Often no consideration (or finances) is given to time for post-release monitoring through to publication of results. The result has been, unfortunately, the repetition of past mistakes (Short *et al.* 1992). My opinion, if sufficient time and staff are not available for all aspects of the project it should not proceed.

4.2.5 Equipment and technology

Like other reintroductions, those of macropods have often failed to take advantage of existing technology such as radiotelemetry (Short *et al.* 1992) and other monitoring techniques. Few published accounts exist of adaptation post release of marsupials (Ellis *et al.* 1990; Christensen and Burrows 1994; Lapidge 2000; Short and Turner 2000; Southgate *et al.* 2000; Langford and Burbidge 2001), or of other species (Miller and Ballard 1982; Fritts *et al.* 1984; Slough 1989; Beck *et al.* 1991; Rodriguez *et al.* 1995; Fitzsimmons *et al.* 1997; Forbes and Boyd 1997). Most accounts have been restricted to monitoring of animal movements through radio-telemetry. This technique provides useful but limited information on adaptation. Released animals and their progeny must be recaptured to obtain physiological and genetic samples, as well as for reattachment or initial attachment (progeny) of radio-transmitters. Probably because of the funds, staff, expertise, time and equipment required for such programs, studies on ecological, physiological and genetic adaptation of captive-bred animals to the wild are sparse.

An array of new technology in addition to traditional radio-telemetry exists for use in monitoring wild or reintroduced animal populations. Global Positioning System (GPS) transmitters offer a powerful alternative to radio transmitters, particularly for animals with large ranges in rugged habitats that restrict extensive radio-tracking, but their current accuracy, cost and size remain prohibitive (D. Titley, Titley Electronics pers. comm.). Remote sensing, thermal imaging, night-vision equipment and Geographical Information Systems (GIS) are other alternatives to locating animals, studying behaviour and identifying habitat preferences, though all were beyond the resources of this study or were not required (GIS) because *P. xanthopus* is a habitat-specific species.

4.2.6 Education

IUCN (1998) guidelines for reintroduction state “development of conservation education for long-term support; professional training of individuals involved in the long-term program; public relations through the mass media and local community; and involvement of local people in the program” are essential criteria of reintroduction projects. The reintroduction projects of *O. leucoryx* (Stanley Price 1989), *L. rosalia* (Kleiman 1989; Kleiman *et al.* 1986; Kleiman *et al.* 1989) and *E. f. przewalskii* (FAO 1986; Van Dierendonck & Wallis De Vries 1996) all expended considerable funds and time on personnel and community education in their respective countries of Oman, Brazil and Mongolia.

Community support, as discussed in Chapter 2, and knowledge are often different issues. Although a community may support the project, their previous actions may have led to the species' initial decline, and hence education is required to address limiting factors for the species. Alternatively, the community may already be educated about a species' decline, but will not change practices because of its effect on traditions or livelihood, and hence will not support the project. Such is often the case with large predator reintroductions, with education having only a limited impact (Breitenmoser 1998).

The ultimate aim of many reintroductions is to form a self-sustaining population that can be monitored and managed by the local community, hence reducing resources required by the organizing body (Stanley Price 1989; Kleiman 1989; Kleiman *et al.* 1989; Kleiman *et al.* 1986; FAO 1986; Van Dierendonck & Wallis De Vries 1996). With this aim in mind, education programs not only need to gain community support for the project, but also train local biologists in field and management techniques required for the continuation of monitoring. Kleiman (1989) further recommended evaluating community attitudes to determine changes that have occurred through education. Education should clearly address any economic incentives or deterrents that may result from the project. Economic incentives such as increased productivity or employment will often lead to the project being held in high esteem by the community. As was discussed in 2.4.10, the flow-on effects of education and incentives can be considerable.

4.2.7 Post-release monitoring

Resources for post-release monitoring should be considered during the initial planning and site selection for a reintroduction (IUCN 1998). Potential release sites may vary dramatically in the resources available at the site and thus the additional resources required to undertake effective post-release monitoring. The proximity of the site to the researchers' base and to the local community, the effective size of the local community, and the amount of infrastructure already present will determine the funds, staff, time, equipment and education required to undertake post-release monitoring. Effective community size refers to the number of people willing to be employed or to volunteer for the project, and will often be a small fraction of the local population. While requiring more in terms of education, large communities may not necessarily offer greater community acceptance or involvement.

4.2.8 Criteria for success

No specific criteria exist for judging the success of reintroductions. However, success can be directly related to the aim of establishing a viable, self-sustaining, free-ranging population in an area of former range (Griffith *et al.* 1989; IUCN 1998; Fischer and Lindenmayer 2000). Under IUCN (1998) guidelines, short- and long-term success indicators should be identified pre-release for each specific project in the context of agreed aims and objectives. Proposed success definitions include persistence of the reintroduced population for longer than five years (Short *et al.* 1992; Short and Turner 2000), breeding by the first wild-born generation (Sarrazin and Barbault 1996), a three-year breeding population with recruitment exceeding mortality (Sarrazin and Barbault 1996), and an unsupported wild population of more than 500 animals (Beck *et al.* 1994). It has also been suggested that failed reintroductions may also be regarded as successful if the fate of every animal and the reason for their demise is known, and results are analysed and published with future recommendations made (Armstrong *et al.* 1994b; Southgate 1994; Miller *et al.* 1999).

In the long-term, continuing reintroductions can only be considered a success at a given point in time (Seddon 1999). The decline to a non-viable population of reintroduced *O. leucoryx* after 20 years of 'success', a case often considered the benchmark for the field of

reintroduction biology, illustrates this (Gorman 1999). Seddon (1999) proposed that reintroductions comprise a sequence of three objectives: survival of the release animals, breeding by the release animals and their offspring, and persistence of the population without intervention. However, he also pointed out that persistence is a state and not a result, and is only determinable through long-term post-release monitoring that requires considerable resources.

4.2.9 Analysis and publication

There are an estimated 500 or more reintroductions in North America alone every year (Booth 1988), but the post-release monitoring results or less than 20 have been published. Most of these only refer to causes of death or to animal movements. This is possibly because failures in wildlife management are often not reported (McNab 1983). In addition, the long-term nature of animal reintroductions results in a slower rate of publication. IUCN guidelines (1998) specify that post-release activities should include evaluation of cost-effectiveness and success of reintroduction techniques, along with regular publication in scientific and popular literature. Unfortunately, the progress of even the best-known reintroductions is reported at irregular intervals. For example, no public report on the reintroduction of *L. rosalia* in Brazil has appeared in the last decade.

4.3 Resources in Queensland

4.3.1 Funding

The reintroduction of *P. x. celeris* to Lambert Station, southwestern Queensland was a project instigated by the author and Professor Ian Hume, and as such was underwritten by Professor Humes' financial resources. The Queensland Environmental Protection Agency (QEPA) contributed AUD\$8,000 p.a. to the project for three years as well as substantial in-kind assistance including captive animal maintenance, initial vehicle usage, predator baiting (including aerial and ground baiting by the QEPA and supply of processed baits) and loan of equipment (radio-tracking and trapping). Further donations of AUD\$2,000 p.a. over three years were obtained from the South Australian National Parks Foundation, AUD\$560 from Brian Slee (project volunteer), AUD\$500 from the Australian Geographic Society, and USD\$3,500 from Palm Beach Zoo at Dreher Park through the American Zoo Association Marsupial and Monotreme Taxon Advisory

Group. Total direct revenue raised was thus AUD\$38,100. Expenses relating to the reintroduction are detailed in Table 4.1.

4.3.2 Staff

The reintroduction of *P. x. celeris* was undertaken by the author with assistance from I. Hume, G. Lundie-Jenkins, P. McRae, C. Evenson and R. Griffiths (QEPA). A commitment of core staff and required finances was made for a minimum duration of three years, at which time QEPA would take over future monitoring (with periodic visits by the investigators), and the author would analyse and publish results. A commitment was further obtained that in-kind predator control, both by QEPA and the local community, would continue indefinitely.

Independent experts and staff volunteered various tasks for the project. S. Conaghty medically screened *P. x. celeris* pre-release for disease and general condition (under general anesthetic) and fitted initial radio-collars. M. Eldridge undertook captive and wild-born progeny genetic analysis. Exotic predator control was undertaken through shooting by S. Henshall (field macropod harvester and professional shooter), P. Bredhauer and S. Bredhauer. Thirty-three volunteers from Australia, North American, South America, Asia and Europe assisted in fieldwork at Lambert Station.

Table 4.1 Costs (AUD) associated with the reintroduction of *Petrogale xanthopus celeris* in Queensland.

Item	1998	1999	2000	2001	Total
Salary- stipend of author (all other was voluntary)	\$10,000	\$10,000	\$10,000	\$8,000	\$38,000
Transport- 4,000km per fieldtrip @ \$0.50km	\$12,000	\$12,000	\$8,000	\$2,000	\$34,000
Fieldwork supplies, accom. & consumables- \$500 wk	\$7,500	\$6,000	\$3,500	\$1,000	\$18,000
Radio equipment - collars, CBs and equipment services	\$9,300	\$200	\$1,100	-	\$10,600
Traps- foam, shade-cloth, welding	\$700	\$1,500	-	-	\$2,200
Notebook computer	\$4,100	-	\$300	-	\$4,400
Veterinarian for release- transport for S. Conaghty	\$500	-	-	-	\$500
Genetic analysis- M. Eldridge	In kind	-	-	\$500	\$500
Blood analysis- Sydney Uni Vet Pathology, RPA Hospital	\$1,000	\$750	\$750	\$300	\$2,800
Field Metabolic rate- O ¹⁸ , analysis, postage, supplies	-	-	\$13,000	-	\$13,000
Predator control- in-kind QEPA, S. Henshall	\$2,800	\$2,400	\$1,400	\$1,000	\$7,600
	\$47,900	\$32,850	\$38,050	\$12,800	\$131,600

At the time of the last trapping (January 2001) a population of 50 animals was recorded. Hence, the cost per surviving *P. x. celeris* (released or wild-born) is \$2,632.

4.3.3 Expertise

The author had previous experience of *P. xanthopus* reintroductions, having assisted on nearly every facet of the South Australian *P. xanthopus* release. Further training was undertaken in *P. xanthopus* husbandry and trapping with veterinarians D. Schultz and S. Conaghty, and ecologist M. Lethbridge. I. Hume has previous experience with wild and captive *P. x. celeris* and koala reintroductions. Collaborators G. Lundie-Jenkins has expertise in macropod reintroductions and predator control (Rufous Hare-wallaby, *Lagorchestes hirsutus* and Bridled Nailtail wallaby, *Onychogalea fraenata*), P. McRae in wild *P. x. celeris* ecology, husbandry and range, and R. Griffith with *P. x. celeris* husbandry. In addition, S. Conaghty and D. Schultz were the consulting veterinarians and *P. xanthopus* reintroduction biologist (S.C.), M. Eldridge the consulting geneticist, D. Griffin and M. Krockenburger (Sydney University Veterinary Anatomy and Pathology) the consulting biochemist and pathologist respectively, M. Evans (QEPA) the consulting ecologist on mapping radio-telemetry data and professional shooter S. Henshall for direct predator control.

4.3.4 Time

For the author, the time devoted to the project was a minimum of three years of fieldwork and six months of data analysis and publication. Core funding for the project was guaranteed for a minimum of three years by I. Hume and QEPA. A total of 40 weeks was spent in Queensland on fieldwork, including a two-week survey trip in November 1997, three two-week pre-release fieldtrips in February, April and June 1998, a seven-week release/post-release monitoring fieldtrip in August and September 1998, two-week post-release monitoring fieldtrips in November 1998, January, March, May, July, September, November 1999, January, April, July, and October 2000, and a three-week final fieldtrip in January 2001. The QEPA has agreed to undertake annual monitoring of the release and feral animal control indefinitely.

4.3.5 Equipment and technology

Equipment required to undertake the reintroduction of *P. x. celeris* was a healthy, viable population of the species in appropriate captive facilities, a 4WD vehicle, office space and supplies (computer) for the author, infrastructure and access to the release site, radio-telemetry equipment (transmitters and three receivers), exotic predator control supplies, 30 traps, fieldwork supplies, genetics and pathology laboratories, and veterinarian assistance. A 4WD vehicle, office space and supplies, fieldwork supplies, radio-telemetry transmitters and one receiver, 13 traps and associated costs for pathology and veterinary assistance were guaranteed by I. Hume, S. Conaghty and the University of Sydney. A suitable captive *P. x. celeris* population, exotic predator control supplies (and labour for QEPA baiting), 17 traps and two receivers were guaranteed by QEPA. Genetic facilities were supplied by M. Eldridge and Macquarie University, Sydney. Infrastructure and access to Lambert Station were guaranteed by the landowners.

4.3.6 Education

Due to the small but knowledgeable community at the site (the Bredhauer family and S. Henshall), no specific community education was required. The author however gave numerous local and national radio interviews promoting the reintroduction, and visits were undertaken to the meeting place of the local pastoral community (Scrubby Creek Gun Club) to discuss the species, the project and benefits of exotic predator control. All education of project volunteers in animal husbandry, predator control and radio-telemetry was undertaken by the author and S. Conaghty. Education in the surrounding towns was deemed unnecessary due to the remoteness of the release site (>150km from any town).

4.3.7 Post-release monitoring

Further to fulfilling site selection criteria (Chapter 2), Lambert Station was considered an excellent release site due to its available resources for post-release monitoring. The site can be easily accessed by road, the Bredhauer family and S. Henshall were supportive of the project, and the use of a house and workshop were guaranteed for the project's duration. Furthermore, the Bredhauer family and S. Henshall volunteered to undertake

feral animal control and grade a track for use in survey work (macropod and exotic predator) around one of the chosen release sites that had limited access.

4.3.8 Criteria for success

In accordance with published success criteria, aspects of the reintroduction of *P. x. celeris* would be considered a success if:

1. baseline data on captive *P. x. celeris* were established,
2. the majority of *P. x. celeris* released on Lambert Station survived,
3. the cause of any *P. x. celeris* deaths was determined,
4. breeding occurred by the release animals and their offspring,
5. insight into ecological, physiological and genetic adaptation of released *P. x. celeris* was obtained,
6. results were analysed and published in both thesis and journal format, and
7. a viable, free-ranging and self-sustaining meta-population of *P. x. celeris* was formed.

Although the seventh criterion can only be addressed in the longer-term, criteria 1 to 6 are assessed throughout the thesis.

4.3.9 Analysis and publication

The two aims of the *P. x. celeris* reintroduction were to gain insight into how captive-bred individuals adapt to the wild ecologically, physiologically and genetically, and to publish findings in readily available scientific journals. Regular analysis of data collected pre- or post-release occurred throughout the project and at the completion of fieldwork. One paper has been published in *Australia Mammalogy* (Appendix II), with at least six more planned. Ten presentations have been made to scientific bodies and two popular articles have also been published:

- Lapidge S. and Hume I. (1999) Towards reintroducing captive-bred Yellow-footed Rock-wallabies: Pre-release planning. *Australian Mammal Society Conference Proceedings*, Sydney: p. 43.
- Lapidge S. and Hume I. (2000) Adjusting to life on the outside: The reintroduced Yellow-footed Rock-wallaby story. *Australian Mammal Society Conference Proceedings*, Alice Springs: p. 23.

- Lapidge S. Eldridge M. & Hume I. (2000) All things considered: Genetic approaches to Rock-wallaby reintroduction. *2nd National Rock-wallaby Symposium*, Alice Springs.
- Lapidge S. J. (2000). Macropod reintroduction in Australia: Past and present results. *Periodic lecture to the Royal Zoological Society of South Australia*.
- Lapidge, S.J. and Hume I.D. (2001). Adjusting to life on the outside: The reintroduced Yellow-footed Rock-wallaby story. *15th Conference of the Society for Conservation Biology*, Hilo, Hawaii, USA, July 2001, pp 91.
- Lapidge, S.J. (2001). Adjusting to life on the outside: The reintroduced Yellow-footed Rock-wallaby story. *Centre for Reproduction of Endangered Species, San Diego Zoo*, San Diego, USA, August 2001.
- Lapidge, S.J. and Hume I.D. (2001). Reintroduction genetics of the Yellow-footed Rock-wallaby. *8th International Theriological Congress*, Sun City, South Africa, August 2001, pp 86-87.
- Lapidge, S.J. and Hume I.D. (2001). Ecological and physiological adaptation to the wild of reintroduced captive-bred Yellow-footed Rock-wallabies. *8th International Theriological Congress* Sun City, South Africa, August 2001, pp 86.
- Lapidge, S.J. (2001). Adjusting to life on the outside: The reintroduced Yellow-footed Rock-wallaby story. *Roger Williams Park Zoo*, Providence, Rhode Island, USA, August 2001.
- Lapidge, S.J. (2001). Adjusting to life on the outside: The reintroduced Yellow-footed Rock-wallaby story. *National Zoological Park/Smithsonian Institution*, Washington DC, USA, August 2001.
- Torr G. and Beaton E. (2000) Yellow-footed Rock-wallabies: between a rock and a hard place. *Australia Geographic* **60**: 88-101.
- Torr G. (2001) Wallabies don't need water. *Australia Geographic* **62**: 120.

4.4 Resources in South Australia

4.4.1 Funding

The reintroduction of *P. x. xanthopus* to the northern Flinders Ranges, South Australia was undertaken by the Royal Zoological Society of South Australia (RZSSA), NRG Flinders, and the South Australian Department of Environment and Heritage (DEH), with financial and in-kind (staff) assistance from the Zoological Parks Boards of New South Wales and Victoria, Northern Flinders Soil Conservation Board, Leigh Creek Area School, the National Heritage Trust-funded Aroona Catchment Biodiversity Enhancement Project, and Los Angeles Zoo. Local residents and the pastoral stations of Myrtle Springs, Leigh Creek, Puttapa, North Moolooloo and Beltana also assisted with in-kind support (exotic predator control labour). Exact figures cannot be obtained for the release, although a conservative estimate places direct costs at AUD\$150,000+ not including in-kind assistance and wages (A. Emmerich pers. comm.). At the time of the last trapping (January 2001); approximately 25 animals inhabited Aroona Sanctuary, equivalent to AUD\$6,000 per animal.

The trapping program undertaken as part of this project was supported by the Royal Zoological Society of South Australia (AUD\$7,000 p.a. stipend), NRG Flinders (AUD\$5,000 p.a. for field expenses), the Aroona Catchment Biodiversity Enhancement Project (Natural Heritage Trust) (AUD\$4,000 for radio-collars), and the Zoological Parks Board of New South Wales (AUD\$1,000), a total of AUD\$41,000. Expenses related to the project are detailed in Table 4.2.

Table 4.2 Costs (AUD) associated with retrapping *Petrogale xanthopus* at Aroona Sanctuary.

<u>Item</u>	<u>1998</u>	<u>1999</u>	<u>2000</u>	<u>2001</u>	<u>Total</u>
Salary- stipend of author (all other was voluntary)	\$7,000	\$7,000	\$7,000	-	\$21,000
Transport- airfares, Fleet SA 4WD hire	\$2,400	\$3,200	\$3,200	\$800	\$9,600
Fieldwork supplies & consumables- \$400 wk	\$1,200	\$1,600	\$1,600	\$400	\$4,800
Radio equipment - collars	\$2,800	\$900	\$2,400	-	\$6,100
Traps- including foam and shadecloth	\$1,500	-	-	-	\$1,500
	\$14,900	\$12,700	\$14,200	\$1,200	\$43,000

The trapping program at Aroona Sanctuary caught 21 wild-born animals, equating to approximately \$2,000 per animal. If the monitoring of released *P. x. xanthopus* is included, the cost is closer to \$1,500, which ranged between one and 20 captures per animal. There was a total of 133 *P. x. xanthopus* captures (not including pouch young) from 12 trapping fieldtrips at a cost of \$323 per capture.

4.4.2 Staff

The *P. x. xanthopus* reintroduction project plan was written by Ferris and Macdonald (1995). A supplement was completed by Conaghty (1996). A reintroduction committee comprising C. Macdonald (project coordinator), E. McAlister, M. Craig, D. Schultz, B. Campbell, (RZSSA), B. Odermatt, K. Waters (NRG Flinders), P. Copley and P. Alexander (DEH) was formed in 1995. S. Conaghty subsequently coordinated the project during the release phase, followed by S. Barlow (former Species Management Officer RZSSA) post-release from 1996 to 1999, T. Morley (Reptile Curator RZSSA) from mid-1999 to mid-2000, and A. Emmerich (current Species Management Officer RZSSA) mid-2000 to present. J. Crutchett (Property Manager for Monarto Zoological Park RZSSA) coordinated feral animal control throughout the project, and B. Odermatt continues to coordinate the project for NRG Flinders.

The author was responsible for all aspects of the trapping program initiated as part of the current study, with guidance from veterinarians S. Conaghty and D. Schultz, and biochemist D. Griffin. Regular volunteer assistance was provided by I. Arthur and C. Davis (Leigh Creek) in free-baiting traps at Aroona Sanctuary. Twenty-seven volunteers, not including members of the Leigh Creek community, assisted in trapping at Aroona Sanctuary.

4.4.3 Expertise

Little was known regarding *Petrogale* reintroduction biology prior to the commencement of this project. Many of the acknowledged wild and captive *P. x. xanthopus* experts formed part of the project committee. A geneticist was not consulted because a detailed studbook existed for the species (coordinated by C. Macdonald). Experts consulted as

part of the trapping program were veterinarians D. Schultz and S. Conaghty, and ecologist M. Lethbridge.

4.4.4 Time

The location of each *P. x. xanthopus* was triangulated twice daily for 40 days after their release into Aroona Sanctuary in September 1996 by RZSSA staff and volunteers. The pulse rate of each collar continues to be checked daily to determine if an animal is alive (regular beat 40-60 beats per minute, mortality 80 bpm) by the Aroona Sanctuary caretaker. Triangulation was reduced to one week every month in 1997, one week every three months in 1998 and one week in every six months in 1999. The Automated Tracking System (ATS) has been logging the position of each animal hourly since 2000, although further testing and upgrading is required before the system and results are reliable. Feral animal control has remained on a one week per three-month schedule since release.

For the author, the time devoted to the project was a minimum of three years of fieldwork and six months of data analysis and publication. Core funding for the project was guaranteed for a minimum of three years from RZSSA and NRG Flinders. A total of 14 one-week fieldtrips were undertaken to Aroona Sanctuary, including an initial *P. x. xanthopus* survey to determine trap placement (December 1997), actual trap placement (January 1998) and 12 three-monthly trapping fieldtrips in April, July, October and January thereafter until January 2001. Eight fieldtrips had been undertaken to Aroona Sanctuary prior to December 1997 as part of an B.Sc. (Honours) research project (Lapidge 2000) and voluntary radio-tracking and feral animal control. Future monitoring of the reintroduction will be continued by the RZSSA and Leigh Creek Area School.

4.4.5 Equipment and technology

Equipment required to undertake the reintroduction of *P. x. xanthopus* was a healthy, viable population of the species in appropriate captive facilities, a 4WD vehicle, office space and supplies, infrastructure and access to the release site, radio-telemetry equipment (transmitters and three receivers), exotic predator control supplies, and fieldwork supplies. *P. x. xanthopus* released were chosen from a captive stock of 60

animals at Monarto Zoological Park. A 4WD for the initial monitoring was donated by Nissan Australia, thereafter vehicles were hired from the South Australian Government Fleet (Fleet SA). Office space, supplies (office and fieldwork) and initial radio-telemetry transmitters were guaranteed by the RZSSA. Infrastructure at Leigh Creek (house) and access to Aroona Sanctuary were guaranteed by NRG Flinders. Telemetry receivers and hand-held UHF radios were supplied by Transceiver Services, Adelaide. Equipment and consultants (M. Lethbridge) for ATS were supplied by Transceiver Services through a grant received by Leigh Creek Area School. Labour to erect and test the system was provided by teachers and students of Leigh Creek Area School. Baits used in predator control, undertaken by J. Crutchett and volunteers, NRG Flinders staff and local pastoralists, were supplied by NRG Flinders.

The grant received from NRG Flinders to undertake the trapping project was sufficient to purchase required equipment and technology including specially-designed traps (Crestware Industries, Adelaide), some radio-collars (Titley Electronics), 4WD hire (Fleet SA), return flights from Sydney to Adelaide, and fieldwork supplies. NRG Flinders guaranteed the use of a house and boat for the duration of the project. Further radio-collars were supplied by Aroona Catchment Biodiversity Enhancement Project (Natural Heritage Trust).

4.4.6 Education

Extensive promotion of and education about the reintroduction of *P. x. xanthopus* was undertaken by the RZSSA and NRG Flinders at considerable cost (no values could be obtained). The northern Flinders Ranges community was informed and updated on the project through a town meeting, flyers distributed in letter-boxes, posters around town, radio, company newsletters, *Rock-wallaby News* (produced by Leigh Creek Area School), a pageant float and a display at the Leigh Creek Visitor Centre. Local pastoralists owning stations bordering Aroona Sanctuary were invited to participate in the buffer zone at a barbeque in April 1996. Students at Leigh Creek Area School were involved in the project through regular lessons led by RZSSA employees and myself, the production of *Rock-wallaby News*, and assisting in radio-tracking and trapping. The South Australia public were educated about the project through radio, the Adelaide newspaper *The Advertiser* and a pamphlet sent to every home in the state with their electricity bill. The

international zoo community and RZSSA members were informed through conference presentations and the Society's newsletter *Zootimes*. As the trapping program was a continuation of the monitoring, and as the author had worked on the project since fieldwork commenced, no additional education was required. However, trapping updates were supplied on ABC radio, in *Rock-wallaby News*, and through emailed trapping fieldtrip reports.

4.4.7 Post-release monitoring

Aroona Sanctuary was considered an ideal site for post-release monitoring, hence requiring fewer resources to undertake effective post-release monitoring. The sanctuary is easily accessed by sealed road and is locally controlled by NRG Flinders, a house and electronic facilities were supplied, the Leigh Creek township (and associated infrastructure including shops, hospital, airport) are a 10 km drive from the release site, a supportive community and school existed in Leigh Creek with many volunteers, and the high peaks surrounding the release site are ideal for triangulation radio-tracking, yet are rugged enough to minimise the chance of public interference with released or wild-born *P. x. xanthopus*. A major addition to the trapping program was a boat supplied by NRG Flinders that could be used to check the 12 traps placed around the dam. This reduced walking time by approximately two hours per day and meant that more traps could be used, captured animals could be freed earlier in the morning and traps set later in the afternoon before dark.

4.4.8 Criteria for success

No success criteria were established or documented by the RZSSA or NRG Flinders (Ferris and Macdonald 1995; Conaghty 1996). Known objectives include:

1. the establishment of a exotic predator-free sanctuary and buffer zone,
2. the survival of released *P. x. xanthopus*,
3. breeding by the release animals and their offspring,
4. the cause of any *P. x. xanthopus* deaths be identified,
5. promotion of the reintroduction to the international zoo community,
6. a sense of project ownership by the local community, and

7. the formation of a viable, free-ranging and self-sustaining population of *P. x. xanthopus*.

In relation to the trapping program, additional success criteria were:

8. the regular capture of all remaining released and wild-born *P. x. xanthopus* for physiological monitoring,
9. the collection of data on population growth and seasonal health trends for *P. x. xanthopus*.

Success of individual objectives and criteria are discussed in Chapter 6, 7 and 9.

4.4.9 Analysis and publication

Numerous presentations and publications discussed the reintroduction of *P. x. xanthopus* to Aroona Sanctuary. Because many of the articles are not readily available, they are not referred to in this thesis, but a list is supplied below.

- Barlow S. (1997) Yellow-footed Rock-wallaby reintroduction- a conservation partnership with the zoo community, sponsors and the public. *ARAZPA/ASZK Conference Proceedings*, Auckland: pp. 107-112.
- Barlow S. and McAlister E.J. (1997) A release program for Yellow-footed Rock-wallaby (*Petrogale xanthopus xanthopus*) to the Flinders Ranges, South Australia. *52nd IUDZG Conference Proceedings*, Berlin: pp. 155-60.
- McAlister E.J. and Barlow S. (1997) Using release programs to advance one objective of the modern zoo. *7th SEAZA Conference Proceedings*, Singapore: pp. 38-40.
- Conaghty S. and Crutchett J. (1997) Feral animal control at Aroona Sanctuary for the reintroduction of the Yellow-footed Rock-wallaby. Unpublished RZSSA Report. 7 pgs.
- Conaghty S. and Schultz D. (1998) Veterinary investigation of the Yellow-footed Rock-wallaby (*Petrogale xanthopus xanthopus*) for reintroduction. *American Association of Zoo Veterinarians Conference Proceedings*, Nebraska: pp. 294-300.
- Barlow S. (1999) Reintroduction of the Yellow-footed Rock-wallaby (*Petrogale xanthopus xanthopus*) to Aroona Sanctuary, northern Flinders Ranges, South Australia. *Thylacinus* **23**: 12-16.
- Craig M. and Barlow S. (1998) The reintroduction of the Yellow-footed Rock-wallaby (*Petrogale xanthopus xanthopus*) to Aroona Sanctuary, northern Flinders Ranges, South Australia. *7th World Conference on Breeding Endangered Species*, Cincinnati: p. 241.

- Lapidge S.J. (2000) Dietary adaptation of reintroduced Yellow-footed Rock-wallabies, *Petrogale xanthopus xanthopus* (Marsupialia: Macropodidae) in the northern Flinders Ranges, South Australia. *Wildlife Research* **27**: 195-201.
- Lapidge S. and Odermatt B. (2000) Returning Yellow-foots to a welcoming community. *2nd National Rock-wallaby Symposium*, Alice Springs, June, 2000

Lapidge S. and Hume I. (2000, 2001) and Lapidge S. J. (2000) presentations listed in 4.3.9 also contained research results from the reintroduction of *P. x. xanthopus* Furthermore, three of the five papers planned for later publication will involve South Australian data.

4.5 Discussion

The total cost of the present project is estimated at approximately \$174,600, of which \$79,100 was raised through sponsorship and a minimum of \$7,600 was in-kind assistance. Of the total, 39% of costs were equipment, supplies and sample analysis, 35% salary and 26% transport. It does not include salaries for the 80 people who volunteered their time to directly assist with fieldwork or other aspects of the project. The combined population of both reintroductions at the cessation of trapping in January 2001 was 75 animals, hence each animal has cost \$2,300 to monitor. This compares to USD\$80,000 (AUD\$160,000) per *E. luturis* rehabilitated and reintroduced after the Exxon Valdez oil spill (Estes 1998) or USD\$22,563 (AUD\$45,000) per reintroduced *L. rosalia* in Brazil (Kleiman *et al.* 1991). These are the only other values per animal that are available.

A total of 54 weeks of fieldwork was undertaken on the research reported in this thesis. Consequently, over a third of the time allocated to the project (not including thesis preparation) has been field research. In terms of financial resources, each week of fieldwork has cost about \$3,200. This may be a more directly relatable and less anthropomorphic guide for any potential reintroduction project, rather than cost per surviving animal.

Success of reintroduction parameters such as education or population growth should always be stated cautiously and within a time frame. Despite the years of education devoted to the Oman community in the *O. leucoryx* reintroduction (Stanley Price 1989), the project has returned to captive-breeding after almost two-thirds of the wild

population was poached; the remaining 96 animals was considered no longer viable (Gorman 1999). Similarly, despite the intensive community education efforts involved in the *L. rosalia* reintroduction project in Brazil (Kleiman 1989; Kleiman *et al.* 1989; Kleiman *et al.* 1986), eight animals or 21% were lost to theft (Beck *et al.* 1991). Thus, while education is important, it cannot replace site protection.

Table 4.3 summarizes and compares the resources used for the two reintroduction sites using the selection criteria discussed. For all except education and publications, Aroona Sanctuary only refers to the trapping program.

Table 4.3 Comparison of Lambert Station and Aroona Sanctuary for resource criteria.

Resource criteria	Lambert Station <i>Queensland</i>	Aroona Sanctuary <i>South Australia</i>
1. Funding	QEPA, NPF, AZA, AG	RZSSA, NRG, ACBEP, ZPBNSW
2. Staff	Author, I. Hume, QEPA	Author, RZSSA
3. Expertise	QEPA, RZSSA	RZSSA, DEH
4. Time	40 weeks of fieldwork	14 weeks of fieldwork
5. Equipment (suppliers)	I. Hume, QEPA	NRG
6. Education	Nominal	Extensive
7. Criteria for success	Scientifically based	Community based
8. Post-release monitoring	Extensive	Extensive
9. Publications (combined)	6 conferences, 4 articles	4 conferences, 4 articles

Key to abbreviations: QEPA: Queensland Environment Protection Agency, NPF: National Parks Foundation of South Australia Inc., AZA: American Zoo Association Marsupial and Monotreme Taxon Advisory Group, AG: Australian Geographic, RZSSA: Royal Zoological Society of South Australia Inc., NRG: NRG Flinders Pty. Ltd., ACBEP: Aroona Catchment Biodiversity Enhancement Project, ZPBNSW: Zoological Parks Board of New South Wales, DEH: Department of Environment and Heritage South Australia, I. Hume: Professor Ian Hume, Project Supervisor, The University of Sydney.

SEASONAL TRENDS AT RELEASE SITES

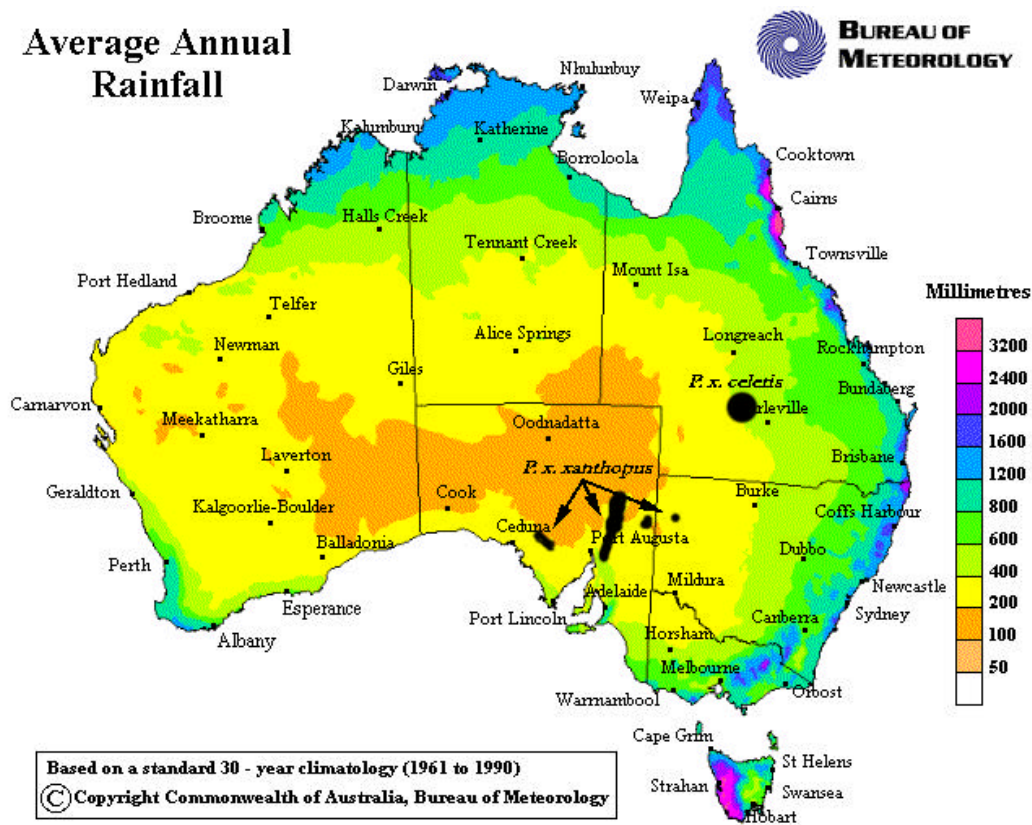
5.1 Introduction

The Yellow-footed Rock-wallaby, *Petrogale xanthopus*, occupies the hottest and driest areas of all *Petrogale* species (Lim *et al.* 1997). High summer temperatures are a consistent feature of all parts of the species' range (Figs. 2.2, 2.5). The two subspecies occupy two distinct climatic regions (Fig. 5.1). In Queensland, *P. x. æleris* occurs in the dry-tropics, with an annual rainfall between 391 mm (Adavale) and 528 mm (Blackall) falling predominantly in the summer monsoon season. In South Australia and New South Wales *P. x. xanthopus* occurs in semi-arid to arid areas with annual rainfall between 346 mm (Port Pirie) and 215 mm (Leigh Creek) falling predominantly in winter or sporadically (BOM 2001). Rainfall was reported by Lim *et al.* (1987) and Copley and Alexander (1997) to directly affect *P. xanthopus* population density, but Sharp and Norton (2000) concluded that it had only a minimal effect. Robinson *et al.* (1994) suggested that there was a positive time-lagged relationship between effective rainfall and the number of births for *P. x. xanthopus* at Middle Gorge in South Australia, but no correlation was performed. Thus the importance of variations in total annual rainfall and its incidence on *P. xanthopus* is unclear.

An initial aim of this project was to examine the influence of a sympatric species on the establishment of a reintroduced species. The native species most similar to *P. xanthopus* in terms of ecological requirements is the Euro, *Macropus robustus erubescens* (Dawson and Ellis 1979; Lim *et al.* 1987; Lim *et al.* 1992). This sedentary species is also reported to be a strong defender of territory (Ealey 1967; Croft 1985). Consequently, *M. r. erubescens* population densities were monitored at both the Queensland and South Australian

reintroduction sites. Feral goats (*Capra hircus*) also occupy a similar niche to *P. xanthopus*, but their numbers were kept low at both fieldsites by mustering and shooting. Five methods are commonly used to survey macropod numbers: total counts, mark-recapture, addition-removal, sample counts (including faecal pellet quadrats), and catch-effort. The merits and disadvantages of each have been discussed by Southwell (1989) and Jarman and Capararo (1997).

Figure 5.1 Annual rainfall patterns over the range of *Petrogale xanthopus* (BOM 2001).



Lim *et al.* (1987) and Robinson *et al.* (1994) used effective rainfall, defined as the monthly total divided by a third of the evaporation rate, while Copley and Alexander (1997) and Sharp and Norton (2000) used total rainfall in their respective investigations into the role of rainfall in the regulation of *P. xanthopus* populations. Besides predation and stochastic events, herbivore populations are regulated by the abundance and quality of vegetation (Begon *et al.* 1990), which is intern related to rainfall. However, in my opinion, rainfall is not directly related to vegetation growth as the rainfall rate per unit time, total amount, topography, soil substrate and seed bank also determine vegetation growth. A heavy

downpour onto a steep cliff with a thin soil layer can erode remaining vegetation and the seed bank causing long-term damage to vegetation communities present. Thus, vegetation abundance and diversity were surveyed directly at regular intervals at both fieldsites.

This chapter describes the rainfall and survey results of macropods and vegetation throughout the study period and forms a basis for later chapters on the ecological and physiological adjustments of captive-bred *P. xanthopus* in Queensland and South Australia.

5.2 Lambert Station in Queensland

5.2.1 Climate

Rainfall at Lambert Pastoral Station was measured using a standard rainfall gauge at the homestead and checked daily by the station owner. The location of the gauge was 4.4 km south of Site 1, 2.8 km north-northwest of Site 2 and 2.4 km north of Site 3 (Fig. 5.4). Records commenced in 1957 (Fig. 5.2), with a mean annual rainfall of 415 ± 152 mm being recorded; lower than the closest towns of Charleville (498 mm, $n=82$) and Blackall (528 mm, $n=120$) (BOM 2001). However, mean annual rainfall has been increasing since records commenced as indicated by the linear trendline in Fig. 5.2. The station received its highest rainfall for 23 years (672 mm) in 1997, the year before the project commenced and received above-average rainfall of 520 mm, 511 mm and 677 mm throughout the project in 1998, 1999 and 2000 respectively (mean 569 mm). However, little rain fell in the winters of 1999 and 2000, with near-drought conditions prevailing (Fig. 5.3).

5.2.2 Fauna

Sample count variable strip-transect spotlight surveys (Southwell 1989) were started on Lambert Station six months before the release, and continued twice per fieldtrip throughout the project. Thirty-five kilometres of transects were consistently surveyed (Fig. 5.4). A survey was undertaken at the start and end of each fieldtrip on relatively windless nights without rain, commencing 1 h after dusk and continuing for approximately 4 h until complete. The replicate survey for each fieldtrip occurred in the

Figure 5.2 Annual rainfall for Lambert Pastoral Station, southwestern Queensland 1957-2000. ➔ Indicates mean of 415 ± 152 mm. Linear trendline indicates mean annual rainfall has been increasing since records for the station commenced.

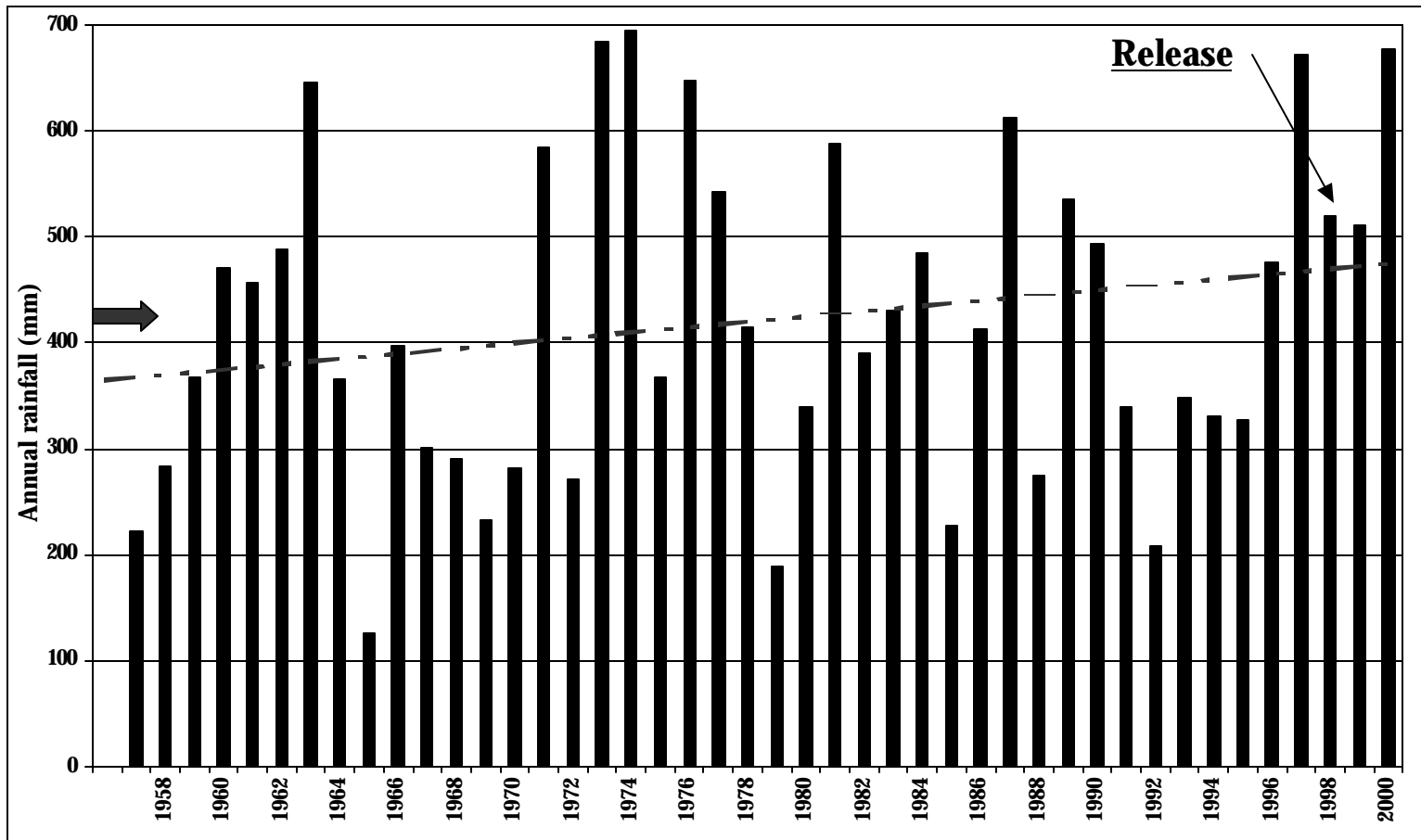


Figure 5.3 Monthly rainfall for Lambert Pastoral Station, southwestern Queensland, throughout the study period. Month of release and bi-annual trapping fieldtrips are indicated (T).

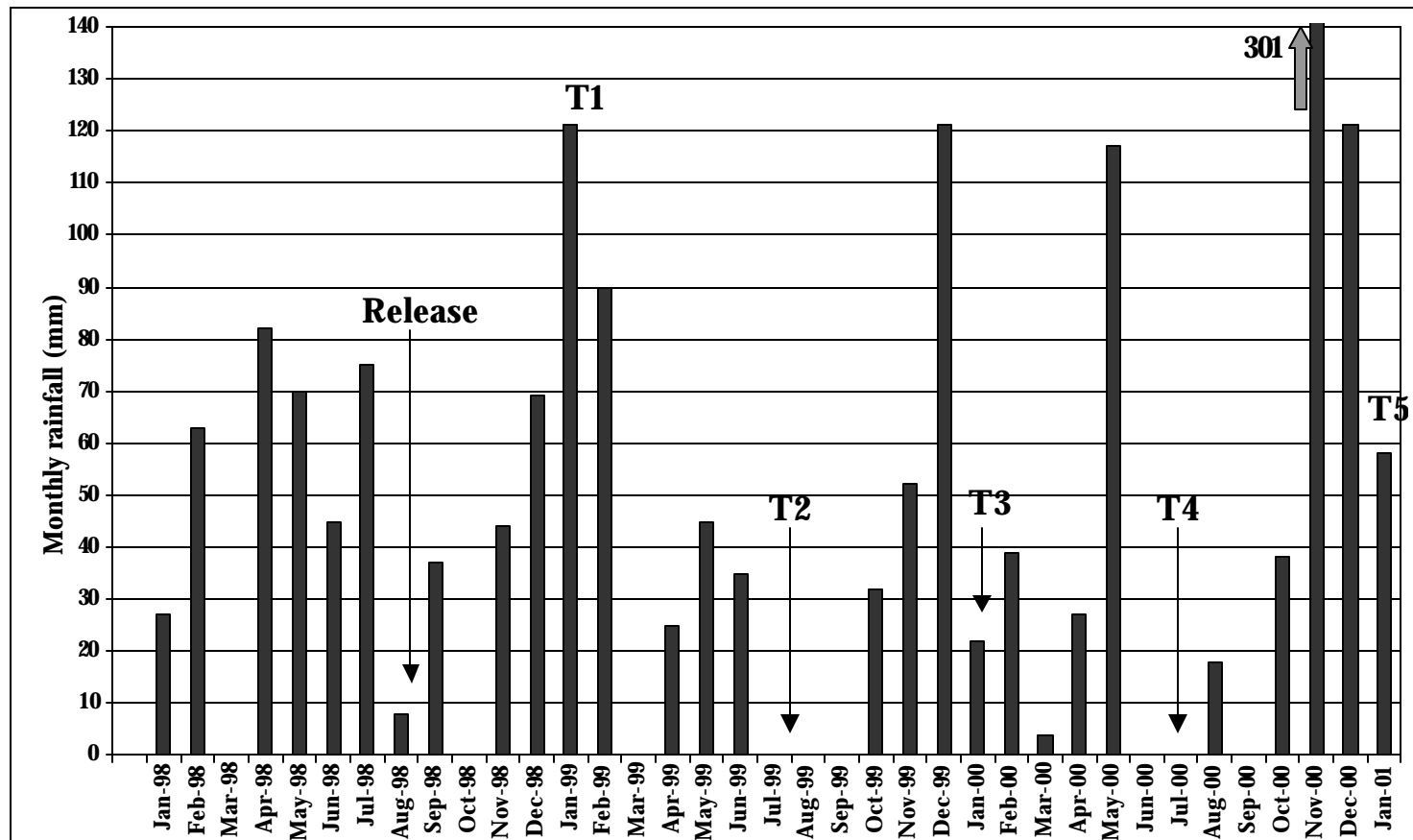
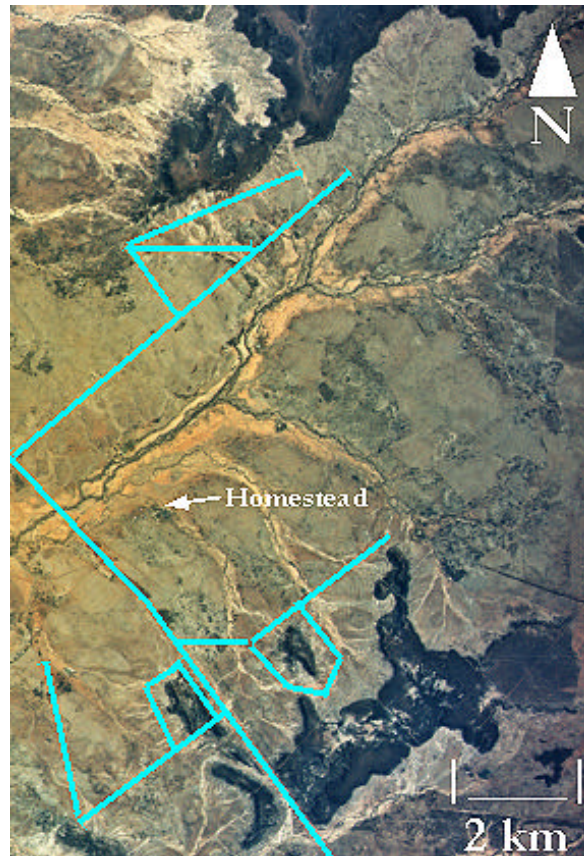


Figure 5.4 Spotlight survey transects on Lambert Station. Suitable tracks for vehicle surveys did not exist directly north of Site 1.



reverse site order as the first to minimise time of night as a bias. Transects were paddock tracks, as vegetation prevented easy vehicle access to paddocks, and varied in strip width in relation to topography and vegetation density from 400 m to 10 m (average 200 m). A vehicle speed of 5 - 15 km/h was maintained throughout surveys, depending on track condition. The vehicle (4WD) was fitted with a three million candlepower roof-mounted spotlight (Kanga) on the front passenger side and a one million candlepower handheld spotlight (Lightforce) on the right rear passenger side. The operator of the front light conducted 110° of the 180° scan. A rifle (Ruger M77 MkII 223REM) was carried for the destruction of European foxes (*Vulpes vulpes*), Feral cats (*Felis catus*) and Feral pigs (*Sus scrofa*). Distant animal identification was confirmed using 12X binoculars (Optics) or a 9X riflescope (Tasco) with the aid of the brighter spotlight. No attempt was made to differentiate among the three species of large macropod at the release sites, *M. r. erubescens*, *M. rufus* (Red Kangaroo), and *M. giganteus* (Eastern Grey Kangaroo), because of the inexperience of naive volunteers. However, the majority of macropods sighted were *M. r. erubescens*, as surveys were concentrated near hills. Although Southwell (1989) cautioned against the use of the technique for estimating absolute density of a species, it

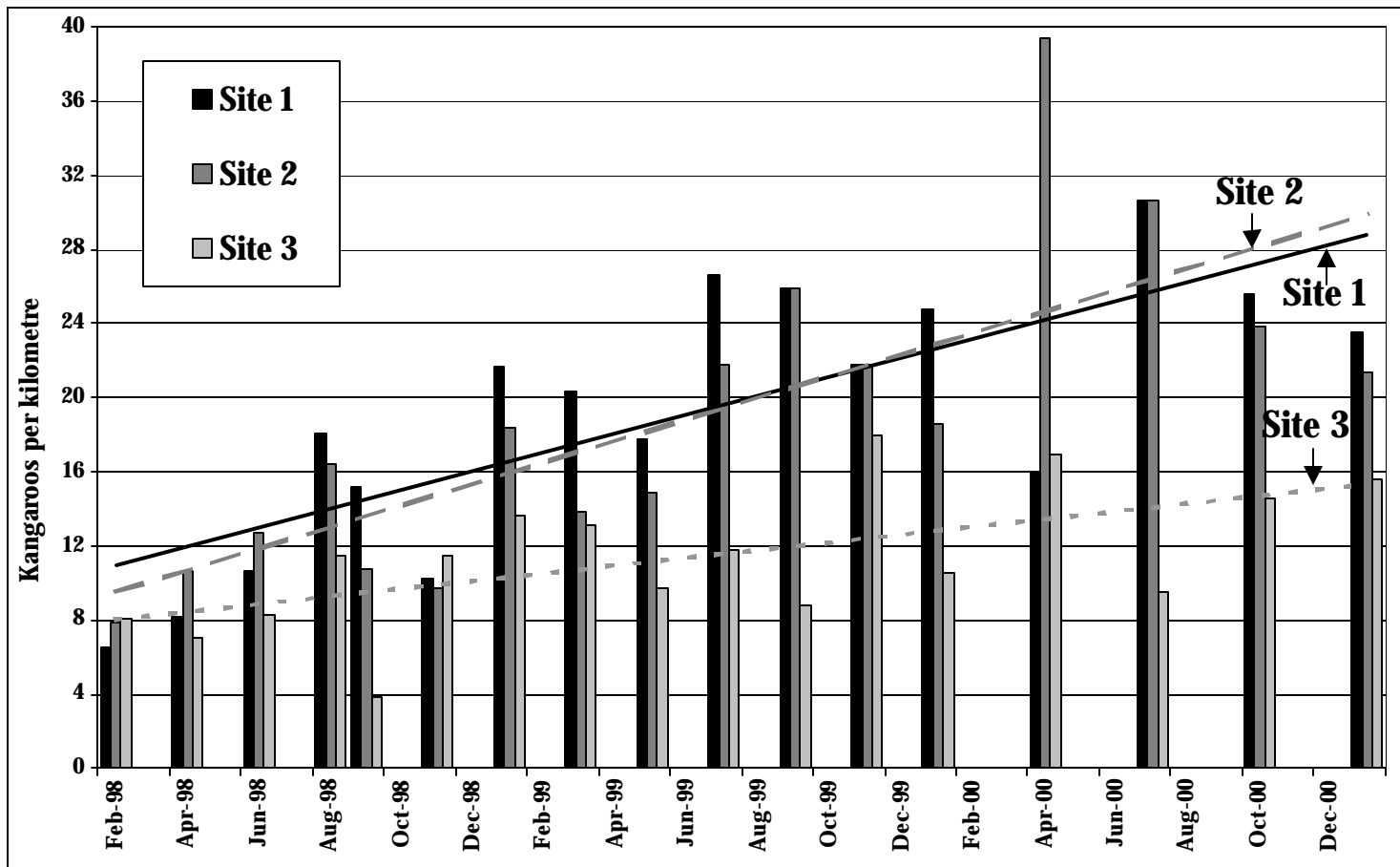
has been recommended for monitoring population trends (Southwell 1989), the purpose of these surveys.

Results of the macropod spotlight surveys are shown in Fig. 5.5. Macropod numbers varied between 6.5 and 39.4 per km. Site 1 had an overall average of 19.0 macropods per transect km, Site 2 had 18.7 and Site 3 had 11.3. Initial ground surveys in November 1997 indicated that Site 1 may have a greater macropod population; it was consequently deemed an 'experimental' site to test the impact of sympatric macropods on reintroduced *P. x. celeris*. However, subsequent spotlight surveys indicated no significant difference between Sites 1 and 2 but a consistent and highly significantly lower macropod population at Site 3 when tested using one-factor ANOVA ($F=7.42$, $F_{0.05(1),2,48}=3.19$, $P<0.002$) (Fig 5.5). Survey distances, topography, mesa area and proximity to free-water were all similar among the three sites; the lower macropod numbers at Site 3 must therefore reflect other aspects of habitat quality.

The reason for the increase in macropod numbers throughout the study is unknown (Fig. 5.5). Annual population increases for the three large macropod species surveyed have been observed by the Queensland Environmental Protection Agency Macropod Survey Unit over recent years and have been related to above-average rainfall (M. Evans pers. comm.). Local exotic predator reduction may have further increased macropod recruitment to an unknown extent; macropod harvesting yields have remained steady between years (S. Henshall pers. comm.).

To examine habitat use by *M. r. erubescens*, *P. x. celeris* and *C. hirus*, nine 1-m diameter fixed faecal pellet quadrats were established at each release site. Each species has distinctive faecal pellets that can be identified on sight (Lim *et al.* 1992). Three quadrats were positioned along a transect at the point of release; one on top of the hill, one mid-slope and one at the base. Two further quadrat transects occurred at each site in areas known to be occupied by *P. x. celeris* from radio-telemetry records. Quadrats were placed in level areas that initially contained fresh *M. r. erubescens* faecal pellets and the diameter was lightly marked with paint (Dy-mark). Quadrats were cleared of all existing faecal pellets at the start of each fieldtrip, then checked 2 - 7 days later. Variation in collection period resulted from weather patterns; quadrats were checked early if storms were approaching, as water runoff washed away most faecal pellets present. All collected faecal

Figure 5.5 Macropod survey results with trendlines for Lambert Pastoral Station, southwestern Queensland, throughout the study period.
 Significantly less macropods occurred at Site 3 (one-factor ANOVA: $F=7.42$, $F_{0.05(1),2,48}=3.19$, $P<0.002$).



pellets were identified to the species from which they came, counted and grouped into quadrat height. Totals were divided by the number of days of collection to standardise results between surveys. Results for *M. r. erubescens* are shown in Table 5.1. The survey technique was not appropriate for *P. x. celeris* or *C. hirus* as few scats were found in quadrats throughout the study, possibly because of the low number or position of quadrats.

Table 5.1 *Macropus robustus erubescens* faecal pellet quadrat counts. Results are the number of faecal pellets deposited per quadrat divided by days of collection. Height data (Top, Mid, Base) are means of three quadrats \pm standard deviation. Site and height totals are means \pm standard error. There was no significant difference between faecal density between sites ($F=1.24$, $F_{0.05(1),2,36}=3.25$, $P=0.30$). Shading indicates Sites 2 and 3 had significantly more faecal pellets mid-slope ($F=5.66$ (Site 2) and 5.46 (Site 3), $F_{0.05(1),2,36}=3.25$, $P<0.01$). No significant difference was detected at Site 1.

	SITE 1				SITE 2				SITE 3			
	Top	Mid	Base	Total	Top	Mid	Base	Total	Top	Mid	Base	Total
Sep-98	0.9 \pm 0.3	0.7 \pm 0.5	0.7 \pm 0.2	0.8\pm0.1	0.4 \pm 0.3	1.8 \pm 0.9	0.6 \pm 0	0.9\pm0.6	0.9 \pm 0.3	0.6 \pm 0.1	0.3 \pm 0.1	0.6\pm0.2
Nov-98	0.9 \pm 0.4	0.4 \pm 0.1	0.4 \pm 0.2	0.6\pm0.2	0.3 \pm 0.3	1.0 \pm 0.3	0.4 \pm 0.1	0.6\pm0.3	1.2 \pm 0	1.0 \pm 0.3	0.4 \pm 0.1	0.9\pm0.3
Jan-99	2.3 \pm 0.5	0.8 \pm 0.8	2.0 \pm 0.0	1.7\pm0.6	2.6 \pm 1.2	3.3 \pm 1.1	2.6 \pm 1.0	2.8\pm0.3	2.7 \pm 1.4	2.6 \pm 1.1	3.6 \pm 1.0	2.9\pm0.4
Mar-99	1.4 \pm 0.5	1.6 \pm 0.7	0.8 \pm 0.4	1.3\pm0.3	1.1 \pm 0.3	0.9 \pm 0.5	0.5 \pm 0.1	0.8\pm0.2	1.2 \pm 0.6	1.2 \pm 0.6	1.3 \pm 0.5	1.2\pm0.1
May-99	1.5 \pm 0.5	1.2 \pm 1.2	2.2 \pm 1.4	1.6\pm0.4	0.4 \pm 0.6	2.8 \pm 1.7	0.7 \pm 0.5	1.3\pm1.1	0.8 \pm 0.9	0.8 \pm 0.3	0.5 \pm 0.7	0.7\pm0.1
Jul-99	2.1 \pm 0.9	2.0 \pm 1.4	1.4 \pm 0.1	1.8\pm0.3	1.9 \pm 1.1	1.6 \pm 0.8	1.4 \pm 1.1	1.6\pm0.2	1.0 \pm 0.9	1.4 \pm 1.2	0 \pm 0	0.8\pm0.6
Sep-99	2.3 \pm 0.3	1.2 \pm 0.6	2.2 \pm 0.5	1.9\pm0.5	1.3 \pm 1.1	1.9 \pm 0.9	1.4 \pm 0.8	1.5\pm0.3	0.6 \pm 0.7	0.9 \pm 0.8	0.2 \pm 0.2	0.6\pm0.3
Nov-99	1.4 \pm 0.2	2.6 \pm 3.0	0.7 \pm 0.7	1.6\pm0.8	1.0 \pm 1.4	2.8 \pm 2.1	0.5 \pm 0.3	1.4\pm1.0	0.3 \pm 0.3	2.1 \pm 1.3	1.7 \pm 1.0	1.4\pm0.8
Jan-00	0.9 \pm 0.1	1.3 \pm 0.8	2.6 \pm 1.7	1.6\pm0.7	0.5 \pm 0.4	1.2 \pm 1.0	1.4 \pm 0.8	1.0\pm0.4	0.3 \pm 0.3	0.8 \pm 0.2	1.0 \pm 0.7	0.7\pm0.3
Apr-00	1.5 \pm 0.7	1.0 \pm 0.7	1.0 \pm 0.3	1.2\pm0.2	1.7 \pm 1.0	1.1 \pm 0.6	0.5 \pm 0.5	1.1\pm0.5	0.3 \pm 0.5	0.7 \pm 0.5	0.2 \pm 0.2	0.4\pm0.2
Jul-00	1.5 \pm 0.9	2.9 \pm 0.9	0.9 \pm 1.2	1.8\pm0.8	1.5 \pm 0.7	1.3 \pm 0.7	1.4 \pm 0.6	1.4\pm0.1	1.0 \pm 0.8	2.7 \pm 1.0	0.7 \pm 0.3	1.5\pm0.9
Oct-00	0.3 \pm 0.3	0.8 \pm 0.9	0.9 \pm 0.8	0.7\pm0.3	0.2 \pm 0.3	1.3 \pm 1.2	1.8 \pm 1.6	1.1\pm0.7	0.8 \pm 0.3	2.4 \pm 2.6	0.3 \pm 0.3	1.2\pm0.9
Jan-01	1.4 \pm 1.6	0.5 \pm 0.2	0.5 \pm 0.2	0.8\pm0.4	0 \pm 0	0.4 \pm 0.3	0.3 \pm 0.2	0.2\pm0.2	0.2 \pm 0.3	0.5 \pm 0.3	0.3 \pm 0.4	0.3\pm0.1
TOTAL	1.4\pm0.6	1.3\pm0.7	1.3\pm0.7		1.0\pm0.7	1.6\pm0.8	1.0\pm0.7		0.9\pm0.6	1.4\pm0.8	0.8\pm0.9	

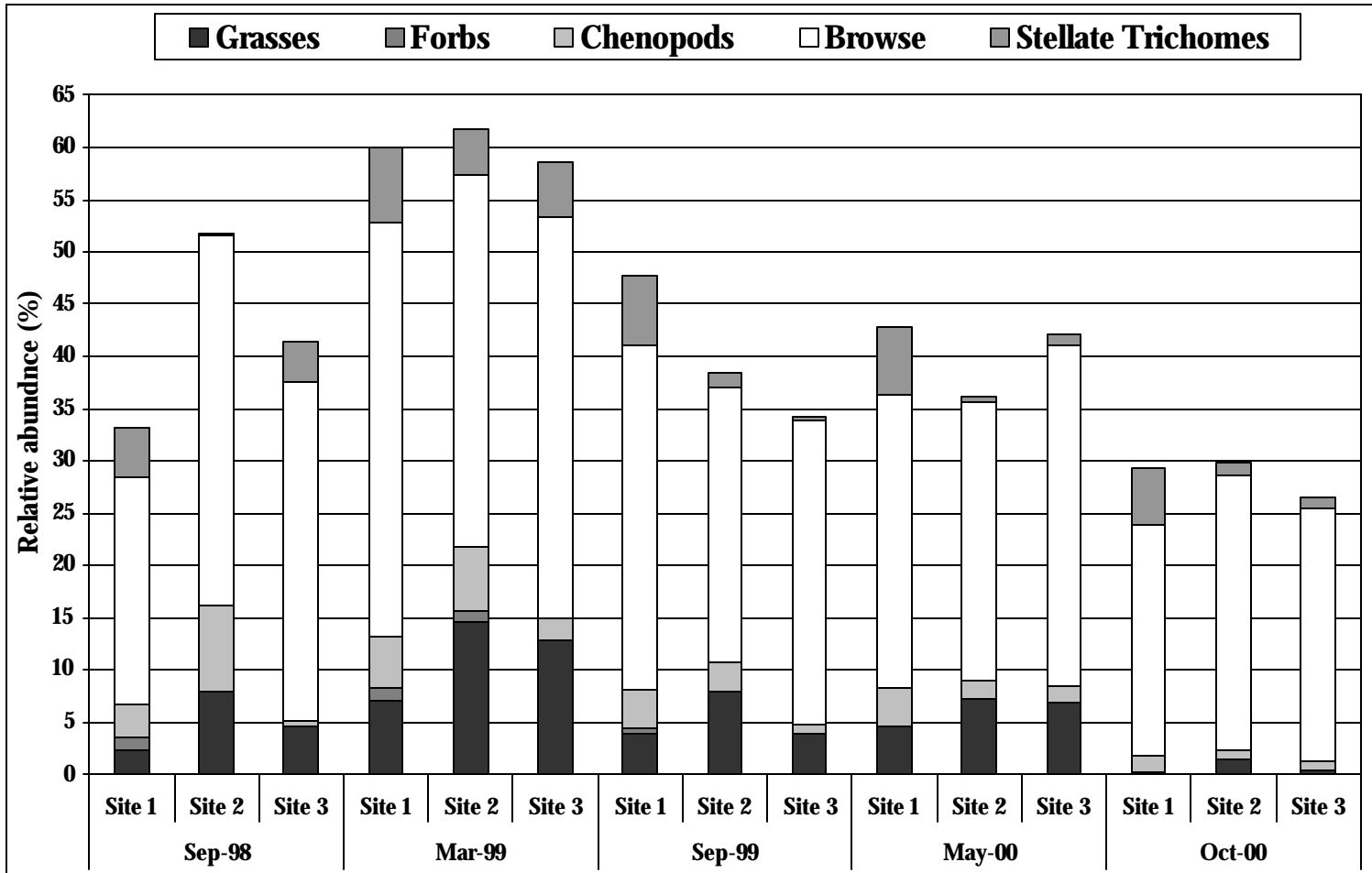
Estimations of *M. r. erubescens* population density were not possible. However, results from faecal pellet quadrat counts indicated similar population trends to spotlight survey counts for *M. r. erubescens*. Two of three Product Moment Correlation coefficients between macropod spotlight surveys results and faecal pellet quadrat results for each sample period were significant (Site 1: $r = 0.579$, $r_{0.05(2)9} = 0.553$, $P < 0.05$ and Site 3: $r = 0.724$, $r_{0.01(2)9} = 0.684$, $P < 0.01$) (Zar 1984). Although faecal pellets tended to be lower in

abundance at Site 3 than Sites 1 and 2 (Table 5.1), the difference was not significant when tested using one-factor ANOVA ($F=1.24$, $F_{0.05(1),2,36}=3.25$, $P=0.30$). Analysis of faecal pellet density at the three height levels (hill top, mid-slope or base) at each site indicated no significant difference for Site 1, but a highly significant difference for Sites 2 and 3 when tested using one-factor ANOVA ($F=5.66$ (Site 2) and 5.46 (Site 3), $F_{0.05(1),2,36}=3.25$, $P<0.01$), with more faecal pellets mid-slope. Sites 2 and 3 were lower mesas with less scree than Site 1 and thus contained more areas suitable for soft scrapes used as daytime refuge by *M. r. erubescens*. Dense *Acacia catenulate* (Bendee) on the slopes of Sites 2 and 3 provided shade and protection from aerial predators; *A. catenulate* was sparse on the slopes of Site 1.

5.2.3 Flora

The vegetation at each site was sampled bi-annually at the point of release to determine the relative density and coverage of plant groups using the line-intercept method (Brower *et al.* 1990). Although not suitable for determining vegetation biomass, the technique provides an effective way of surveying vegetation trends for comparison among sites (Brower *et al.* 1990). Plant groupings chosen for vegetation surveys were those defined by Ellis *et al.* (1977). The method involves scoring the transect length covered by living plants underlying or overlying a tape measure stretched across the community under study. Transects covered 50 m (Sites 1 and 3) or 24 m (Site 2- total width) on top of the hill from the outcrop back towards the hill centre, the hill slope (100 m at Site 1, 66 m at Site 2 and 60 m at Site 3) and 50m from the base of the hill on the plain (all sites). The length of transects was sufficient to sample the range of habitat used for foraging by *P. x. celeris*, as indicated by sightings post-release and radio-telemetry records. Total length of the Site 1 transect was 200 m, Site 2 was 140 m and Site 3 was 160 m. For comparative purposes, results were converted to a percentage cover of the total abundance for each plant group by dividing the total transect distance by each plant group total (Fig. 5.6). No significant difference was detected between vegetation abundance at the three sites through one-factor ANOVA ($F=0.077$, $F_{0.05(1)2,12}=3.88$, $P>0.05$), with varying vegetation densities between sites (Fig. 5.6) likely due to local variations in rainfall. Vegetation abundance was found to be dependent on the total rainfall between surveys at Sites 2 and 3 ($F=25.0$ and $F=28.5$ respectively, $F_{0.02(1)1,3}=20.62$, $P<0.02$) but not Site 1 ($F=2.90$, $F_{0.05(1)1,3}=10.1$, $P>0.05$) when tested by simple linear regression.

Figure 5.6 Relative abundance of plant groups at each Lambert Station reintroduction site throughout the study period.



5.3 Aroona Sanctuary in South Australia

5.3.1 Climate

Rainfall for Aroona Sanctuary was measured by an automated weather station at Leigh Creek Airport, 6.5 km east-southeast of the release site (Fig. 2.6). Records commenced in 1952, with a mean annual rainfall of 215 mm (BOM 2001). Annual rainfall in the years since release has been slightly above average (224 ± 52 mm), with 1997 and 2000 above (242 mm and 298 mm respectively) and 1998 and 1999 below average (196 mm and 159 mm respectively) (Fig. 5.7). Rainfall has been sporadic, with heavy falls recorded in every month except May and June (Fig. 5.7).

5.3.2 Fauna

The sample count variable strip-transect spotlight survey technique used to monitor *M. r. erubescens* population fluctuations in Queensland was not suitable for Aroona Sanctuary because of the inaccessibility of the release site. Consequently, *M. r. erubescens* surveys were initially based on visual sightings per unit time while checking traps each morning. However, the species became accustomed to the traps, and therefore the number of *M. r. erubescens* trapped per fieldtrip was used to monitor their presence (Fig. 5.8). Trapped animals were not marked, and thus identification of individuals was not possible, and estimates of population size using mark-recapture analysis could not be made. Number of *M. r. erubescens* trapped was inversely dependent on vegetation abundance (simple linear regression: $F=30.1$, $F_{0.005(1),6}=18.6$, $P<0.005$) once *M. r. erubescens* had become accustomed to traps one year after trapping commenced (Fig. 5.8). This may reflect movement of *M. r. erubescens* to Aroona Dam, the only source of free water in the area in drier periods, or that *M. r. erubescens* were attracted to food in traps more frequently when vegetation abundance was low.

Figure 5.7 Monthly rainfall at Leigh Creek, South Australia, throughout the study period. Month of release, trap placement and quarterly recapture fieldtrips (T) indicated.

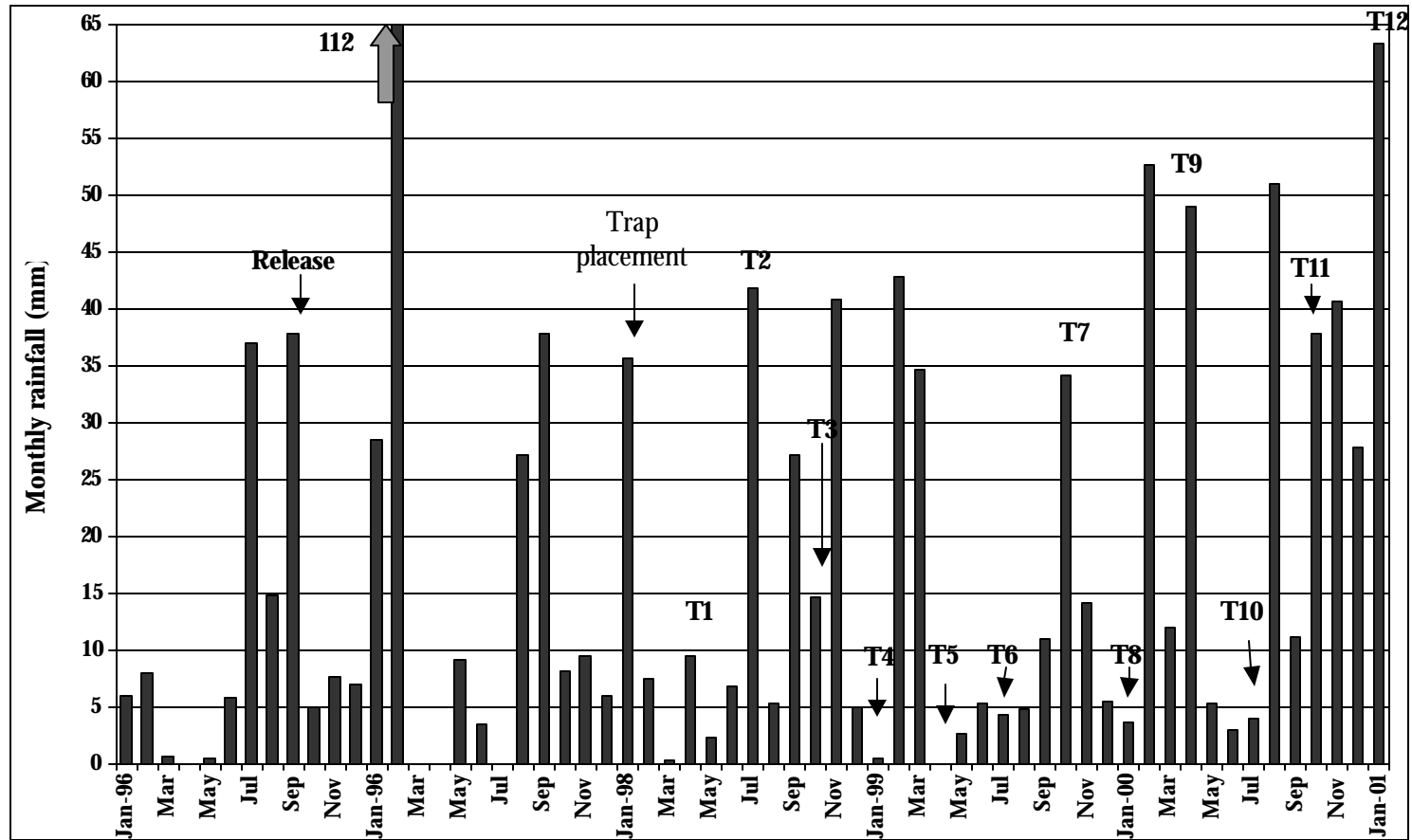
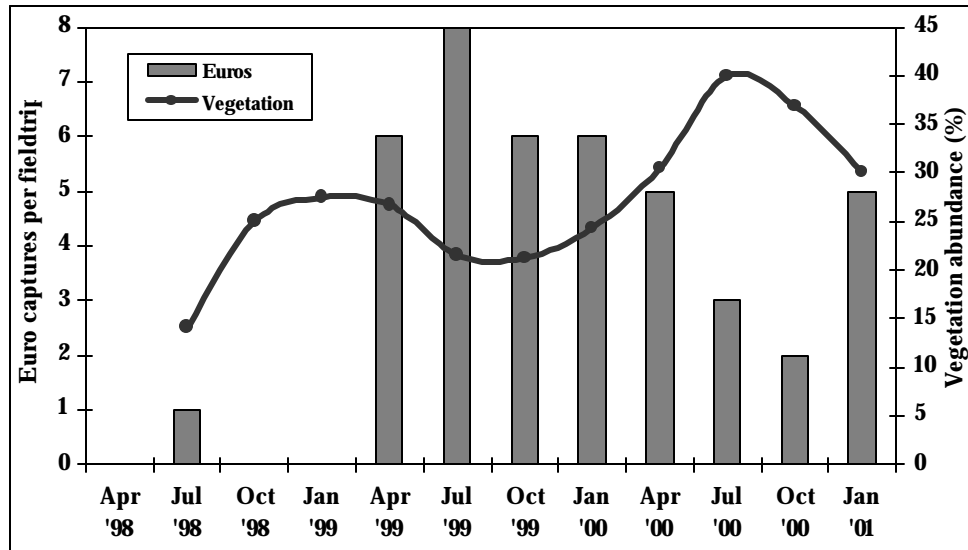


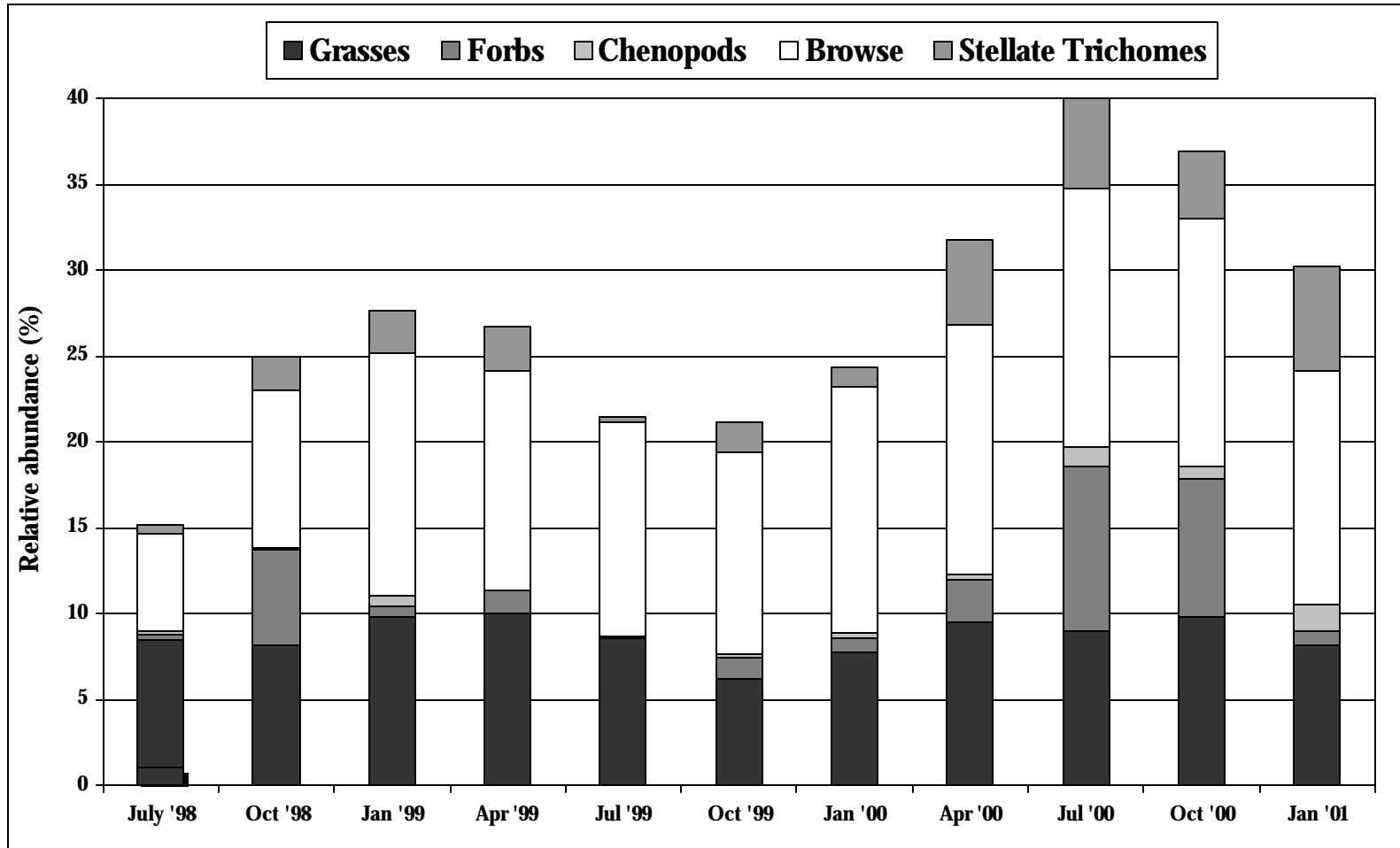
Figure 5.8 Euro captures and vegetation abundance at Aroona Sanctuary, South Australia, throughout the study period. Number of *M. r. erubescens* trapped was inversely dependent on vegetation abundance (simple linear regression: $F=30.1$, $F_{0.005(1),6}=18.6$, $P<0.005$) once *M. r. erubescens* had become accustomed to traps one year after trapping commenced (April '99).



5.3.3 Flora

Vegetation surveys were conducted at Aroona Sanctuary during each trapping fieldtrip using the line-intercept method employed in Queensland (Brower *et al.* 1990). Two 100-m transects were conducted at the outer limits of the *P. x. xanthopus* colony range as indicated by sightings and unpublished radio-telemetry records. Each transect started at the ridgeline and finished at the water (transect 1) or valley (transect 2) below. Transect 1 was sited adjacent to the outcrop complex where approximately half the *P. x. xanthopus* colony resided, and contained predominantly plants with stellate trichomes (*Sida*, *Abutilon*, *Solanum* and *Ptilotus spp.*) and browse species (*Hakea*, *Casuarina* and *Goodenia*). Transect 2 was sited in a Spinifex (*Triodia irritans*) hummock grassland containing Silver Senna (*Senna artemisioides artemisioides*) and Rock Emubush (*Eremophila freelingii*) where animals had been sighted foraging. Results from the two surveys were combined for each plant group, then halved to represent respective abundance per 100 m or a percentage by each plant group (Fig. 5.9). Vegetation abundance was not significantly dependent on rainfall between surveys ($F= 4.53$, $F_{0.05(1),9}= 5.12$, $P>0.05$) when tested using simple linear regression.

Figure 5.9 Relative abundance of plant groups at the Aroona Sanctuary reintroduction site throughout the study period.



5.4 Discussion

In the 30 months between release and cessation of fieldwork in January 2001, Lambert Station received 1424 mm of rain (Fig. 5.3), an average of 47 mm per month. In contrast, Aroona Sanctuary received 978 mm in the 52 months between release and cessation of fieldwork (Fig. 5.7), an average of 19 mm per month. Thus, Lambert Station received 250% more rain than Aroona Sanctuary since the respective reintroductions occurred. According to Lim *et al.* (1987), Robinson *et al.* (1994) and Copley and Alexander (1997), a greater rate of recruitment would be predicted for *P. x. celeris* at Lambert Station. Mean annual rainfall at Lambert Station and Aroona Sanctuary has generally increased since release (Figs. 5.3, 5.7), with the highest total recorded in 2000 at both fieldsites, thus recruitment should have increased throughout the study period at both fieldsites (Lim *et al.* 1987; Robinson *et al.* 1994).

No direct comparison can be made between *M. r. erubescens* densities at Lambert Station and Aroona Sanctuary. However, visual sightings indicated that the species is more abundant on Lambert Station, as no more than 10 *M. r. erubescens* were sighted in any one day at Aroona Sanctuary, whereas greater than 100 sightings were always recorded during surveys around each release site on Lambert Station, an area of similar size.

Vegetation surveys indicated that the release sites on Lambert Station had higher vegetation abundance than Aroona Sanctuary. Average vegetation abundance at Lambert Station was $42.3 \pm 11.2\%$ (Fig. 5.6) compared to $27.2 \pm 7.0\%$ at Aroona Sanctuary (Fig. 5.9). The higher vegetation abundance on Lambert Station was predominantly in the form of *Acacia* woodland on hill slopes. The higher abundance of woody browse species on Lambert Station should provide greater protection from the sun and aerial predators, and greater food reserves in times of drought (Lim 1987).

Rainfall was only significantly related ($P < 0.05$) with vegetation abundance at half the release sites. In four other studies, by Lim *et al.* (1987), Robinson *et al.* (1994) and Copley and Alexander (1997), and Sharp and Norton (2000), no correlation between rainfall, vegetation abundance and associated population dynamics was recorded.

Two of three correlations between macropod spotlight surveys (Fig. 5.5) and macropod faecal pellet quadrat surveys (Table 5.1) at each release site were significant ($P < 0.05$). Both techniques indicated an increasing *M. r. erubescens* population throughout the study period. However, the two survey techniques were undertaken for different reasons. Spotlight surveys were used to count and remove exotic predators from the release sites as well as to assess population trends of macropods. Faecal pellet quadrat surveys were undertaken to investigate habitat use by *M. r. erubescens*, *P. x. celeris* and *C. hirus*, and not to infer density. Faecal pellets from the two latter species were rare in quadrats and thus not included. The technique was however suitable for *M. r. erubescens*, possibly due to the high abundance of the species. If the technique was to be adopted for studies where *M. r. erubescens* is less abundant, or for *P. x. celeris* or *C. hirus*, a much greater number of quadrats would be required.

Lambert Station has a larger area of suitable habitat for *P. xanthopus* than Aroona Sanctuary (2.3.4 and 2.4.4), a higher mean annual rainfall, and higher vegetation abundance, and thus has a greater potential carrying capacity for the species (Lim 1987; Lim *et al.* 1987). However, Lambert Station contains a higher density of *M. r. erubescens* than Aroona Sanctuary, reducing the available carrying capacity of the area for *P. x. celeris* through habitat and dietary competition (Dawson and Ellis 1979; Lim *et al.* 1992). Furthermore, free water is available without travelling at Aroona Sanctuary, potentially increasing the carrying capacity of the site (Lim 1987). Thus direct comparison of available carrying capacity between the two release sites is not possible.

ECOLOGICAL ADJUSTMENT TO THE WILD

6.1 Introduction

Survival and dispersal of released animals are the most commonly studied features of modern reintroductions and translocations (Miller and Ballard 1982; Fritts *et al.* 1984; Kleiman *et al.* 1986; Stanley Price 1986, 1989; Slough 1989; Ellis *et al.* 1990; Beck *et al.* 1991; Short *et al.* 1992; Soderquist 1994; Southgate and Possingham 1995; Short and Turner 2000; Southgate *et al.* 2000). However, few studies have compared results with pre-release or source populations and wild populations to determine ecological changes post-release and therefore most studies are uncontrolled (Slough 1989; Ellis *et al.* 1990; Short and Turner 2000; Southgate *et al.* 2000). Dispersal of translocated males is often reported to be greater than of wild males at the release site (Miller and Ballard 1982; Fritts *et al.* 1984; Ellis *et al.* 1990), but results are confounded by the presence of established wild male hierarchies that may promote dispersal by translocated animals. Previous research on reintroduced populations indicates that animals form home ranges of smaller to similar size than wild counterparts (Slough 1989) and within three months post-release (Short and Turner 2000), although both studies reintroduced wild-born animals. Stanley Price (1986) and Southgate and Possingham (1995) reported that reintroduced captive-bred animals continued to expand their home range from the release site for a number of years. However, home range size was not analysed in discrete time-units, and therefore it is unknown whether it became asymptotic during this time or how its location changed. Consequently, range establishment and duration are currently unreported for captive-bred animals.

Survival of reintroduced captive- or island-born animals to mainland sites is reported to be low (Griffith *et al.* 1989; Beck *et al.* 1991; Wolf *et al.* 1996; Fischer and Lindenmayer 2000), particularly among marsupials (Short *et al.* 1992; Dufty *et al.* 1994; Soderquist 1994; Gibson *et al.* 1994; Christensen and Burrows 1994; Short and Turner 2000). Although higher fecundity has been reported for reintroduced than source populations (Southgate and Possingham 1995; Short and Turner 2000; Southgate *et al.* 2000), survival of wild-born pouch young is often lower, contributing to rapid population extinction (Southgate and Possingham 1995; Southgate *et al.* 2000). Smaller marsupials (< 5 kg) have generally survived less than 12 months, with death of most animals by three months (Dufty *et al.* 1994; Soderquist 1994; Gibson *et al.* 1994; Christensen and Burrows 1994; Short and Turner 2000). Larger macropod marsupials (>5 kg) have fared no better (Short *et al.* 1992), although some encouraging results are emerging in Western Australia (Morris 2000). The principal reason for all failed marsupial reintroductions has been predation by exotic predators (Short *et al.* 1992). One previous mainland macropod reintroduction has been successful (24 years), that of the Brush-tailed Bettong (*Bettongia penicillata*) in Perup Nature Reserve, Western Australia, where founding stock were wild-born and transferred 13 km north within the same reserve. Thus no currently successful mainland reintroductions of captive-bred macropods have occurred in Australia (Short *et al.* 1992).

Despite four previous mark-recapture ecological studies on wild *P. x. xanthopus* by Lim (1987) at Middle Gorge, South Australia; on *P. x. celeris* at Idalia National Park, Queensland by Sharp (1994, 1997); on *P. x. xanthopus* at the Gap and Coturaundee Ranges, New South Wales by Sharp and Norton (pers. comm.); and on *P. x. xanthopus* at various colonies in the Flinders Ranges, South Australia by Lethbridge (pers. comm.), only the first and sections of the second have been published and are thus available for comparison. Lim (1987), Lim *et al.* (1987) and Robinson *et al.* (1994) (same data) reported *P. x. xanthopus* to be a highly fecund (68 pouch young in 3.5 years from 27 dams), continuous and non-seasonal breeder with a two-phase reproductive strategy. It seems that young females produce an even sex ratio of offspring for natal colony recruitment, but older females produce a male-biased sex ratio for dispersal that maintains a male-exchange system between colonies. Fecundity and population fluctuations were reportedly related to rainfall. Adult (3 to 6 years of age) survival was high (53% : 63%) throughout the 3.5-year study, but sub-adult and pouch-young survival was not measured. In other *Petrogale* species it has been found to be low, at c.12% (Delaney 1997).

Few (4%) adult *P. x. xanthopus* survived past 6 years of age for females and 6.5 years for males, and the known-to-be-alive (KTBA) population fluctuated between 11 and 20 animals throughout the study period despite 75 individuals being marked (Lim 1987).

Lim (1987) reported home range (90% habitation) and core area (50% habitation) of *P. x. xanthopus* in summer to be 210 ha and 41 ha () and 134 ha and 25 ha () respectively, and in winter to be 169 ha and 27 ha () and 164 ha and 27 ha () respectively. Thus the home ranges and core areas of male wallabies were larger than females in summer, leading Lim to suggest that females may be dominant over males in the selection of refuge sites. Home range analysis of *P. x. celeris* by Sharp (1994) was quite different, with this sub-species ranging over only 20-40 ha. Sharp (1994) suggested that the greater home range of the southern counterparts could be related to more marginal habitat, requiring a greater foraging area. Sharp's (1994) findings at Idalia National Park were more similar to those reported for other *Petrogale* species of 5-15 ha (Batchelor 1980; Short 1980; Horsup 1994).

Inter-colony dispersal occurrences and distances are reported to be low for *P. xanthopus*. Lim (1987) observed no inter-colony dispersal, and Sharp (1997) only 3% of males (1/38; distance 2.5 km). Female dispersal was not reported in either study. However, *P. xanthopus* density was high at both study sites and in surrounding colonies, with high levels of genetic homogeneity reported between colony groups (Lim 1987; Pope *et al.* 1996; Sharp 1997). Consequently, males may have traveled the short distance between groups undetected. Dispersal of wild *P. xanthopus* in a fragmented landscape has not been reported (although is currently being studied by M. Lethbridge in the Flinders Ranges, South Australia). Therefore maximum dispersal distances for the species are unknown. Dispersal by reintroduced marsupials has been reported to be high (Soderquist 1994; Southgate and Possingham 1995; Short and Turner 2000). Thus, the reintroduction of *P. x. celeris* provides a unique opportunity to test the dispersal ability of the species in non-contiguous habitat.

The aim of the post-release trapping and radio-telemetry studies on *P. x. celeris* at Lambert Station and *P. x. xanthopus* at Aroona Sanctuary was to examine the ecological changes that captive-bred animals undergo during adjustment to the wild. Survival, fecundity, population trends, home range and dispersal were examined. Findings are

compared with environmental factors, pre-release baseline data, wild *P. xanthopus*, other wild macropods and results from other reintroduced animals.

6.2 Methods

6.2.1 Trapping

6.2.1.1 Lambert Station

Treadle-operated wire cage traps (height 38 cm, width 38 cm, length 76cm) (Mascot Wire Works, Sydney) were used to recapture *P. x. celeris* reintroduced to Lambert Station (Fig. 6.1). Cage or 'hard' traps were chosen because of the continual reinvasion of predators at each site and the possibility that they would predate animals captured in 'soft' traps (Fig. 6.5). All traps were lined with high-density foam on the inside of doors, sides and roof to prevent injury to the animal once captured, and re-lined with 95% shade-cloth to protect the foam from damage by captured animals (Fig. 6.1). Traps were positioned perpendicular to pathways to allow access from both directions (Fig. 6.1) in areas frequented by *P. x. celeris*, established from radio-telemetry undertaken in the two months post-release (6.3.4). Final trap placement was guided by *P. x. celeris* faecal density in each identified area. Traps 1-6 at Sites 1 and 2 (Fig. 6.2 and 6.3) and traps 15 at Site 3 (Fig. 6.4) were positioned in November 1998, three-months post-release and two months before the first two-week recapture fieldtrip (January 1999), using traps previously used for trap training in captivity. The poor quality of figures 6.2-6.4 is due to the highest resolution aerial photography of Lambert Station being 1: 80,000; thus the 800 m-long Sites 2 and 3 are 1 cm long on the map. A reduction in quality was unavoidable in producing the enlargement required for figures 6.2-6.4. Traps were secured to the ground using two 20-30 cm steel or plastic pegs and locked open using D-shackles. Each trap contained two metal feeders, locked into position using slide wires to prevent spillage. Traps were free-baited with water, macropod pellets and universal bait (peanut butter and rolled oats) twice in November 1998, and three times during subsequent non-trapping fieldtrips to maintain animal interest. Traps were sprayed with diluted aniseed essence to attract distant animals, a technique found useful by former and current *P. xanthopus* ecologists (A. Sharp and M. Lethbridge pers. comm.). Traps 7-10 at Sites 1 and 2 and 6-10 at Site 3 (Crestware Industries, Adelaide) were positioned in March 1999 in the core areas of animals not captured during the January trapping. Trap positions remained fixed

throughout the study. Subsequent trapping fieldtrips were undertaken for two-weeks in July 1999, and January and July 2000, and three-weeks in January 2001 to remove radio-collars.

Figure 6.1 Cage trap (hard) used to recapture *Petrogale xanthopus celeris* at Lambert Station, Queensland.



Trapping at the three sites was undertaken on a rotational nightly basis until all adults known to be alive had been re-caught, or the site had been trapped for a minimum of four nights. Traps were baited and set within two hours of dusk and checked the following dawn. Empty traps were locked open to allow access to bait and water. Captured animals were grasped in the trap by the base of the tail, transferred to hessian bags and processed on site without the use of sedatives. Each animal was weighed, bled (3.3.1.4), measured (tail length, hindfoot, tibia, pouch depth/scrotum, forearm, ear and head length), ear tagged if wild-born, radio-collared (wild-born > 4 kg) or re-collared if required (typically every 12 months for released animals), examined for ecto-parasites and scored for condition. Pouch young were weighed only if thrown, measured (head, hindfoot and tail of in-pouch joeys, all measurements of thrown joeys), and ear marked using small holes, with tissue samples retained for genetic analysis. Birth dates for pouch young were determined from head, hindfoot and tail lengths using Poole *et al.* (1985) and Bach (1998), and later reassessed using growth curves developed for the *celeris* sub-species (Ch. 7). Pouches of all dams carrying large joeys (>13 week), regardless of whether they had been thrown, were sutured closed to prevent ejection upon release. Newly radio-collared animals were returned to the trap and left momentarily to ensure collar acceptance. All independent animals were injected intramuscularly with Vitamin-E-Selenium to prevent capture myopathy. Animals were normally released within 20

Figure 6.2 Site 1 radio-telemetry stations (S1-S3) and trap locations (1-10), Lambert Station, Queensland.

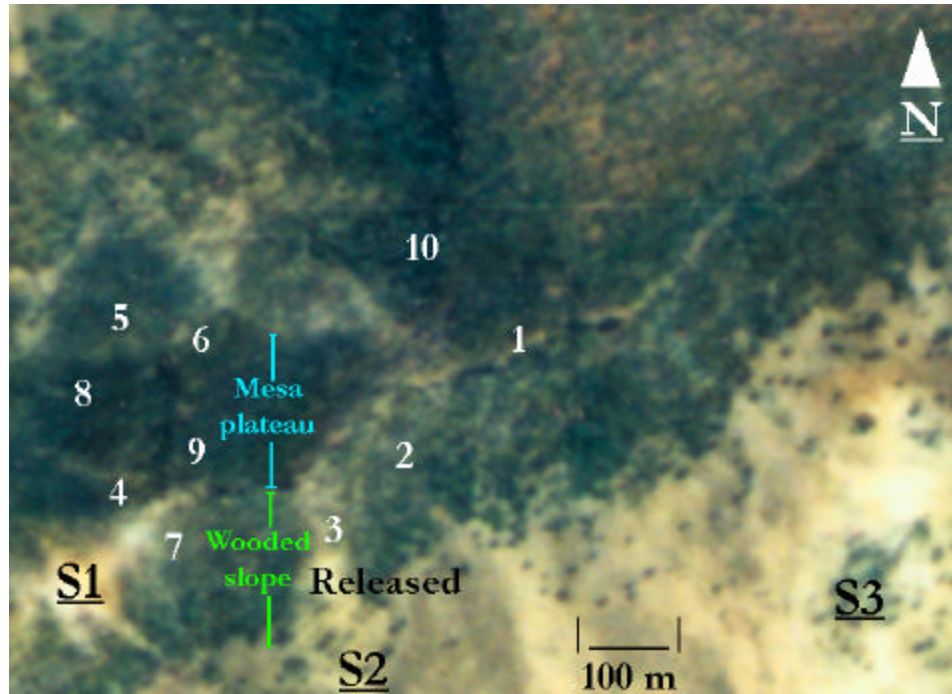


Figure 6.3 Site 2 radio-telemetry stations (S1-S4) and trap locations (1-10), Lambert Station, Queensland.

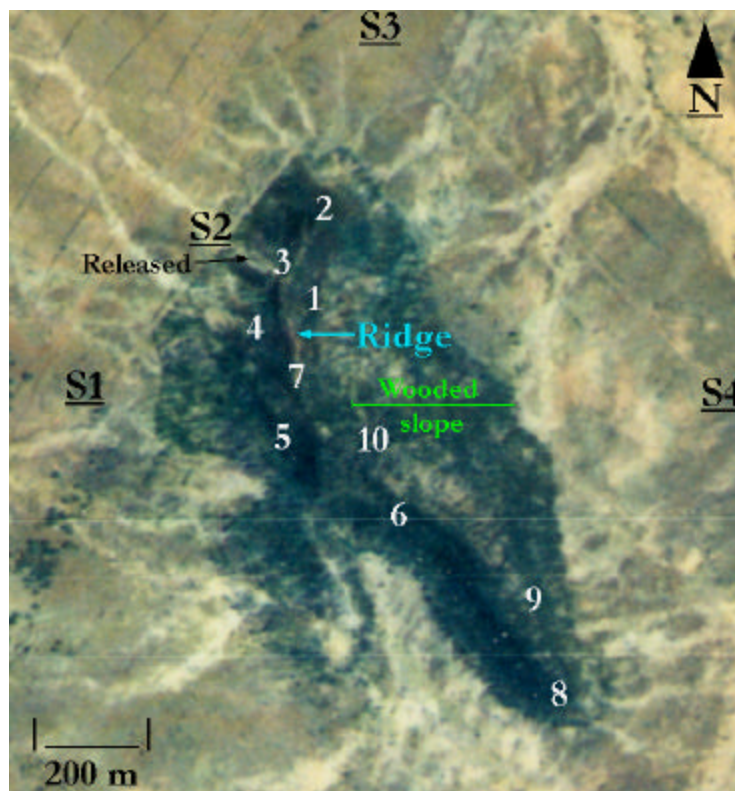
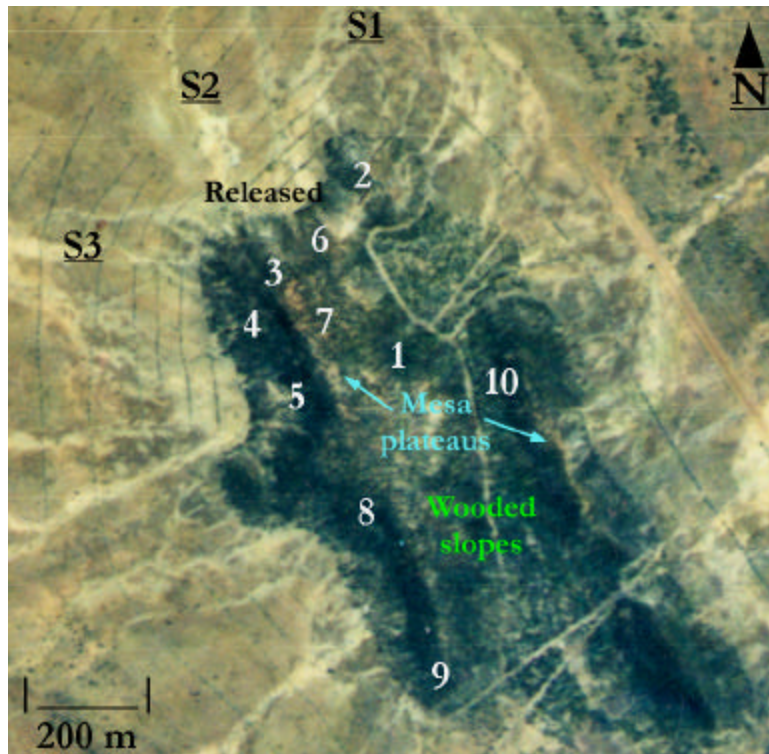


Figure 6.4 Site 3 radio-telemetry stations (S1-S3) and trap locations (1-10), Lambert Station, Queensland.



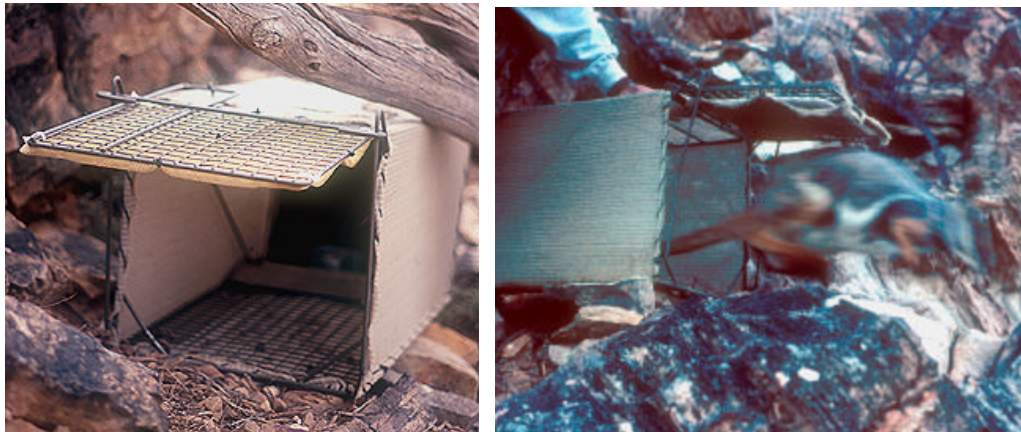
minutes of being transferred to the hessian bag. No animal injuries were recorded throughout trapping, and no overt stress was seen in animals through the recapture process (Fig. 6.1). No capture myopathy was recorded.

6.2.1.2 Aroona Sanctuary

Treadle-operated wire frame traps (height 38 cm, width 38 cm, length 76cm) (Crestware Industries, Adelaide) were used to recapture *P. x. xanthopus* reintroduced to Aroona Sanctuary (Fig. 6.5). Traps consisted of a reinforced wire frame surrounded by 95% shad-cloth with a fine metal weave. The trap door was reinforced with wire to prevent escape and consequently padded with foam. 'Soft' traps were chosen to minimize any chance of injury and because predators had been largely eradicated from the area. Furthermore, they provided a cooler environment for trapped animals in summer. Traps were positioned in January 1998 (traps 1-10) based on radio-telemetry results since the animal's release (September 1996), *P. x. xanthopus* surveys undertaken by the author in

December 1997, and *P. x. xanthopus* faecal density (Fig. 6.6). Traps 11 and 12 were added in October 1998 (Fig. 6.6). Trapping procedures followed those used at Lambert Station, but wild-born animals were not collared until they had reached 5 kg mass. The higher mass in South Australia was at the request of the Royal Zoological Society of South Australia.

Figure 6.5 Frame trap (soft) used to recapture *Petrogale xanthopus xanthopus* at Aroona Sanctuary, South Australia.



Trapping fieldtrips were undertaken for five consecutive nights starting in April 1998 (three months after trap placement) and continued in July, October and January of each year until January 2001 (n=12). An additional recapture fieldtrip was undertaken by Adelaide Zoo veterinarians in September 1998 to assess animal condition after the death of an animal through starvation. Traps were free-baited with macropod pellets at 1-2 week intervals throughout the project by I. Arthur, Leigh Creek resident. Higher traps (1, 2, 6 and 9) were not set during the extreme summer heat (>45°C) of January 1999 due to the time taken to reach the traps and the increased possibility of animal stress through dehydration. No animal injuries or myopathy resulted from trapping.

Figure 6.6 Location of traps in Aroona Sanctuary, South Australia.



6.2.2 Radio-telemetry

6.2.2.1 Lambert Station

All *P. x. celeris* released at Lambert Station were fitted with six-hour mortality-sensing radio-collars. Signal transmissions were checked at 1-2 week intervals by the author or S. Henshall to monitor for mortality signals (60+ beats per minute, instead of the usual 40). The date of any mortality signal was recorded and the carcass recovered as soon as possible by the author. Animals recovered within 24 hours, and for which a reason for death was not obvious, were frozen and transported to the Department of Veterinary Pathology at the University of Sydney for necropsy. For decomposed remains potential reasons for an animal's demise were determined from the location and condition of the carcass.

Single-bearing triangulation radio-telemetry was used to monitor animal movements throughout the project. Radio-collared animals were tracked from three fixed stations simultaneously, with the aid of hand-held UHF radios (ICOM, Transciever Services, Adelaide) used for communication between stations. Each tracking station consisted of a mobile telemetry receiver [two Biotelemetry TX-3 (Adelaide, Australia) and a Titley Electronics Regal 2000 (Ballina, Australia), with occasional use of a Telonics TR2 (U.S.A.)] combined with 3-element hand-held Yagi directional antennas [Titley

Electronics and Sirtrack (New Zealand)]. Where possible (Site 1), stations were positioned atop mounds to facilitate signal reception. Although high tracking towers with huts at each station at each site would have been preferable, this was beyond the means of the project.

Tracking commenced the morning after the animals were released and continued in 6 hour shifts per day (two hours per site) for 26 days post release. Compass bearings to each animal's location were recorded for each hour period in the day during each of the first two months (n=24 locations per month per animal), after which tracking was reduced per fieldtrip to accommodate other activities. No more than three consecutive bearings were recorded in a day, each separated by one hour, to avoid auto-correlation (Swihart and Slade 1985). One-hour separation between fixes was sufficient time for an animal to move a substantial distance, and thus animal locations were independent of time as determined using auto-correlation analysis in *Ranges V* (Fig. 6.7; Kenward and Hodder 1996). Locations were recorded as a magnetic north bearing from the station to the transmitter (animal) using a compass (Suunto A-1000) mounted to the antenna, a description of signal strength, and of the animal's suspected location on the hill. Descriptions were used to ensure correct recording of bearings once plotted. Original radio-telemetry bearings are presented in Appendix I.

Animal locations were grouped into five time frames post release; 1 month (August 1998), 2 months (September 1998), 6 months (November 1998 and January 1999), 12 months (March, May and July 1999), and 24 months (September and November 1999, January and May 2000). Each period was analysed separately to determine changes in the size and location of the core area (50% polygon) and home range (95% polygon) for each animal. The 50% polygon was chosen to represent core area (Kaufmann 1974) as an initial peak in area occurred at 10-12 or 50% of fixes, followed by a more rapid increase in range size as determined by incremental area analysis in *Ranges V* (Fig. 6.8; Kenward and Hodder 1996). Core area predominantly represents the diurnal habitation of each animal within each time frame, while home range encompasses areas used for foraging and interacting with other animals. The 95% polygon was used to represent home range size and to avoid seldomly used peripheral areas (Horsup 1994; Evans 1996). Incremental area analysis for all animals during Period 1 and Period 5 at each site indicate that 100% of area was generally achieved prior to the maximum number of animal locations. Thus,

the number of animal locations was sufficient to represent home range during each time period.

Figure 6.7 Auto-correlation analysis graphs indicating animal locations were independent of time by 60 min (top) or over a 48 h period (bottom). The latter is indicated by the majority of animal locations occurring above 1.96 on Schoener's index on the y-axis (Swinhart and Slade 1985).

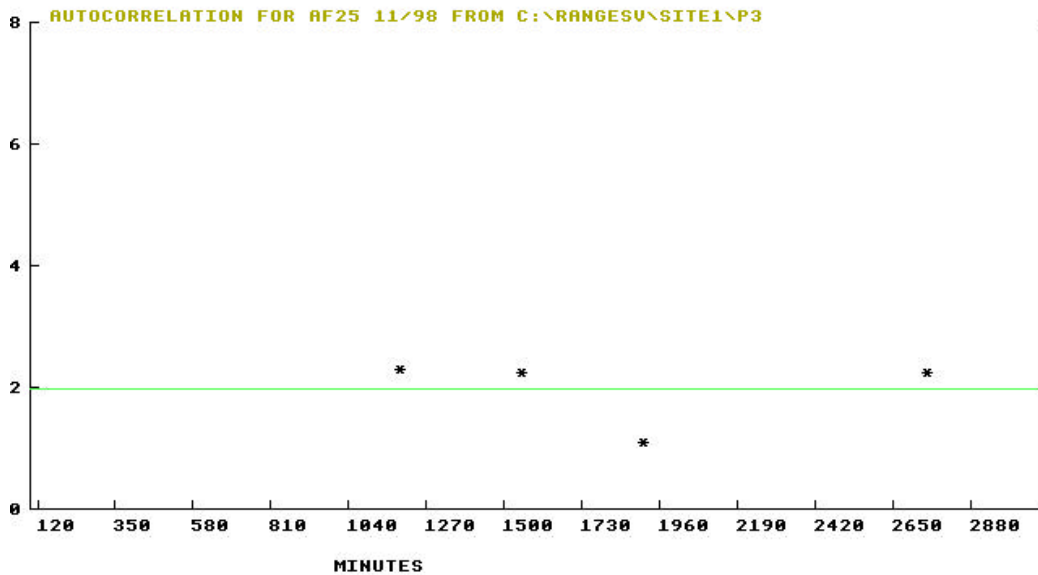
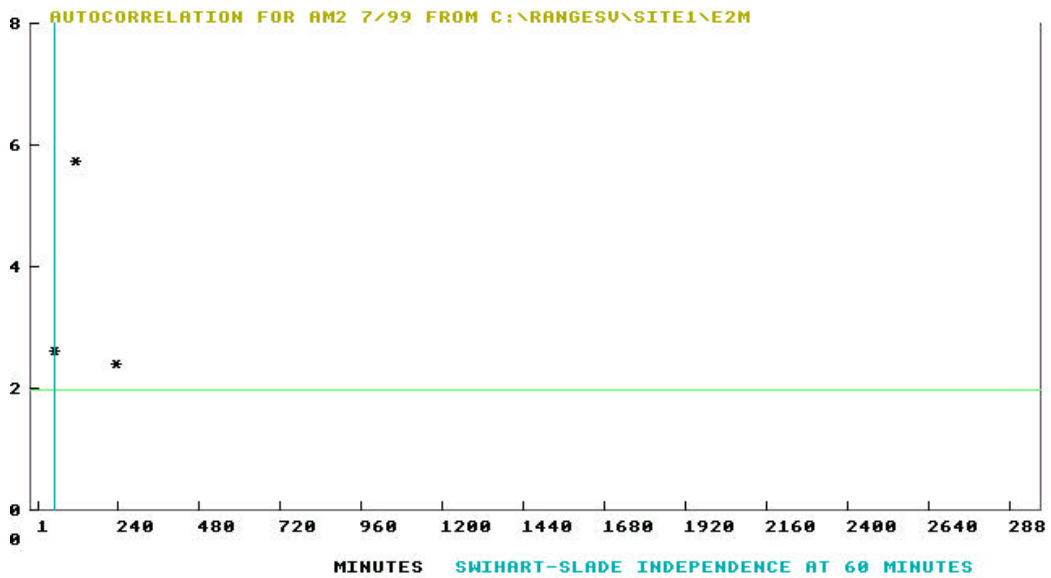
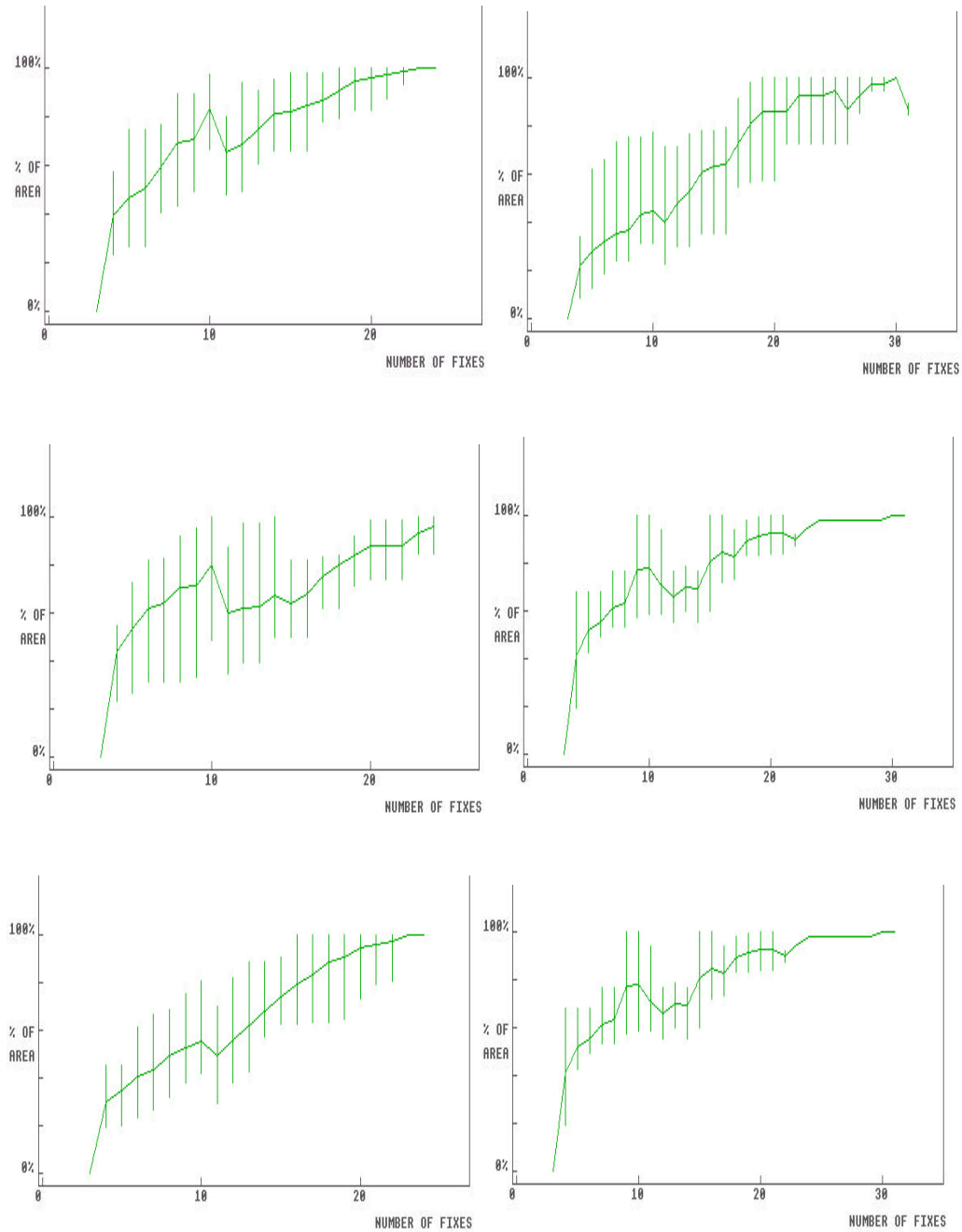


Figure 6.8 Means and limits incremental area analysis for all animals during Period 1 (left graphs; release to 1 month post-release) and Period 5 (right graphs; 12 to 24 months post-release) for Site 1 (top), Site 2 (middle) and Site 3 (bottom). Graphs indicate that 100% of area was generally achieved prior to the maximum number of animal locations during each time period; therefore the number of animal locations was sufficient to represent home range during each time period. An initial peak in area occurs at 10-12 or 50% of fixes, indicative of core area.



Each volunteer was trained for a minimum of 6 hours and tested for accuracy on numerous animals by the author before tracking commenced to minimise user error. Thus, tracking did not commence until user errors were negligible. Topographical error was assessed at each release site. Twenty transmitters were positioned at each site; one at each trap for reference points and 10 at random throughout the site. Transmitter positions were recorded using GPS (Scoutmaster, Trimble Navigation; post-signal scrambling), with magnetic north bearings calculated between each tracking station and the transmitter. After completing normal training, volunteers recorded the bearing too and the suspected position of each transmitter in blind tests. The difference between actual and recorded bearings was calculated and pooled for each site (n=60). Variations and standard deviations between actual and recorded bearings for Site 1 were $7^{\circ}\pm 13^{\circ}$, for Site 2 were $4^{\circ}\pm 5^{\circ}$, and for Site 3 were $0^{\circ}\pm 11^{\circ}$. Bearings were therefore slightly over-estimated at Sites 1 and 2, but the errors were relatively small.

All recorded magnetic north tracking bearings were adjusted to true north (photograph orientation) by the addition of 8° (Note, bearings in Appendix I are magnetic north as the adjustment occurred in *Tracker 1.0*). Adjusted bearings were triangulated onto an aerial photograph and converted to XY map coordinates (in metres) using *Tracker 1.0* computer software (Evans 2000; Fig. 6.9) as *Ranges V* does not triangulate compass bearings. Two bearings were used to indicate an animals' location if a third could not be obtained (Appendix I). Single bearings, although included in Appendix I, were discarded. The location of an animal was estimated from bearings using the geometric mean. Less than 10% of bearings were adjusted using the maximum likelihood method (Lenth 1981) by no more than the standard deviation measured above, and using suspected animal locations as a guide. Core area and home range size were estimated from animal locations by the minimum convex polygon method using *Ranges V* computer software (Kenward and Hodder 1996). Range centre was established from the arithmetic mean (Hayne 1949). Fix resolution, the mean overall distance at which two near animal locations are distinguishable from all tracking stations, ranged between 10 m if an animal was directly in front of a tracking station to 100 m if an animal was up to 1 km away, and thus was set at 50 m (Kenward and Hodder 1996). The mean number of animal locations for all periods was 24 ± 4 (S.D.).

Figure 6.9 Photo reproduction of computer screen showing *Tracker 1.0* triangulating compass bearings into XY map coordinates at Site 2. Yellow and red dots indicate animal locations and green crosses tracking stations. Direct image importation was not possible due to software incompatibility while the program is in operation.



Two-factor (site and time period) repeated-measure ANOVA for unequal replication was used to test for significant differences between sites and time periods among core area or home range size (Zar 1984; SYSTAT 1998). One-factor ANOVA was used to test for significant differences in core area and home range between sexes and re-introduced and wild animals. Simple linear regression was used to test for association between rainfall and core area/home range.

Overlap in core area and home range between all animals at each site was assessed using the range overlap analysis function in *Ranges V* (Kenward and Hodder 1996). The purpose of the analysis was to determine if and how post-release range overlap between animals, and thus potential interactions, changed over time. Overlap matrices were produced for Period 1 and Period 5 for both core area (50% polygon) and home range

(95% polygon). Overlap was categorized as male-male, female-female, male-female (based on the males core area/home range) and female-male (based on the females core area/home range), with differences between categories and time periods assessed. Differences in male-female versus female-male overlap were compared to determine if males had a higher overlap percentage with females than vice versa. Overlap within each sex was compared to determine if either sex had a higher rate of overlap. Pooled same sex results were compared to pooled opposite sex results to determine if overlap was higher within or between sexes. Results were analysed for significance using one factor ANOVA (Zar 1984).

6.2.2.2 Aroona Sanctuary

All *P. x. xanthopus* released into Aroona Sanctuary were fitted with mortality-sensing radio-collars. Signal transmissions were checked daily for mortality signals. Transmitters emitting a mortality signal were tracked down as soon as possible. Although substantial post-release tracking data have been collected at Aroona Sanctuary, their analysis is the subject of another study.

6.3 Results for *Petrogale xanthopus celeris*

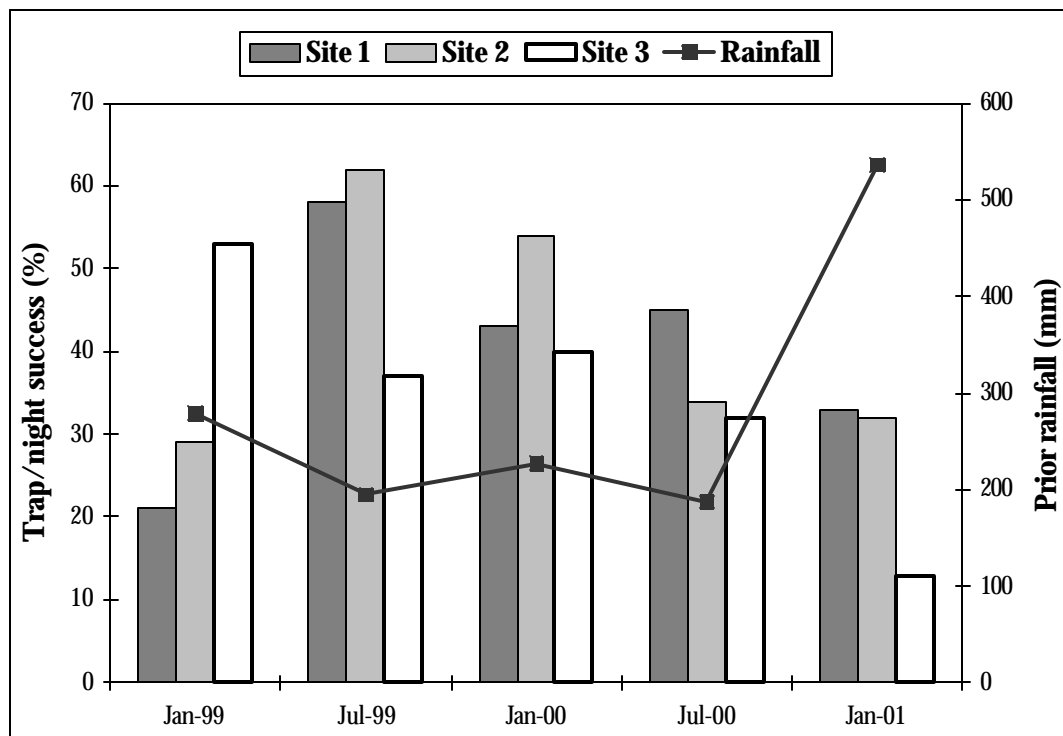
6.3.1 Trapping

A total of 168 *P. x. celeris* captures resulted from 448 trapping nights, an average of 38% trap/night success over the five trapping sessions. Released females were significantly more likely to be captured each session than released males (84%: 44%; $\chi^2=12.5$, $P<0.01$), although recapture rates per session were similar, with both sexes likely to be recaptured at least once more during the trapping session. An average of $73\pm 11\%$ of adults known-to-be-alive (KTBA) were recaptured each trapping session. Although used as an indicator of likely trapping success (with high rainfall thought to reduce trapping success due to increased food plant abundance), trapping success was independent of total rainfall for the six months prior to a trapping sessions when tested by simple linear regression ($F=4.69$, $F_{0.05(1,3)}=10.1$, $P>0.05$; Fig. 6.10). While not significantly different, released males were captured in more traps (5 ± 3) than released females (4 ± 2), indicating that males had slightly larger ranges, were more exploratory, or learnt not to re-enter

traps in which they had previously been captured. A similar result was found by Lim (1987).

Individual trapping records, and the status of each animal at the cessation of trapping, are shown separately for Site 1 (Table 6.1), Site 2 (Table 6.2), and Site 3 (Table 6.3). Population trends of KTBA animals from trapping and telemetry records are shown for each site and combined (Fig. 6.11). A combined potential population, assuming survival of animals with an unknown status at the cessation of trapping, is also indicated (Fig. 6.11). Both KTBA and potential population was found not to be independent of rainfall ($F=6.32$ and $F=0.78$ respectively, $F_{0.05(1),3}=10.1$, $P>0.05$) or vegetation abundance ($F=1.26$ and $F=1.31$ respectively, $F_{0.05(1),3}=10.1$, $P>0.05$) when tested using simple linear regression. The trapping technique was also found suitable for Swamp Wallabies (*Wallabia bicolor*) with 63 captures and Common Brushtail Possums (*Trichosurus vulpecula*) with nine captures.

Figure 6.10 Trapping success and rainfall for *Petrogale xanthopus celeris* at Lambert Station, Queensland. Trapping success was independent of prior rainfall when tested by simple linear regression ($F=4.69$, $F_{0.05(1),3}=10.1$, $P>0.05$).



Key to following animal numbering system and status for all Sites: pouch young were labeled with the dams identity followed by the birth number for each sex, then the sex. Thus 30(3M) is the third male pouch young of female 30 or 43(2F/1F) is the first female pouch young (second generation wild-born) of 43(2F). Animals for which the dam was unknown were designated the original site reference (Site 1: E experimental or Site 2: C1- control 1; refer to Chapter 5 for reasoning) followed by the order of capture. Thus E3M was the third male captured at Site 1 for whom the dam was unknown. Mortality sensors in radio-collars were used to assess survival of untrapped but radio-collared adult animals. This was not possible for juveniles, who were below radio-collaring mass of 4 kg.

Table 6.1 Trapping results for individual *Petrogale xanthopus celeris* at Site 1, Lambert Station, Queensland. Shading indicates initial release animal. Number indicate captures per trapping session; † indicates wild male; * indicates pouch young. Status at the cessation of trapping: A- alive, D- dead, U- unknown.

Animal	Sex	Release	Jan-99	Jul-99	Jan-00	Jul-00	Jan-01	Status
9		1	0	0	2(S2)	0	0	A
12		1	Decomposed- possibly predation					D
13		1	Fox predation					D
25		1	2	Fox predation				D
30		1	0	3	2	2	6	A
31		1	1	3	2	1	1	A
35		1	0	2	0	1	0	A
43		1	3	3	2	0	0	A
13(1M)		1*	Died with dam					D
43(1F)			3*	1	0	0	0	U
25(1F)			2*	Died with dam				D
E1M	†			1	3	1	1	A
E2M	†			2	0	1	1	A
35(1F)				2*	0	0	0	U
30(1M)				3*	0	0	0	U
43(2F)				3*	0	2	0	U
30(2M)					2*	1	0	U
43(3F)					2*	0	0	U
30(3M)					2*	0	0	U
43(2F/1F)						2*	0	U
E3M						1	2	A
35(1M)						1*	0	U
E4M	†						3	A
30(1F)							6*	A

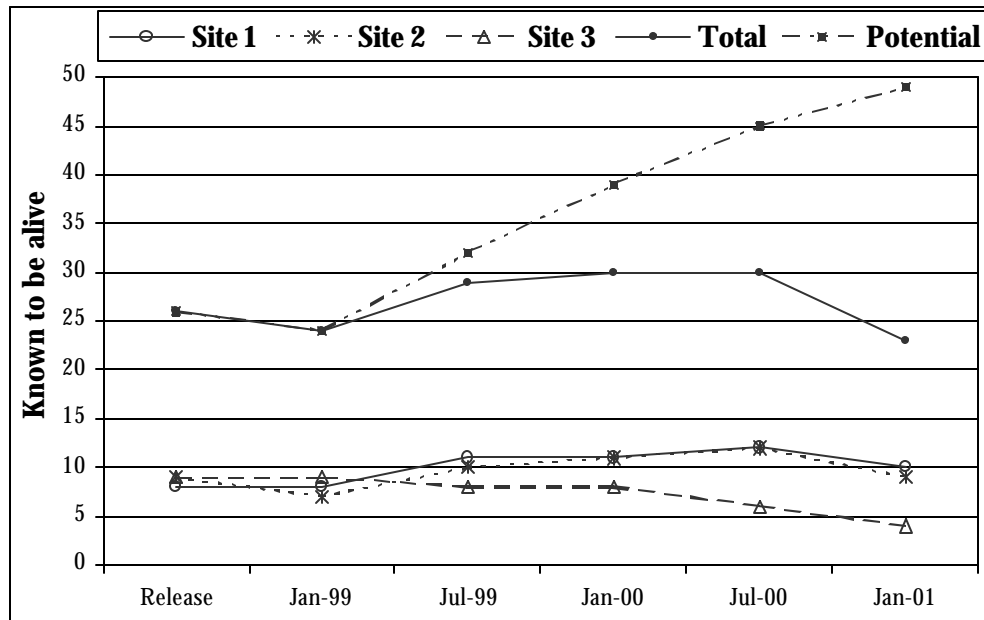
Table 6.2 Trapping results for individual *Petrogale xanthopus celeris* at Site 2, Lambert Station, Queensland. Shading indicates initial release animal. Number indicate captures per trapping session; * indicates pouch young. Status at the cessation of trapping: A- alive, D- dead, U- unknown.

Animal	Sex	Release	Jan-99	Jul-99	Jan-00	Jul-00	Jan-01	Status
2		1	0	0	0	0	0	A
6		1	Sand cholic					D
15		1	1	3	2	Decomposed		D
19		1	0	Decomposed- possibly predation				D
24		1	2	3	4	2	5	A
29		1	3	2	3	1	3	A
32		1	Necropsied- unknown					D
41		1	0	3	2	3	5	A
15(F)		1*	0	2	1	2	1	A
C1M1				1	2	2	3	A
29(1F)				2*	1	1	0	U
15(1F)				3*	1	1	0	U
41(F)				3*	0	0	0	U
29(2F)					3*	0	0	U
41(1F)					2*	0	0	U
15(1M)					1*	Died with dam		D
29(3F)						1*	0	U
15(F/1F)						2*	0	U
29(1F/1F)						1*	0	U
41(1M)						1*	0	U
29(1M)							3*	A
15(F/2F)							1*	A
41(2F)							5*	A

Table 6.3 Trapping results for individual *Petrogale xanthopus celeris* at Site 3, Lambert Station, Queensland. Shading indicates initial release animal. Number indicate captures per trapping session; * indicates pouch young. Status at the cessation of trapping: A- alive, D- dead, U- unknown.

Animal	Sex	Release	Jan-99	Jul-99	Jan-00	Jul-00	Jan-01	Status
5		1	Wedge-tailed Eagle Predation					D
14		1	Decomposed- not predation					D
17		1	3	3	2		5	A
22		1	0	0	0		0	A
27		1	1	2	1	2	Decomposed	D
28		1	1	3	2	2	0	A
34		1	2	Only collar found				U
36		1	1	2	2	Decomposed		D
27(M)		1*	1	0	0		0	U
36(M)		Aborted*						D
17(1F)			3*	1	1	1	1	A
36(1M)			1*	0	0	0	0	U
27(1M)				2*	Aborted			D
17(1M)				3*	0	0	0	U
36(1F)				2*	0	0	0	U
36(2F)					2*	Died with dam		D
27(2M)					1*	0	0	U
17(2F)					2*	0	0	U
27(1F)						2*	Decomposed	D
17(3F)						2*	0	U
17(2M)							5*	A

Figure 6.11 Numbers of *Petrogale xanthopus celeris* known to be alive at Lambert Station through radio-telemetry or trapping records. Potential population growth assumes survival of animals with an unknown fate, predominantly pouch young.



6.3.2 Fecundity

All sexually mature females (> 18 mth) captured post-release were carrying pouch young, feeding young-at-foot or both. The 100% fecundity recorded was slightly higher than that in captivity in Charleville (86%). Thirty-six pouch young (excluding the 3 :1 pouch-young released and 3 wild immigrants) from 10 released and three wildborn dams were recorded. The mean age of dams that produced female joeys was 33±11 months, and 39±11 months for male joeys. Wild-born dams first conceived at 13±1 months and 3.4±0.3 kg (n=3), and all produced female offspring. First conception was therefore earlier than that recorded in captivity (16±2 months and 3.8±0.4 kg; n=5). Although unity in captivity, the sex ratio of wild-born young post-release was 13 male to 24 female ($\chi^2=2.8$, $P=0.09$). Females at Sites 1 and 2 produced a female bias in the sex ratio (4 :7 , 4 :11 respectively), while Site 3 was closer to unity (5 :6).

All females had a 28±1 week (n=22) continual breeding cycle, indicating that pouch life (birth to permanent exit) for *P. x. celeris* was 189-203 days. Two joeys (5%) were aborted before full-term, as indicated by a less than 190-day dam cycle, one release [27(M)] and

one wild-born [27(1M)], and both from the same dam at Site 3 (mean age 11 weeks). Births were recorded in all months of the year (Fig. 6.12), with peaks in autumn and spring. The number of births between trapping sessions was significantly dependent on vegetation abundance ($F=10.5$, $F_{0.05(1),3}=10.1$, $P<0.05$), but not rainfall ($F=1.4$, $F_{0.05(1),3}=10.1$, $P>0.05$) (Fig. 6.13) when tested using simple linear regression.

Figure 6.12 Monthly frequency of *Petrogale xanthopus celeris* births at Lambert Station.

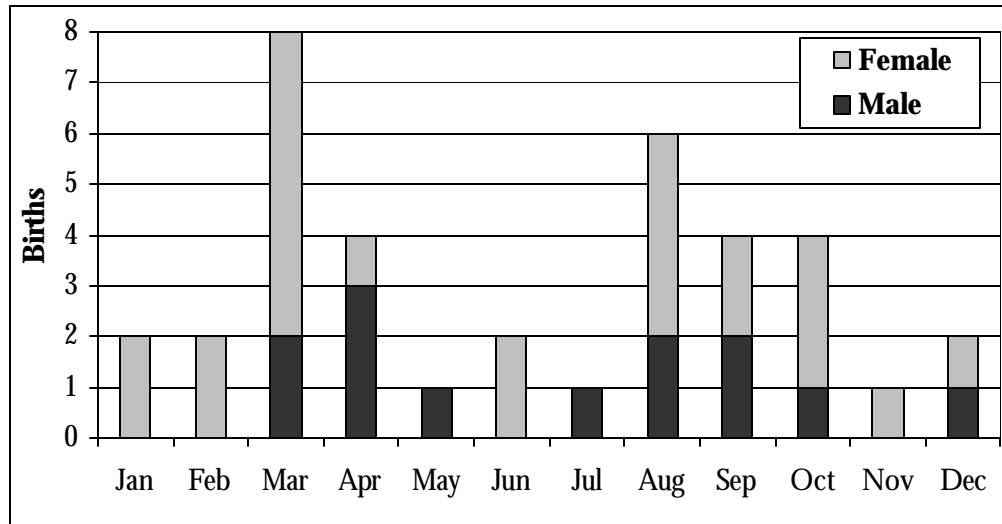
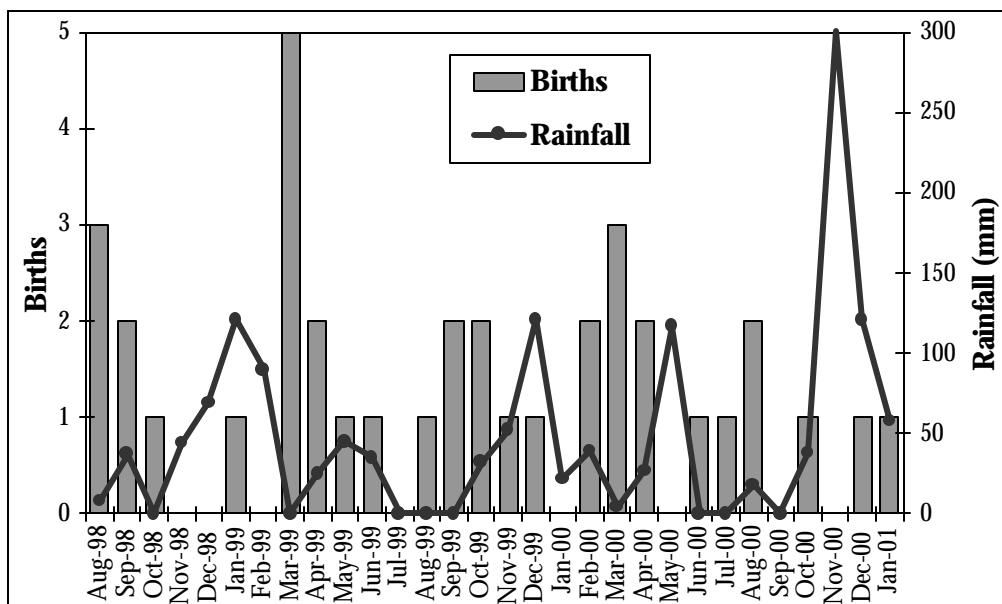


Figure 6.13 Relationship between *Petrogale xanthopus celeris* births and rainfall at Lambert Station. The number of births was not significantly dependent on rainfall.



6.3.3 Survival

Twelve of 24 reintroduced *P. x. celeris* were alive at the cessation of trapping, thus 50% of founders survived two and a half years post-release. In captivity, 7 of 41 adults (17%) died during the pre-release monitoring period (5 months). However, fewer deaths have been recorded in captivity since release (R. Griffith pers. comm.) and therefore mortality was similar between captive and reintroduced populations. Forty two percent of the 12 released animals that died were lost within 21 days of release. No significant difference was detected in the mean age, mass or condition of alive versus deceased animals at the time of release using one factor ANOVA (Table 6.4). The sex ratio of deceased animals was 3 : 9 (1:3). The sex ratio of released animals was similar, 9 : 15 (1:1.7). Thus 67% of released males and 40% of released females survived during post-release monitoring (Table 6.4). Causes of mortality are indicated in Table 6.4 and timing in Fig. 6.14. European Fox, *V. vulpes*, predation (F13 and F25 at Site 1) (Fig. 6.15) and suspected *V. vulpes* predation (M12 Site 1, M19 and F15 Site 2, and F36 and F27 Site 3) was the major cause of mortality, accounting for 58% of deaths (Tables 6.1-6.3; Appendix II). *V. vulpes* predation was suspected if the decomposed carcass was found on the plain in areas known to be used for foraging by the animal (from radio-telemetry), and bones contained bruising and teeth scrapings. A slight female bias (2 :5) was observed in *V. vulpes* predation. Mean age and mass of predated males was 20 months and 4.4 kg respectively, and predated females 41 months and 5.5 kg respectively. The mean age and mass of all males at death was 25 ± 3 months and 5.3 ± 0.8 kg, and 35 ± 10 months and 5.3 ± 0.7 kg for females; thus males were more likely to die as sub-adults and females as adults.

Other causes of mortality accounted for only one death each (8%). These include Wedge-tailed Eagle, *Aquila audax*, predation (F5, Site 3, Fig. 6.15), possible sand cholic indicated by stomach sand impaction during necropsy (red sand from captivity; F6, Site 2), inconclusively necropsied at the University of Sydney (F32, Site 2), decomposed but predation unlikely F14, Site 3- no puncture wounds, undisturbed), and one animal for which only the collar was found, with the collar and site presenting no evidence of predation (M34, Site 3). It is possible that the animal was able to remove the collar and remains alive, as occurred with C1M1 at Site 2. The stomach of the female that died from putative sand cholic was compacted with red sand consistent with that in captivity, and different from the black soil of Lambert Station. F6 was one of two females (F35 the

other who remains alive) that had a bloated abdomen and slight lung crackling during veterinary preparation (3.3.2.3). X-ray examination would have likely discovered the sand and prevented the animal's release.

Table 6.4 Survival of captive-bred release animals in relation to pre-release age, mass and condition. Initial age, mass or condition was not significantly related to post-release survival when tested using one factor ANOVA (Zar 1984).

Status	Site	Animal	Sex	Age (mth)	Mass (kg)	Condition (out of 5)	Cause of death
Alive	1	9		30	6.75	4	-
Alive	1	30		24	4.80	3.75	-
Alive	1	31		23	5.75	3	-
Alive	1	35		27	4.95	4	-
Alive	1	43		25	4.80	4	-
Alive	2	2		51	8.25	4	-
Alive	2	24		14	4.20	3.5	-
Alive	2	29		16	4.00	2.5	-
Alive	2	41		23	4.00	2.5	-
Alive	3	17		28	5.00	2.5	-
Alive	3	22		38	7.00	3.5	-
Alive	3	28		24	5.00	2.5	-
Mean±SD				27±10	5.4±1.3	3.3±0.6	
<1 mth	1	12		24	5.00	3	Decomposed- poss. pred.
<1 mth	1	13		27	4.50	2	Fox predation
<1 mth	2	6		38	5.60	2.5	Sand cholic
<1 mth	2	32		15	4.00	3	Unknown
<1 mth	3	5		31	5.50	3	W-T Eagle predation
<3 mths	3	14		33	6.00	3	Decomposed- not pred.
<8 mths	2	19		15	4.00	3.5	Decomposed- poss. pred.
<10 mths	1	25		29	4.95	3	Fox predation
<10 mths	3	34		22	5.60	3	Only collar found
<20 mths	2	15		48	6.50	3.75	Decomposed- poss. pred.
<18 mths	3	36		22	4.50	3.5	Decomposed- poss. pred.
<29 mths	3	27		15	4.00	3.5	Decomposed- poss. pred.
Mean±SD				27±10	5.0±0.8	3.1±0.5	

Survival of pouch young has previously not been reported for *P. xanthopus*. Survival of wild-born pouch young, excluding joeys killed within the dam (n=4) and pouch-young from the final trapping session (n=5), was 15% (2/13) for males and 10% for females (2/20). This is likely to be an underestimate as numerous wild-born young were caught independently as sub-adults and reproductive adults, although they were not captured during the final trapping session when status was assessed. Furthermore, young at foot were often sighted next to trapped dams, and known to be alive wild-born young were captured less regularly than released animals (Tables 6.1, 6.2, 6.3), indicating that wild-

born joeys had a greater wariness of being captured. Hence, 31% (4/13) of male and 30% of female wild-born joeys were captured independently as sub-adults or reproductive adults. Pouch young survival in captivity was recorded as 67% (6/8), although the estimate is likely over-estimated due to the short (5 months) baseline collection period in captivity, with over half (7/12) of the joeys being euthanased or hand-reared and dying before reaching full-term prior to the dam's release.

Figure 6.14 Causes of mortality for released *Petrogale xanthopus celeris* on Lambert Station. Twelve of 24 released animals remained alive at the cessation of fieldwork. Cause of death: A- Fox predation, B- Wedge-tailed Eagle predation, C- possible sand cholic, D- necropsied but unknown, E- decomposed- suspected fox predation, F- decomposed - not predation, and G- only radio-collar found.

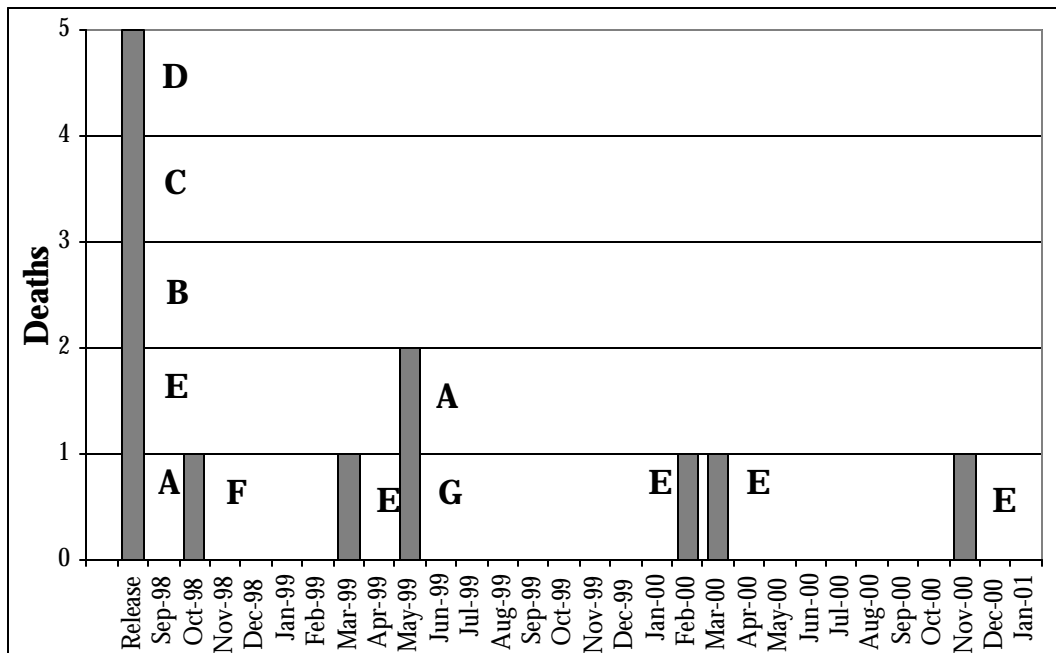


Figure 6.15 European Fox, *Vulpes vulpes* (top) and Wedge-tailed Eagle, *Aquila audax* (bottom) predation on reintroduced *Petrogale xanthopus celeris* on Lambert Station. Both females (F25 Site 1 and F5 Site 3 respectively) were photographed untouched within 24 hours of death. Note the preference of *V. vulpes* for thoracic viscera, and *A. aquila* for abdominal viscera and cranium (removed and found in nearby *A. aquila* nest; Parker 2000).



6.3.4 Home range

Radio-telemetry bearings are presented in Appendix I. A summary of core areas (50% polygon) and home ranges (95% polygon) at each release site, and for each time period post-release, is represented in Table 6.5. Mean core area steadily increased throughout the 24-month analysis period, while home range increased to a peak at 12 months before slightly decreasing by 24 months. Core areas were generally restricted to the hill while home ranges tended to include foraging sites on the surrounding plain. Although habitat analysis would confirm this, it was not possible due to fine-scale raster maps not existing for Lambert Station. Core area and home range size significantly differed between time periods ($F=13.3$ and $F=15.9$ respectively, $F_{0.001(1),4,69}=5.2$, $P<0.001$), but not between sites ($F=0.27$ and $F=0.43$ respectively, $F_{0.05(1),2,69}=3.1$, $P>0.05$) when tested by two-factor ANOVA for unequal replication (Fig. 6.16)(Zar 1984; SYSTAT 1998). There was no significant difference (one factor ANOVA, $P<0.05$) between the size of male and female core areas or home ranges during any time period. Lim (1987) reported the same finding for male and female *P. x. xanthopus* in winter but not in summer.

Table 6.5 Minimum Convex Polygon core area (50% polygon) and home range (95% polygon) of reintroduced *Petrogale xanthopus celeris* on Lambert Station. Values are means \pm S.D. in ha with sample size indicated in parentheses. Core area and home range size significantly differed between time periods ($F=13.3$ and $F=15.9$ respectively, $F_{0.001(1),4,69}=5.2$, $P<0.001$), but not between sites ($F=0.27$ and $F=0.43$ respectively, $F_{0.05(1),2,69}=3.1$, $P>0.05$) when tested by two-factor ANOVA for unequal replication (Zar 1984; SYSTAT 1998). Home range and core area size did not significantly differ between re-introduced and wild *P. x. celeris* during the same time period when tested by one-factor ANOVA ($F=1.27$ and $F=0.20$ respectively, $F_{0.05(1),1,14}=4.60$, $P>0.05$).

Post release		Site 1	Site 2	Site 3	Mean	Wild males at Site 1
1 mth	Core	3.3 \pm 1.7 (6)	1.7 \pm 0.3 (7)	1.5 \pm 0.1 (7)	2.1\pm1.2 (20)	
	Range	8.2 \pm 3.7 (6)	5.1 \pm 1.1 (7)	4.4 \pm 0.6 (7)	5.8\pm2.6 (20)	
2 mths	Core	2.2 \pm 0.3 (6)	2.3 \pm 0.9 (5)	2.2 \pm 0.5 (7)	2.2\pm0.6 (18)	
	Range	6.9 \pm 0.9 (6)	6.5 \pm 3.2 (5)	8.0 \pm 2.0 (7)	7.2\pm2.1 (18)	
6 mths	Core	2.5 \pm 0.7 (7)	3.2 \pm 0.9 (5)	3.2 \pm 0.6 (5)	2.9\pm0.8 (17)	
	Range	7.1 \pm 1.6 (7)	12.1 \pm 4.4 (5)	11.1 \pm 3.1 (5)	9.8\pm3.7 (17)	
12 mths	Core	5.3 \pm 2.9 (6)	6.6 \pm 1.4 (5)	3.6 \pm 1.1 (4)	5.3\pm2.3 (15)	
	Range	16.8 \pm 8.9 (6)	19.3 \pm 5.1 (5)	10.4 \pm 1.9 (4)	15.9\pm7.1 (15)	
24 mths	Core	6.0 \pm 4.7 (4)	6.4 \pm 4.2 (6)	5.7 \pm 1.7 (4)	6.0\pm3.6 (14)	7.3 \pm 3.5 (2)
	Range	14.1 \pm 7.3 (4)	14.6 \pm 8.4 (6)	15.8 \pm 3.7 (4)	14.8\pm6.6 (14)	20.6 \pm 8.3 (2)

Petrogale core area and home range size has been reported to be seasonal (Lim 1987; Horsup 1994). Average monthly rainfall for the 26 month period was 59 mm per month (n=4), for the 6-12 month period was 33 mm (n=6), and for the 10 month period from then until the cessation of tracking (22 months post-release) was 43 mm. Although low sample sizes, neither core area or home range size was significantly dependent on rainfall when tested using simple linear regression ($F=0.22$ and $F=0.23$ respectively, $F_{0.05(1),1,2}=18.5$, $P>0.05$).

Two wild *P. x. celeris* males immigrated to Site 1 by the second trapping session (discussed further in the next section on dispersal). Both animals were radio-collared and tracked during the 12-24 month time frame. Mean core area and home range of the two males was 7.3 ± 3.5 ha and 20.6 ± 8.3 ha respectively (Fig. 6.16). This finding supports that reported by Sharp (1994) of a home range of 20-40 ha for the sub-species in similar habitat at Idalia National Park. Comparisons of core area and home range size between re-introduced and wild *P. x. celeris* during the same time period and using a similar number of animal locations indicated no significant difference in core area or home range size ($F=1.27$ and $F=0.20$ respectively, $F_{0.05(1),1,14}=4.60$, $P>0.05$)(Fig. 6.16).

Mean overlap in core area and home range between all animals at each site during Period 1 and Period 5 is shown in Table 6.6. There was no significant difference between female-female, female-male, male-female or male-male overlaps in core area during either period or in home range during Period 1. Male-female home range overlap was significant greater than that of other categories during Period 5 ($F=2.74$, $F_{0.05(1)3,68}=2.74$, $P=0.05$). This indicates male-instigated overlap of female home ranges was the highest level of overlap within each colony, and thus potential level of interaction, and that males formed home ranges that on average overlapped the range of each female by 65%. There was no significant difference between Periods 1 and 5 for either core are or home range overlap categories. This indicates the level of overlap did not change throughout monitoring, consequently re-introduced animals established their range structure upon release.

Differences in male-female versus female-male overlap were directly compared to determine if male-instigated overlap with females was higher than vice versa. Differences were not significant for core area or home range during Period 1 ($F=1.50$ and $F=3.37$

respectively, $F_{0.05(1),1,62-65}=4.00$, $P>0.05$), but were by Period 5 ($F=7.22$ and $F=7.15$ respectively, $F_{0.05(1),1,40-43}=4.07$, $P<0.05$). This result confirms males had a higher overlap percentage with females than vice versa, and thus males developed core areas and home ranges to maximize potential interactions with females. Overlap within each sex was compared to determine if either sex had a higher rate of range overlap. No significant difference occurred between male-male rates of overlap and female-female rates of overlap for core area or home range in either time period ($F=0.49-1.48$, $F_{0.05(1),1,28-51}=4.03-4.20$, $P>0.05$). Similarly, no significant difference was detected between same sex overlaps compared to opposite sex overlaps ($F=0.04-2.65$, $F_{0.05(1),1,70-118}=3.92-4.00$, $P>0.05$). Both findings indicate males and females are distributed relatively evenly throughout the colony. Reintroduced *P. x aeleris* formed core areas and home ranges that overlapped with conspecifics an average of 43% and 54% respectively. Conversely, core areas formed by re-introduced *P. x aeleris* two years post-release were 57% free of conspecifics and home ranges 46% free.

Table 6.6 Site-pooled core area and home range overlap analysis for Period 1 and Period 5. Values are mean percentages \pm standard deviation, with number of animals indicated in parentheses. Overlap was categorized as male-male, female-female, male-female (based on the males core area/home range) and female-male (based on the females core area/home range), with differences between categories and time periods assessed by one factor ANOVA. Overlap categories did not significantly differ between category or period, except within Period 5 home range* ($F=2.74$, $F_{0.05(1)3,68}=2.74$, $P=0.05$). This result indicates male-female overlap is greater than any other category, with males forming home ranges that overlapped the range of each female by 65% on average.

		Female-Female	Female-Male	Male-Female	Male-Male
Core area	Period 1	34.1 \pm 35.0 (12)	41.4 \pm 29.9 (20)	51.0 \pm 34.0 (20)	43.9 \pm 30.1 (8)
	Period 5	41.1 \pm 32.3 (9)	28.2 \pm 24.2 (16)	51.5 \pm 33.7 (16)	50.2 \pm 41.0 (7)
Home range	Period 1	47.4 \pm 27.1 (12)	52.3 \pm 22.6 (20)	63.5 \pm 26.3 (20)	57.0 \pm 18.5 (8)
	Period 5	48.4 \pm 22.9 (9)	43.6 \pm 24.5 (16)	64.8 \pm 26.9(16)*	59.7 \pm 32.5 (7)

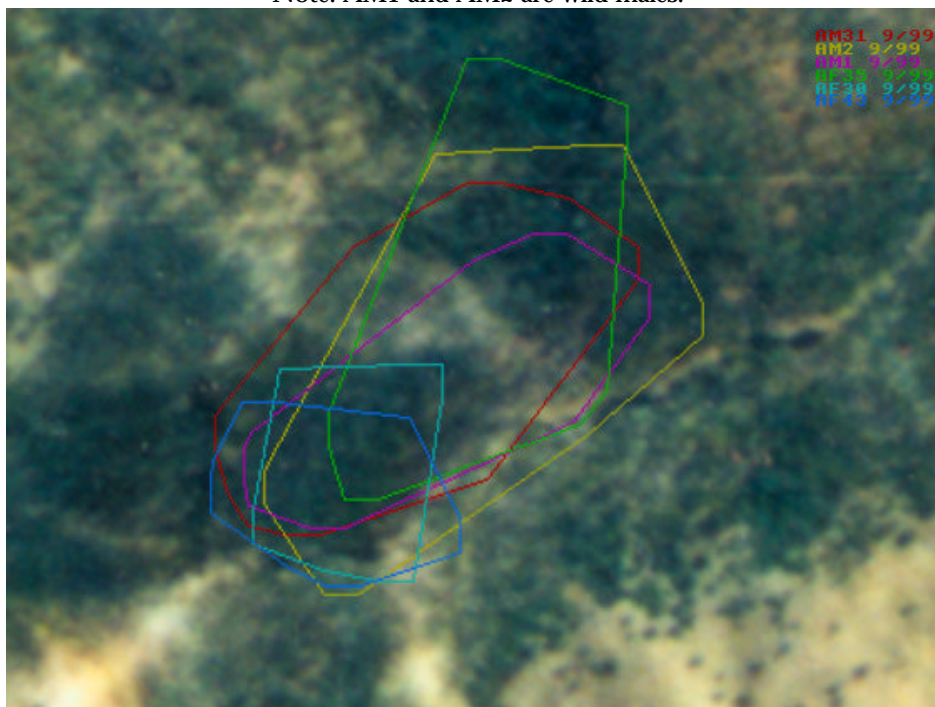
Figure 6.16 Increase in minimum convex polygon home range (95% polygon) of reintroduced *Petrogale xanthopus celeris* at Lambert Station superimposed onto site photographs. Range sizes are means \pm standard deviation.

SITE 1: 1 month post-release- 8.2 ± 3.7 ha



SITE 1: 24 months post-release- 16.3 ± 7.5 ha

Note: AM1 and AM2 are wild males.



SITE 2

1 month post-release

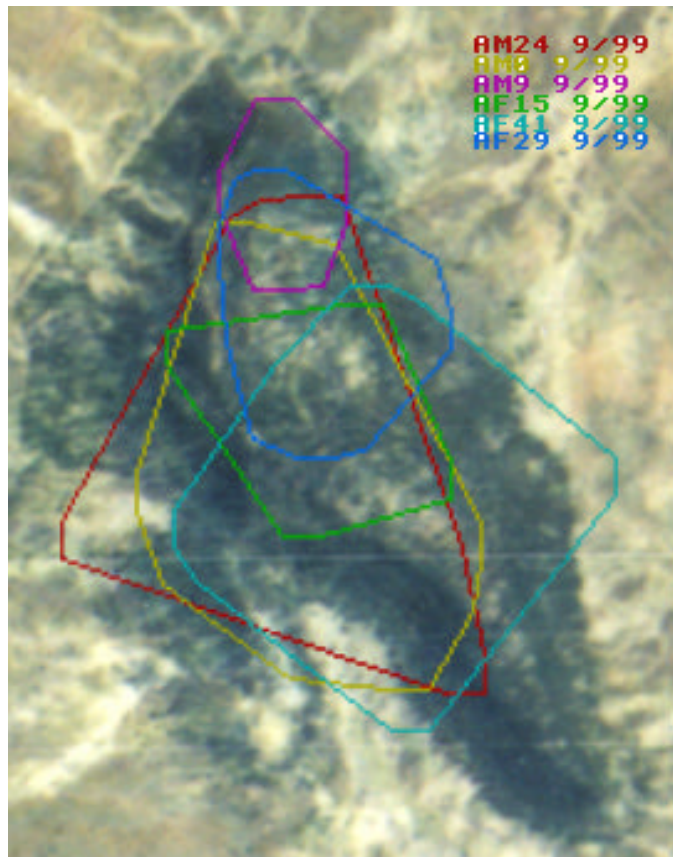
5.1 ± 1.1 ha



SITE 2

24 months post-release

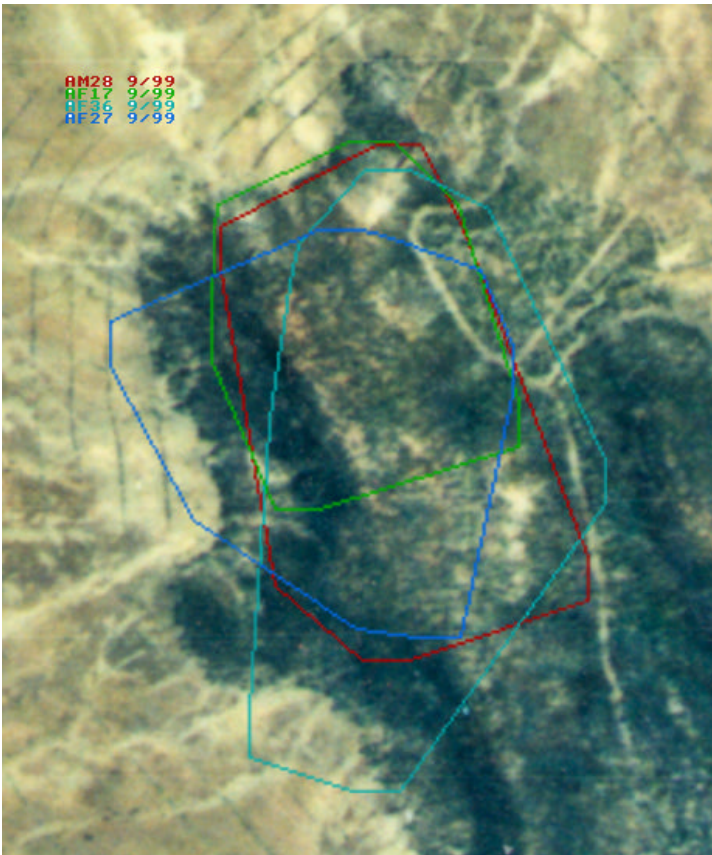
14.6 ± 8.4 ha



SITE 3
1 month post-release
4.4 ± 0.6 ha



SITE 3
24 months post-release
15.8 ± 3.7 ha

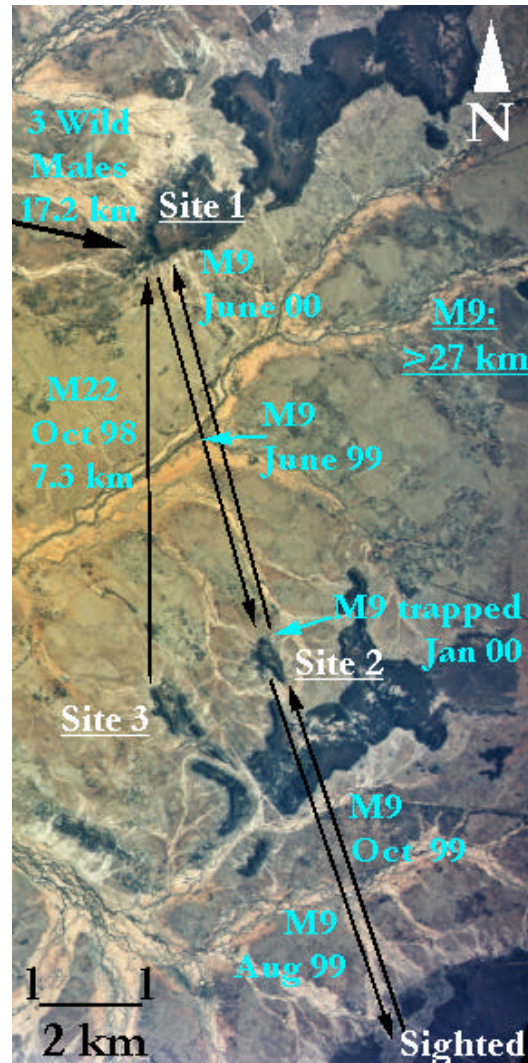


6.3.5 Dispersal

One of nine reintroduced males (M22) dispersed 7.3 km across open grassland between 2 and 4 months post-release. Another male (M9) undertook a minimum exploratory movement of 27 km over 12 months (Fig. 6.19). No female was recorded undertaking dispersal or exploratory movements. Male 22 (38 months old and 7.0 kg in mass) was released (August 1998) at Site 3 and remained at the site until late September 1998 (Fig. 6.17). The animal was tracked to Site 1 at the start of the November 1998 fieldtrip, and remained there until the cessation of fieldwork (Fig. 6.15). The animal left the only colony showing evidence of intra-specific aggression prior to release (3.3.2.2). Male 9 was released at Site 1 where it remained until late May - early July 1999. The animal was tracked to Site 2 in July 1999, a minimum distance of 6.5 km south across open grassland. Between July and September 1999 the animal travelled to hills on the neighbouring Acton Station (Fig. 6.19), a direct distance of 6.9 km south predominantly across open grassland. M9 was tracked and sighted by the author and volunteers in woodland at the base of the hill on numerous occasions. By January 2000 M9 had returned to Site 2, where it was captured twice (Fig. 6.16: 24 months post-release). It remained at Site 2 until April 2000 before returning to Site 1 by July 2000. All movements were across open grassland, as rocky substrate did not occur between sites (Fig. 6.19). M9 was 48 months old and 6.9 kg when captured at Site 2. All distances were calculated from GPS coordinates.

A minimum of three wild male *P. x. celeris* immigrated to Site 1 during post-release monitoring (Fig. 6.19). E1M and E2M were initially captured during the second trapping session on July 24, 1999, and subsequently during each trapping session thereafter (except E2M in trapping 3) (Table 6.1). E4M was captured during the final trapping session on January 27, 2001. Growth curves and mass records indicated that the three males were born prior to the reintroduction, with a mean age of 29 ± 5 months and mass of 6.7 ± 0.8 kg at first capture. E3M was first captured as a sub-adult (13 months and 4.0 kg) and likely born on site to unknown parentage. The closest known wild *P. x. celeris* colony is on Lynbrydon Station, 17.2 km west of Site 1 across open grassland or 19 km via suitable rocky habitat (Fig. 2.1). A closer wild colony to Site 1 may however exist. Immigration of wild males to Site 1 from Lynbrydon could not be foreseen, as previously reported dispersal distances for the species were less than 5 km (Lim 1987; Sharp 1997).

Figure 6.19 Inter-colony dispersal (M22), exploratory movements (M9) and immigration (wild males) of *Petrogale xanthopus celeris* on Lambert Station.



6.4 Results for *Petrogale xanthopus xanthopus*

6.4.1 Trapping

A total of 133 *P. x. xanthopus* captures resulted from 633 trapping nights, averaging 21% trap/night success rate over the 13 trapping sessions. Males were significantly more likely to be captured each session than females (66%: 45%; $\chi^2=6.5$, $P<0.025$), although both sexes had a near equal chance (41%: 38%) of being recaptured at least once during the same session. An average of $44\pm 16\%$ of KTBA adults were recaptured each fieldtrip. Overall trapping success per fieldtrip was independent of total rainfall ($F=2.34$, $F_{0.05(1)1,9}=5.12$, $P>0.05$) between trapping sessions or vegetation abundance at the time of trapping ($F=2.57$, $F_{0.05(1)1,9}=5.12$, $P>0.05$) when tested with simple linear

regression (Fig. 6.20). Males were captured in 4 ± 3 different traps and females 3 ± 2 , similar to *P. x. celeris* in Queensland. Reintroduced and wild-born *P. x. xanthopus* had an equal chance of recapture, in contrast to *P. x. celeris*, possibly due to the time lag (18 months) between reintroduction and the commencement of recapture, and because *P. x. xanthopus* was not trap trained. Trapping records and the status of each animal at the cessation of trapping are shown in Table 6.7. Population trends of KTBA animals from trapping and telemetry records indicated similar survival of males and females (Fig. 6.21). The decline in known and potential populations during 1998 coincided with drought. Potential population growth, although assuming the unlikely survival of all animals with unknown status, was significantly dependent on vegetation abundance at the time of trapping ($F = 10.22$, $F_{0.05(1),9} = 5.12$, $P < 0.05$) and rainfall in the three months prior to trapping ($F = 6.99$, $F_{0.05(1),9} = 5.12$, $P < 0.05$) when tested using simple linear regression. The soft traps resulted in 42 captures of Euros (*Macropus robustus erubescens*). This is in contrast to Lambert Station (Chapter 5), which had a higher abundance of *M. r. erubescens* yet none was captured in the hard traps used in that part of the study.

Figure 6.20 Trapping success, vegetation abundance and rainfall for *Petrogale xanthopus xanthopus* at Aroona Sanctuary, South Australia. Trapping success was not dependent on vegetation abundance or rainfall occurring between quarter-yearly trapping sessions.

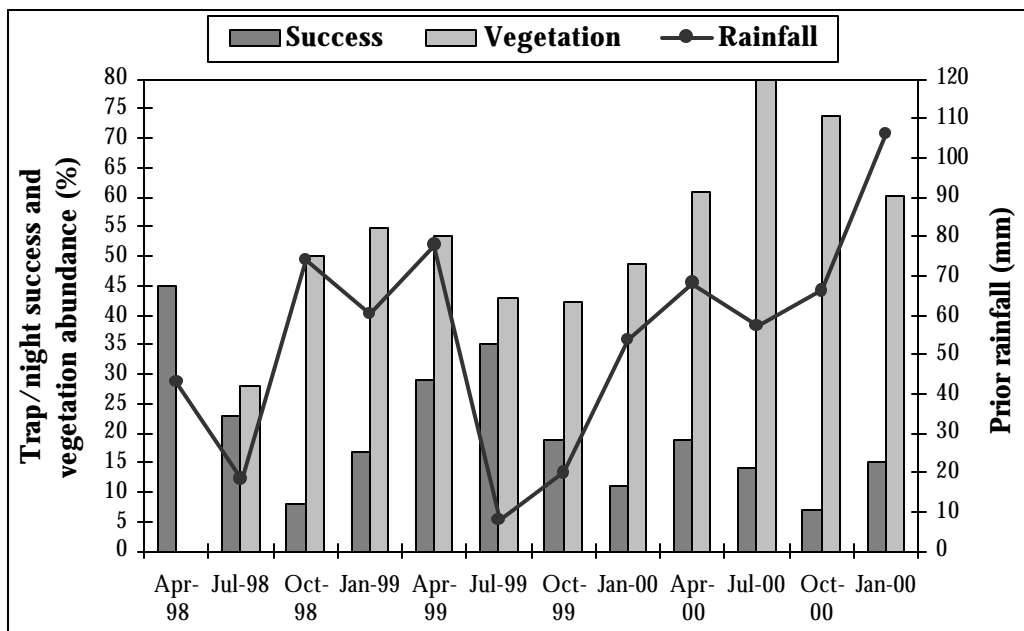
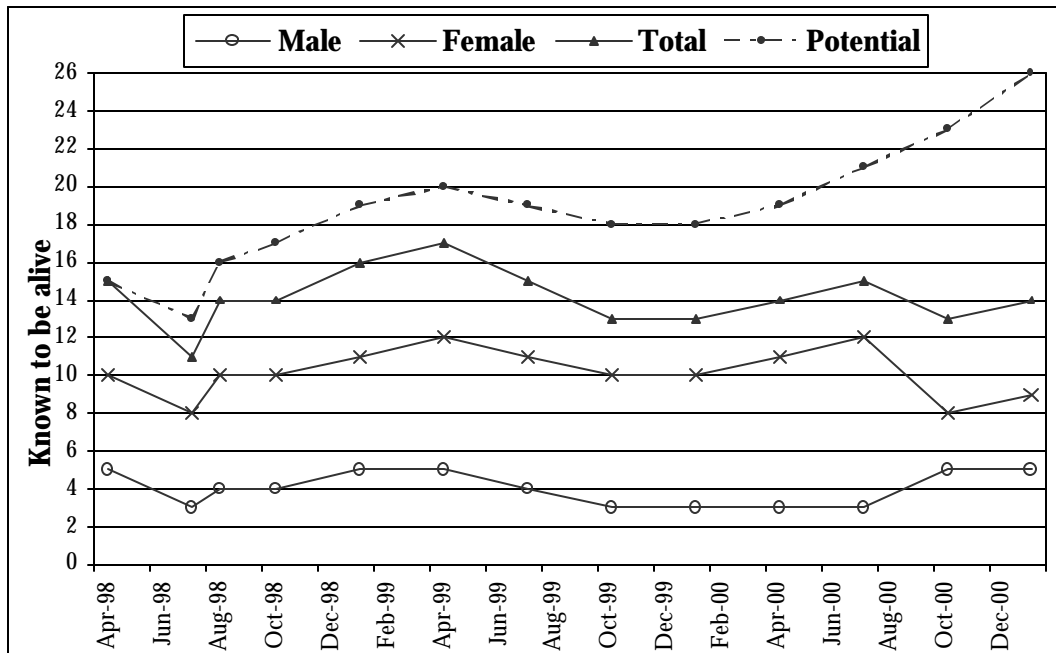


Table 6.7 Trapping results for individual *Petrogale xanthopus xanthopus* at Aroona Sanctuary, South Australia. Shading indicates initial release animal. Number indicate captures per trapping session; * indicates pouch young. Status: D- dead, A- alive, U- unknown. Mortality sensors in radio-collars were used to assess survival of untrapped but radio-collared adult animals. This was not possible for juveniles, who were below radio-collaring mass of 5 kg.

Animal	Sex	Apr-98	Jul-98	Aug-98	Oct-98	Jan-99	Apr-99	Jul-99	Oct-99	Jan-00	Apr-00	Jul-00	Oct-00	Jan-01	Status	
1															D	
2															D	
3		0	0	0	0	0	0	0	0	0	0	0	0	0	A	
4		2	0	0	0	1	Euthenased								D	
5		1	Decomposed-unknown													D
6		3	Decomposed-unknown													D
7		4	0	0	1	1	2	Decomposed- dietary related?								D
8		0	0	1	1	0	1	3	1	1	2	1	0	1	A	
9		1	0	0	0	0	4	3	4	1	2	2	2	1	A	
10		Decomposed-unknown													D	
11		Decomposed-unknown													D	
12		Decomposed-unknown													D	
13		Removed													A	
14		2	0	2	0	0	0	0	0	0	0	0	0	0	U	
15		3	1	2	0	0	1	2	0	0	1	0	0	0	A	
16		1	1	3	1	0	2	1	0	0	0	0	0	0	A	
17		1	2	2	0	1	1	2	1	1	1	1	0	0	A	
18		1	3	1	0	0	2	2	0	1	0	0	0	1	A	
19		2*	0	0	0	0	0	0	0	0	0	0	0	0	U	
20		1*	0	0	0	0	0	0	0	0	0	0	0	0	U	
21			2	0	0	0	0	0	0	0	0	0	0	0	A	
22				1*	0	2	0	3	1	1	2	1	0	2	A	
23				1	0	0	1	1	0	0	0	0	1	0	A	
24				1	0	0	0	0	1	0	Decomposed			D		
25					1	0	0	0	Decomposed- unknown						D	
26						1*	2*	0	0	0	0	0	0	0	U	
27						1*	4*	3*	1	0	1	1	0	0	U	
28							1	1	2	0	1	1	0	0	A	
29							1*	1*	0	0	0	0	0	0	U	
30										1*	0	0	0	0	U	
31											1	0	0	0	U	
32											1*	0	0	0	U	
33											1*	0	0	0	U	
34												1*	0	0	U	
35												2*	0	0	U	
36													2	0	A	
37														1	A	
38														1*	A	

Figure 6.21 Number of *Petrogale xanthopus xanthopus* known to be alive at Aroona Sanctuary through radio-telemetry or trapping records. The 1999 decrease in population occurred during drought.



6.4.2 Fecundity

Twenty-six pouch or independent young from a minimum of five released and four wild-born dams were recorded since release in a sex ratio close to unity (12 :14). The mean dam age was 40 ± 17 months for female joeys, and 46 ± 11 months for male joeys. Both ages were greater than but in a similar ratio to *P. x. celeris* at Lambert Station. Fifty-three percent of females more than 18 months of age were carrying pouch young, feeding young-at-foot or both when examined. The relatively low fecundity compared to *P. x. celeris* was due mainly to the cessation of breeding during the dry period of June 1999 - January 2000, which was followed by four births (1 :3) in April 2000 after February rainfall (Fig. 6.22). Sexual maturity of two regularly trapped wild-born females (23 and 28; Table 6.6) was 21 ± 3 months, with both producing female joeys. The annual number of births between 1997-2000 was independent of annual rainfall ($F = 4.74$, $F_{0.05(1)1,2} = 18.5$, $P > 0.05$), and births recorded during each trapping session was independent of vegetation abundance ($F = 2.93$, $F_{0.05(1)1,9} = 5.12$, $P > 0.05$) when tested by simple linear regression.

The timing of births was irregular at Aroona Sanctuary, occurring in eight out of the 12 calendar months. Peaks in fecundity occurred in autumn and spring (Fig. 6.23). The irregular capture of many dams, despite over twice as many trapping sessions as in Queensland, meant that pouch young development was difficult to monitor although a pouch life of *c.* 6 months was indicated.

Figure 6.22 Relationship between *Petrogale xanthopus xanthopus* births and rainfall at Aroona Sanctuary. The annual number of births was not significantly related to annual rainfall.

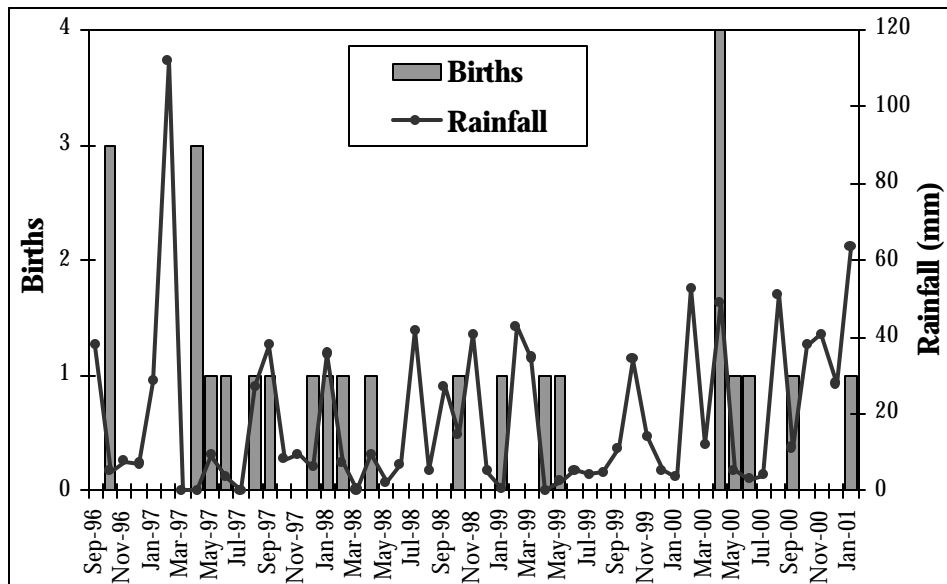
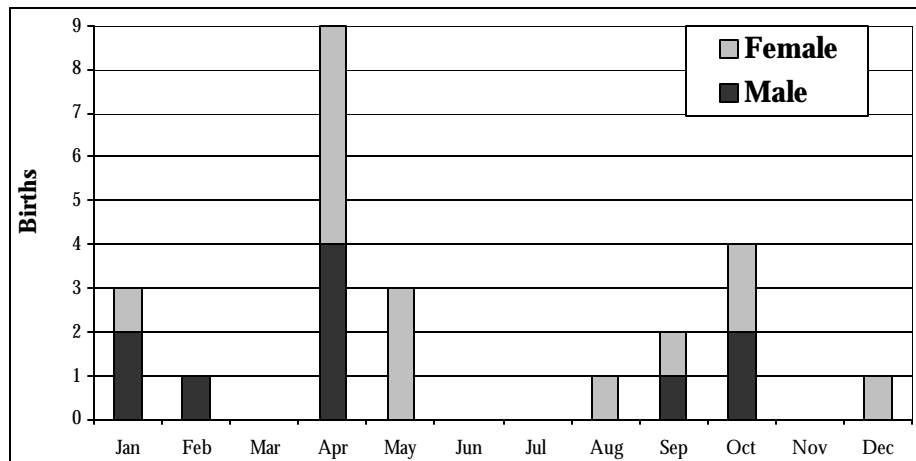


Figure 6.23 Monthly frequency of *Petrogale xanthopus xanthopus* births at Aroona Sanctuary. No significant difference between the sex ratio of offspring occurred (12 :14).

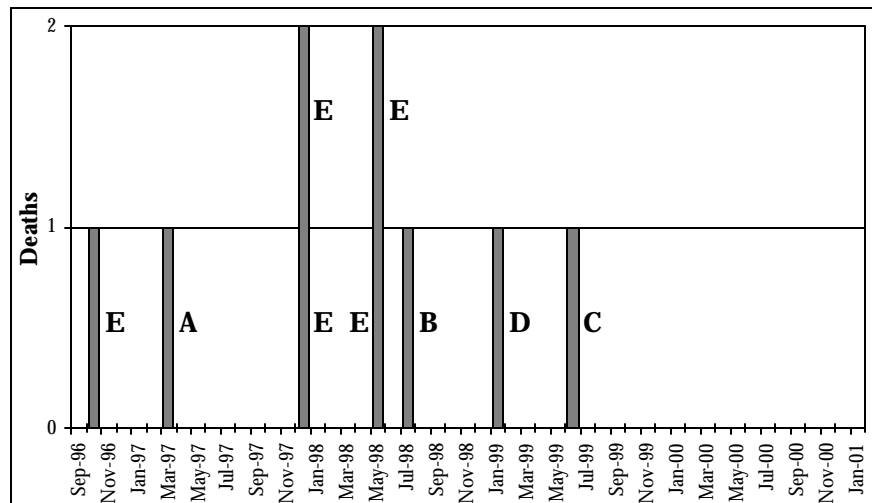


6.4.3 Survival

Three of 12 reintroduced *P. x. xanthopus* were alive at the cessation of trapping, more than four years post-release. As mean release age for *P. x. xanthopus* was 2.5 years, and 10% of wild *P. x. xanthopus* survive past 6 years of age (Lim 1987), survival has been high. Deaths have occurred regularly since release (Fig. 6.24), without the initial high mortality that occurred in Queensland and that has occurred in other macropod re-introductions (Short *et al.* 1992; Christensen and Burrows 1994; Gibson *et al.* 1994; Short and Turner 2000). No significant difference was detected through one-factor ANOVA in the mean age (alive- 28.4±5.7 months; dead- 30.8±2.9 months) or mass (alive-5.6±0.3 kg; dead- 6.0±0.4 kg) of alive versus deceased animals at the time of release (Table 3.8). The sex ratio of deceased animals was 2 :7 (1:3.5), not significantly different from the sex ratio of released animals (2 :10 or 1:5), although this result is limited by the small number of released males. Royal Zoological Society of South Australia veterinarians undertook carcass recovery and necropsy where possible. Necropsy was not possible on 67% of carcasses (M10, F11, F12, F5, F6, and F7) due to their decomposed state and thus potential causes of death were not reported (Fig. 6.24). Although inconclusive, F7 died during the 1999 drought and had suffered 50% mass loss with evidence of acute pulmonary oedema and liver congestion (D. Schultz pers. comm.). Analysis of stomach contents by the author revealed a large proportion of browse, particularly bark, indicating a poor quality diet (Lapidge 2000). F1 died during March 1997 from severe pneumonia, and F2 in the drought of 1999 (two weeks after F7) from starvation (D. Schultz pers. comm.). A female joey found alive in the pouch of F1 (F13) was hand-raised and remains alive at Adelaide Zoological Gardens. Numerous attempts to soft release the animal when a juvenile were unsuccessful. F4 was euthanased during the January 1999 trapping. The animal had a fractured pelvis causing haemorrhaging around the sciatic nerve and paralysis in the left leg. The injury was likely caused by a fall; a low red blood cell count, haemoglobin count, packed cell volume and platelet count indicated the injury had occurred 3-48 hours (D. Schultz pers.comm.) before the animal was bled. It was the oldest of the released animals and was not breeding when euthanased. Two wild-born adults (F24 and M25) also died during the project from unknown causes, although Dingo (*Canis lupus dingö*) predation was implicated in the latter. The mean age and mass (mass at last capture) of deceased males was 39±8 months and 6.3±0.6 kg, and females 45±12

months and 6.3 ± 0.4 kg respectively. Although older and heavier, the ratio between the two sexes is similar to that found for *P. x. celeris* at Lambert Station.

Figure 6.24 Fate of released *Petrogale xanthopus xanthopus* at the cessation of fieldwork. Three of 12 release animals remain alive. Cause of death: A- pneumonia, B- starvation, C- decomposed, possible dietary related, D- euthenased (dislocated hip), and E- decomposed- unknown.



Five male and six female wild-born pouch young (excluding pouch young from the final trapping session) were recorded during the study. One male (M22) but no females were known to be alive at the cessation of trapping, or one male (M22) and one female (F27) were captured independently. Survival of wild-born young first captured as sub-adults (12 months old and 2.8 kg; born post-release in late 1996 and 1997) was 75% and 71% respectively for four males and seven females at the cessation of trapping, with two known deaths (above) and the status of one female unknown

6.5 Discussion

Trapping success (trap/night) was 38% for *P. x. celeris* at Lambert Station compared to 21% for *P. x. xanthopus* at Aroona Sanctuary. Both success rates are higher than that previously reported for the species of 16% (Lim 1987). However, significantly higher trapping success (one-factor ANOVA $F=8.85$, $F_{0.01(1),1,16}=8.53$, $P<0.01$) was achieved for *P. x. celeris* that were trap-trained prior to release compared with *P. x. xanthopus* that

were not trap-trained. Furthermore, 73% of KTBA adults were recaptured each trapping session at Lambert Station compared to 44% at Aroona Sanctuary (one-factor ANOVA $F=11.46$, $F_{0.02(1),1,16}=8.53$, $P<0.01$). Hence, trap training significantly improved recapture success of released captive-bred wallabies, thus facilitated post-release monitoring, and is highly recommended for re-introductions in which detailed post-release monitoring will be undertaken.

Trapping trends were inconsistent between sites, with female *P. x. celeris* and male *P. x. xanthopus* more likely to be captured each trapping session. Each sex of both sub-species had an equally probability of recapture during the same trapping session. Male *P. x. celeris* and *P. x. xanthopus* were both caught in more traps than their female counterparts. Although this might indicate that males have a larger home range, this is not supported by radio-telemetry results. Alternatively, males may be more exploratory than females, or less inclined to re-enter the same trap.

Lim (1987) and Robinson *et al.* (1994) hypothesised that young *P. x. xanthopus* females produce an even sex ratio of offspring for recruitment into the natal colony, and that older females produce a male biased (2:1) sex ratio for dispersal between colonies; a two-phase reproductive strategy. A similar male-biased offspring sex ratio has been observed with *P. x. xanthopus* colonies in captivity (Rix 1978; Poole *et al.* 1985). In the current study, the mean dam age at which female young were produced tended to be lower than that for males of both sub-species, adding further support to this theory. However, a 2:1 female offspring sex bias was recorded for *P. x. celeris* at Lambert Station. Johnson (1989), Stuart-dick and Higginbottom (1989), Cockburn (1990) and Fisher (1999) have suggested the sex ratio produced by macropods is population density and resource dependent, and thus more likely to be related to carrying capacity of the habitat. The four previous studies have been conducted on either increasing and space-limited captive colonies (Rix 1978; Poole *et al.* 1985), or a large and increasing wild colony (Lim 1987; Robinson *et al.* 1994). Increasing colony density would reduce available resources of space (captive) or habitat (wild) for colony members and increase local competition. Under these pressures males may be produced to disperse and reduce competition for habitat. Where resources are abundant and population density is lower than carrying capacity, additional females may be produced to reduce dispersal and increase colony size. Therefore, rather than a

two-phase reproductive strategy, the sex of *P. xanthopus* offspring may be influenced by population density and resource competition.

The even sex ratio of offspring recorded at Site 3 on Lambert Station may be because that site was less suitable for *P. x. celeris*, as suggested by the higher mortality and lower population growth. The lower density of *M. r. erubescens* at Site 3 also suggests a lower carrying capacity for macropods than at Sites 1 and 2 (Ch. 5). *P. x. xanthopus* at Aroona Sanctuary also produced an even sex ratio of offspring; this may also reflect limited resources at the site, a result of lower rainfall and therefore vegetation abundance (Ch.5). The low fecundity of *P. x. xanthopus*, their cessation of breeding in drier times despite the availability of free water (Fig. 6.22), and the apparent bias toward production of females after rain adds support to this theory. Reintroduced *M. lagotis* also switched from an even sex ratio in captivity to a female bias post-release (Southgate *et al.* 2000). The implication for captive management of *P. xanthopus* is that to produce a female-biased sex ratio, desirable for rapid population expansion, small colonies need to be housed in large enclosures.

Fecundity and population growth were dependent on vegetation abundance, and to a much lesser extent (*P. x. xanthopus*) or not at all (*P. x. celeris*) with rainfall. This is possibly due to the greater abundance of browse in *P. x. celeris* habitat (Lim 1987; Chapter 5). As a result, fecundity of *P. x. celeris* may be less regulated by the availability of grasses and forbs than is the case with *P. x. xanthopus* (Copley and Robinson 1983). Vegetation abundance was significantly dependent on rainfall ($P < 0.05$) at only half the release sites (Chapter 5). Lim *et al.* (1987), Robinson *et al.* (1994) and Copley and Alexander (1997) concluded that fecundity and population fluctuations of *P. x. xanthopus* were directly related to annual rainfall in South Australia, but Sharp and Norton (2000) found no distinct relationship for *P. x. xanthopus* in New South Wales. None of the studies measured vegetation abundance. *Acacia* woodland occurs on hill slopes throughout the range of *P. xanthopus* in Queensland and New South Wales (Lim 1987), and the species is known to rely on it in times of drought (Dawson and Ellis 1979; C. Allen pers. comm.). In the present study, *P. x. celeris* maintained peak condition and continued to breed throughout dry winters, while *P. x. xanthopus* lost weight, ceased breeding and deaths occurred in the same seasons. Differences in browse abundance may therefore account for the differences between this and previous studies. That is, the drier and less sparsely

vegetated the area, the more likely *P. xanthopus* population fluctuations will be regulated by rainfall.

Peaks in fecundity occurred in autumn and spring with *P. x. celeris* at Lambert Station (Fig. 6.12), *P. x. xanthopus* at Aroona Sanctuary (Fig. 6.23), and wild *P. x. xanthopus* at Middle Gorge, South Australia (Lim 1987; Robinson *et al.* 1994). Combining the data from the three sites produces a summer: autumn: winter: spring birth ratio of 24: 47: 27: 39 (n=137), with chi-square analysis indicating a significant difference among seasons ($\chi^2=10.0$, $P<0.01$). Therefore, although *P. xanthopus* is described as a continuous breeder (Poole *et al.* 1985; Lim 1987; Robinson *et al.* 1994), seasonal peaks in autumn and spring are indicated throughout its range, regardless of rainfall seasonality (irregular at Aroona Sanctuary, summer at Lambert Station and winter at Middle Gorge). Thus the breeding pattern of the species is more accurately described as semi-seasonal. This pattern may be controlled by seasonal patterns in temperature, which are more consistent throughout the species' range. That is, dams may avoid extreme summer ($>40^\circ\text{C}$) and winter ($<0^\circ\text{C}$) temperatures (Figs. 2.2, 2.5), times of peak thermoregulation.

Pouch life duration of re-introduced *P. x. celeris* (189-203 days) was very similar to that reported for captive *P. x. xanthopus* (190-201 days) (Poole *et al.* 1985). Pouch young survival to independence was low for both sub-species. Few cases (2/37) of pouch young mortality were recorded for *P. x. celeris*, indicated by a pouch life of less than 190 days. However, only 30% of *P. x. celeris* and 20% of *P. x. xanthopus* pouch young were retrapped as adults. It is not known whether the few remaining *V. vulpes* and *F. catus* in the area were predated juveniles, or to what extent percentages are underestimates due to the lower recapture rate of wild-born and trap-shy juveniles (Sharp 1997a). Survival of juveniles could not be monitored remotely through mortality radio-collars, as per adults, due to animals being of insufficient mass to radio-collar. Thus, unless a juvenile was trapped during the last session they were considered dead. Although living juveniles were sighted, they could not be individually identified (not ear-tagged), nor counted as KTBA. Thus, it is known that juvenile estimates were under-estimated, particularly at the end of sampling when status was assessed (Nichols and Pollock 1983; Short and Turner 2000). Nevertheless, percent survival of the two sub-species is similar to that reported for reintroduced *M. lagotis* (25% survival to sexual maturity) (Southgate *et al.* 2000), and greater than that reported for wild Allied Rock-wallabies, *P. assimilis* (c.12% survival of

wild-born young) (Delaney 1997). Survival of sub-adults (mean 12 months old and 2.8 kg) was much higher, at 73%. This is higher than wild *P. x. xanthopus* (58%) of the same age (Lim 1987), and similar to *M. lagotis* (Southgate and Possingham 1995). Most deaths of *P. xanthopus* occur in the juvenile stage between weaning and sub-adulthood (7-12 months), possibly as a result of reduced foraging and poor predator avoidance ability (Southgate and Possingham 1995; Delaney 1997).

Survival of re-introduced *P. x. celeris* was 63% after 1 year, 54% after 2 years, and 50% after 2.5 years or the cessation of trapping. Survival of re-introduced *P. x. xanthopus* was 67% after 1 year, 42% after 2 years, 33% after 2.5 years, and is currently 25% 4.5 years post-release. For *P. x. celeris*, 67% of released males and 40% of released females survived during post-release monitoring. Although opposite to that reported by Lim (1987), where 53% of males and 63% of females survived adulthood (3-6.5 years), this result is consistent with Short *et al.* (1992).

Direct comparison of ecological adaptation between *P. x. celeris* released on Lambert Station and *P. x. xanthopus* on Aroona Sanctuary indicates that the former had higher founder survival after 30 months, despite the large initial loss of *P. x. celeris*. *V. vulpes* predation or suspected predation was the major cause of *P. x. celeris* mortality (Fig. 6.14). The direct cause of most *P. x. xanthopus* mortality is unknown, although drought and starvation were implicated in at least two deaths. Fecundity (100% and 53% respectively) and survival of pouch young to independence (30% and 20% respectively) were also higher for *P. x. celeris*. Neither parameter had previously been reported for wild *P. x. xanthopus*, although fecundity was relatively high (68 pouch young from 27 dams in 3.5 years) at Middle Gorge (Lim 1987). As fecundity was directly related to vegetation abundance at both sites, and Lambert Station has higher vegetation abundance, differences in fecundity and survival between the two sites are in the predicted direction.

The greater post-release fecundity of reintroduced *P. x. celeris* (86% captivity, 100% wild) and lower survival rates of wild-born young (67% captivity, 30% wild) are similar to percentages reported by Southgate and Possingham (1995) and Southgate *et al.* (2000) for reintroduced *M. lagotis*, and by Short and Turner (2000) for reintroduced *B. lesueur*. The lower age and mass of sexual maturity in wild-born females than their re-introduced parents (wild-born: 13 ± 1 months and 3.4 ± 0.3 kg; captive-born: 16 ± 2 months and

3.8±0.4 kg) were also reported for *B. lesueur* (Short and Turner 2000). The higher fecundity of re-introduced than source populations would allow for rapid population expansion and counteract lower survival.

Home range of re-introduced *P. x. celeris* peaked at 12 months (15.9±7.1 ha) post-release while core area continued to increase throughout the two-year tracking period. Both home range and core area did not significantly differ from that of wild counterparts by 12 months. They were also similar to those previously reported for the sub-species (20-40 ha; Sharp 1994). Short and Turner (2000) reported that re-introduced *B. lesueur* established home ranges by three months post-release. This shorter time period may be a result of *B. lesueur* being wild-born. Home ranges of *P. x. celeris* were similar to those of *P. assimilis* (11.9 ha; Horsup 1994), Brush-tailed Rock-wallabies, *P. penicillata* (4.8 ha and 15.2 ha; Batchelor 1980 from Horsup 1994, and Short 1980 respectively), and *W. bicolor* (14.6 ha; Troy and Coulson 1993), although significantly less than that of *P. x. xanthopus* (130-210 ha; Lim 1987). These differences support the suggestion by Sharp (1994) that *P. x. xanthopus* occupies marginal habitat in South Australia, thus requiring a larger home range. The core area to home range size ratio was also less for *P. x. celeris* (1: 2-3) than *P. x. xanthopus* (1:5 in summer, 1:6 in winter), indicating that *P. x. xanthopus* needed to range further from their daytime refuge to forage.

Quantitative core area and home range overlap has previously not been reported for *Petrogale* spp. Results from the current study indicate male-instigated overlap of female home ranges was significantly higher than that of female overlap of male ranges, male overlap of male ranges, or female overlap of female ranges. When directly compared, male-female overlap did not significantly differ from female-male overlap in core area and home range in Period 1, but did for both by Period 5. Thus, males developed core areas and home ranges to maximize overlap, and thus potential interactions with females. However, male overlap of female home ranges averaged 64% throughout sampling and did not differ between Period's 1 and 5, therefore indicating overlap structure was likely established upon release and prior to home range expansion.

Comparison of percentage overlaps between the same sex interactions versus opposite sex interactions, and between percentage overlaps within each sex indicated no significant difference. Both findings indicate males and females are distributed relatively

evenly throughout the colony. Core areas formed by re-introduced *P. x. celeris* two-years post-release were 57% free of conspecifics and home ranges 46% free. This finding supports the suggestion by Lim (1987) that *P. xanthopus* are predominantly asocial within the colony. A similar finding has also been reported for *P. penicillata* (Jarman and Bayne 1997).

On Lambert Station the release sites were isolated mesas surrounded by open plains. This landscape meant that clear telemetry reception over open plains was greater than 10 km at night, thus making it possible to monitor dispersal from the highest point on the station (Site 1). The long-distance dispersal of a released male *P. x. celeris* on Lambert Station (7.3 km from Site 3 to Site 1), and the 27 km exploratory movement of another male (Fig. 6.17), exceeded previous reports for the species and genus (Lim 1987; Sharp 1997). This can possibly be explained by the small release group size and the non-contiguous habitat. The former male was from the only colony that exhibited intra-specific aggression prior to release. The latter male returned to its release site (Site 1) after 12 months, having spent several months at both another release site (Site 2) and a neighbouring pastoral station. Interestingly, no dispersal or exploratory movement was recorded between the two closest release sites (2 km between Sites 2 and 3) as predicted, suggesting males show a preference for a site or members of the other colony rather than dispersal occurring over the shortest distance.

At least three wild *P. x. celeris* males emigrated to Site 1. Although the males may possibly have arrived earlier than the 1 year (E1M and E2M) and 2.5 years (E4M) indicated by trapping data, they were regularly recaptured once tagged, indicating no trap shyness, and were therefore more likely to have recently arrived. No wild females were captured, supporting the findings of Sharp (1997) that dispersal is limited to males. The closest known wild colony to the site is 17.2 km distant. The dispersal of released males indicates the possibility that wild males have traveled this distance. These dispersal events indicate that a meta-population has formed, an initial aim of the study. Although results may be confounded through re-introduction (Soderquist 1994; Southgate and Possingham 1995; Short and Turner 2000), they indicate that *P. x. celeris* has the ability to disperse over long distances, and is more likely to do so in non-contiguous habitat, a feature of much of the species' current range. Recent reports of dispersal over more than 7 km by Black-footed Rock-wallabies (*P. lateralis*) support this finding (Eldridge and Kinnear 1999)

Comparison with previous mainland macropod reintroductions indicates that both the Lambert and Aroona *P. xanthopus* reintroductions have been highly successful to date. Of 10 other mainland macropod re-introductions reported by Short *et al.* (1992), eight had failed by the same stage, despite often multiple releases of additional animals. Morris (2000) reported a 60% success rate (6/10) of pre-1998 mainland macropod reintroductions in Western Australia involving Brush-tailed Bettong (*B. penicillata*), Tammar wallabies (*Macropus eugenii*), Quokkas (*Setonix brachyurus*), and Black-footed Rock-wallabies (*P. lateralis*), however all used wild-born stock. Previously re-introduced populations of captive- or island-bred *S. brachyurus* (Short *et al.* 1992), *M. eugenii* (Short *et al.* 1992), Parma wallabies, *M. parma* (Short *et al.* 1992), Brush-tailed Phascogales, *Phascogale tapoatafa* (Soderquist 1994), Rufous Hare-wallabies, *Lagorchestes hirsutus* (Gibson *et al.* 1994), *B. lesueur* (Christensen and Burrows 1994; Short and Turner 2000), Golden Bandicoots, *Isodon auratus* (Christensen and Burrows 1994), and Eastern Barred Bandicoots, *Perameles gunnii* (Dufty *et al.* 1994) have resulted in high mortality, principally a result of *V. vulpes* and *F. catus* predation, with few animals surviving 12 months post-release and many less than three months. Consequently, the re-introductions of *P. xanthopus* are the only known currently successful mainland macropod releases using captive-bred stock (Short *et al.* 1992; Morris 2000).

The limited success of many past marsupial reintroductions is often associated with the species falling within the Critical Weight Range (CWR); marsupials of less than 5.5 kg in adult body mass that have fared worse since European settlement of Australia (Johnson *et al.* 1989). The primary causes reported to affect CWR species are changed fire regimes and *V. vulpes* predation (Johnson *et al.* 1989). Reintroductions specified in the last paragraph are all of CWR species, including the 60% success rate reported by Morris (2000). The mean adult body mass of *P. xanthopus* is reported as 6.5 kg and therefore beyond the CWR (Johnson *et al.* 1989). However, the mean mass of alive and dead *P. x. celeris* in this study (5.4 kg and 5.0 kg respectively; Table 6.4) falls within the CWR. Moreover, as *P. xanthopus* juveniles are independent at less than 2 kg (pers. obs.) they are as susceptible to predation as CWR species, with *V. vulpes* predation being the biggest threat to *P. xanthopus* (Honsby 1997; Lapidge 2002). Furthermore, Banks *et al.* (2000) has reported a highly significant impact of *V. vulpes* predation on Eastern Grey Kangaroo (*Macropus giganteus*) recruitment, a species with a mean adult body mass of 38.5 kg

(Johnson *et al.* 1989). Thus, comparison of the current results with that of recognized CWR species reintroductions is reasonable.

Despite the likely underestimation of KTBA animals (Nichols and Pollock 1983; Short and Turner 2000), re-introduced populations of *P. x. aeleris* and *P. x. xanthopus* have remained relatively stable since release with the actual population likely to be between potential and KTBA population estimates at each site (Fig. 6.11, 6.21). Initial high mortality of *P. x. aeleris* was confined to 21 days, animals formed home ranges indifferent to wild counterparts by 12 months, and population fluctuations have been similar to those of a wild *P. x. xanthopus* colony (Lim 1987). Survival rates of adults and sub-adults are currently high at both sites, and both populations have survived short-term droughts. Results indicate that captive-bred *P. xanthopus* readily adapt ecologically to wild conditions upon re-introduction, exhibiting similar fecundity, survival and habitat use to wild counterparts and other re-introduced marsupials that have been monitored (Southgate and Possingham 1995; Short and Turner 2000; Southgate *et al.* 2000).

PHYSIOLOGICAL ADJUSTMENT TO THE WILD

7.1 Introduction

Compared with ecological studies on re-introduced animals, physiological studies are rare (Stanley Price 1989; Lapidge 2000; Short and Tuner 2000; Southgate *et al.* 2000). For captive-bred animals to survive in the wild post-release they must adapt their dietary, water and energy requirements to suit the new environment. Moreover, animals must cope with marked seasonal fluctuations in resources, an unfamiliar process in captivity, while retaining homeostasis. The effects of re-introduction on physiological processes such as thermoregulation, osmoregulation, stress, growth and body condition are largely unknown, and thus the effect of re-introduction on captive-bred animals is undetermined.

Stanley Price (1986, 1989) has shown that captive-bred Arabian Oryx (*Oryx leucoryx*) can adjust to the harshness of the desert after long-term captivity. Behavioural changes that allowed animals to avoid temperature extremes and thermoregulate were evident. This included solely nocturnal behaviour in the hotter months, which changed to diurnal behaviour in the cold winter months. However, the animals were supplementary fed and watered to maintain condition.

Results from a previous study showed that captive-bred *P. x. xanthopus* at Aroona Sanctuary adapted their diet within 2-3 months to one similar to that of their free-ranging counterparts in the vicinity of the release site (Lapidge 2000). Animals raised on commercial macropod pellets, when shifted to native and exotic herbaceous plants at Monarto Zoological Park by reducing supplementary food, chose appropriate food plants for the region and with similar selectivity to wild *P. x. xanthopus* colonies (Copley

and Robinson 1983; Lapidge 2000). However, the study did not include the trapping of individual animals, and therefore it is unknown whether all animals underwent similar adaptation, or how dietary changes in turn affected other physiological parameters such as blood chemistry.

Studies of the blood chemistry, including haematology, biochemistry and vitamin status, of marsupials are rare and have not been reported previously for wild *P. xanthopus*. Changes in blood chemistry associated with re-introduction have not been reported for any species. Spencer and Speare (1992) conducted an extensive haematological study on wild Allied Rock-wallabies (*P. assimilis*), but the two species are among the most distantly related in the genus *Petrogale* (Eldridge and Close 1997). Blood chemistry is reported to vary significantly with age, sex, mass, diet and season (Sealander 1962; Parsons *et al.* 1971a; Shield 1971; Barnett *et al.* 1979a; Barnett *et al.* 1979b; Presidente and Correa 1981; Bradley 1990; Spencer and Speare 1992; Gaughwin *et al.* 1984), and may therefore be expected to undergo changes post-release. Baseline haematology, biochemistry and plasma vitamin E concentrations were established for both sub-species in captivity (Conaghty and Schultz 1998; Ch. 3). Emergent trends in blood chemistry of re-introduced *P. xanthopus* are reported in this chapter.

Vitamin E (α -tocopherol) is an important antioxidant linked with embryonic development and muscle degeneration (Rucker and Morris 1997). The role of vitamin E and selenium in marsupial nutrition has previously been reviewed (Hume 1999). The principal source of vitamin E is reported to be vegetable and seed oils and green leaves (Rucker and Morris 1997). Vitamin E deficiency has been linked to capture myopathy in macropods (Kakulas 1961, 1963a, 1963b; Munday 1988). Despite its importance in preventing or reversing myopathy (Kakulas 1961, 1963a, 1963b), plasma concentrations of vitamin E associated with the prevention of myopathy have not been reported for macropod species. Stress and diet are both known to affect plasma vitamin E concentrations in macropods (Kakulas 1961, 1963a, 1963b; Munday 1988; Hume 1999), but the relative importance of each factor is unknown. The effect of vitamin E status on reproduction in macropods is also unreported.

Previous investigations into body condition of macropods have used body mass coupled with a body measurement, normally tibia or hindfoot length or tail circumference, as

indices of animal condition (Bakker and Main 1980; Catt 1981; Moss and Croft 1999; Short and Turner 2000). Such indices rarely take into account the suite of health variables seen in captured animals. Coat condition, parasite loads, stress, lethargy, diarrhoea and reproductive status of sexually mature animals also are indicative of overall animal health. A condition score index was developed for the present study, based on one used by the Royal Zoological Society of South Australia, to assess individual animal condition for comparison with conspecifics and over time.

Although the accuracy of macropod growth equations is often questioned (Wood *et al.* 1981; Poole *et al.* 1982), they do provide a useful tool for approximating the age of animals in the field for analysis of population structure. Growth of *P. x. xanthopus* has previously been studied in captivity (Poole *et al.* 1985; Bach 1999). Despite two previous ecological studies examining growth in wild *P. x. xanthopus* (Lim 1987; Robinson *et al.* 1994) and *P. x. celeris* (Sharp 1994, 1997), growth rates have not been reported and measurement data are unavailable for comparative purposes. The limited period of growth data collection for captive *P. x. celeris* before release was insufficient to determine growth rates for the sub-species, and whether they deviated from captive *P. x. xanthopus*. Sharman *et al.* (1995) reported greater sexual dimorphism in *P. x. celeris*, with males obtaining 12 kg and *P. x. xanthopus* males obtaining 11kg while females of both sub-species obtained 6 kg, however the source of the data is unknown. Thus, developmental differences between the *P. xanthopus* sub-species, and captive and wild animals are undetermined.

The applicability of growth curves developed on captive animals to wild animals or vice versa is a matter of contention. Several authors (Shield and Wolley 1961; Sadleir 1963; Murphy and Smith 1970; Delaney and De'ath 1990) reported that growth rates of captive and wild macropods, predominantly pouch young, of the same species were similar, while others (Ealey and Mains 1967; Inns 1982; Taylor and Rose 1987) reported retarded growth of wild macropods when compared with captive animals maintained on a consistent commercial diet. The only study on captive and wild animals of a *Petrogale* species, *P. assimilis*, indicated no significant difference between growth of captive and wild pouch young, although there was a significant difference between adult individuals in each population (Delaney and De'ath 1990). It is generally agreed that if the nutrition of a wild population is not a limiting factor during the time of development, growth rates

of wild macropods should be similar to captive conspecifics (Shield and Wolley 1961; Sadleir 1963; Ealey and Mains 1967; Murphy and Smith 1970; Inns 1982; Delaney and De'ath 1990).

Knowledge of the physiological adaptation of captive-bred animals to the wild is integral in addressing animal welfare considerations of re-introduction projects. This chapter examines changes in growth rates, haematology, biochemistry, vitamin E and general condition of *P. xanthopus* with time upon re-introduction.

7.2 Methods

7.2.1 Growth rates and age determination

The birth date of all pouch young and juveniles (<12 months of age) was determined using head, hindfoot and tail measurements against growth measurements of Poole *et al.* (1985) and Bach (1999), unless a birth date was known (birth occurred during trapping). The average age from the three measurements was used unless a discrepancy of >10 days was discerned, in which case head length was used, based on its higher accuracy reported by Poole *et al.* (1985). Body mass and head, ear, forearm, tibia, foot and tail measurements from each capture over the following three years were used to form scatterplots for mass and each body measurement for each sex and each subspecies. Logarithmic growth curves were plotted, with the resulting natural logarithm and percent variance accounted for by the growth model (R^2).

Because the study took place predominantly post-release for *P. x. celeris* and completely for *P. x. xanthopus*, repeated measurements for each animal depended on regular recapture. Measurements from 8 male and 15 female *P. x. celeris* were used to establish growth curves, with a mean of 2.7 ± 2.3 (range 1 to 8) measurements recorded for each animal. Measurements from 7 male and 11 female *P. x. xanthopus* were used, with a mean of 3.2 ± 2.5 (range 1 to 9) measurements per animal. The greater mean number of measurements for *P. x. xanthopus* reflects the greater number of trapping sessions. Few records of animals older than three years were obtained. Molar eruption has been used to age *P. xanthopus* older than three years (Poole *et al.* 1985, Close and Bell 1990), but this approach was not used in the current study. Sample sizes and numbers of repeat

measurements were insufficient to determine accurate confidence intervals for either subspecies. Results were compared with those of Poole *et al.* (1985) to examine the validity of logarithms produced, and to determine if growth differed significantly by one-factor ANOVA between the sub-species and captive and wild animals.

7.2.2 Blood analysis

Trapping methods used to obtain blood samples post-release are detailed in Chapter 6. Haematological, biochemical and vitamin E analysis procedures are detailed in 3.3.1.4. Blood sample collection and analysis procedures were kept consistent between pre- and post-release to allow direct comparisons.

Post-release haematology, biochemistry and vitamin E data for *P. x. celeris* and *P. x. xanthopus* were analysed for dependence on age and mass by simple linear regression, and for differences between sex, release site (Lambert Station), subspecies, season, and pre- and post-release values by one-factor ANOVA (Zar 1984). As vitamin E is known to be plant derived (Rucker and Morris 1987), dependence of plasma vitamin E concentration on vegetation abundance and rainfall were investigated by simple linear regression.

7.2.3 Condition Scores

Note: The condition scoring of animals was initially undertaken as a comparative guide of animal health during trapping sessions for report to project proponents (Royal Zoological Society of South Australia and Queensland Environmental Protection Agency), rather than for quantitative analysis. Consequently, the scoring system was not introduced in Chapter 3, despite pre-release condition scores being included in Table 3.4. Its inclusion is the result of significant relationships being found between the technique and independent environmental factors.

All independent animals were scored for body condition and general health upon capture. Scores ranged from 5 for an animal in faultless condition, to 1 for an animal in extremely poor condition. Each animal was initially designated a score of 5, with points being deducted as follows:

½ point for minor loss in mass (<200g) or gain less than expected for the animal's age;

1 point for major loss in mass (>200g) or gain less than expected for the animal's age;

½ point for poor coat condition or alopecia;

½ point for heavy parasite load (light parasite loads are normal in captive and wild *P. xanthopus*);

½ point for excessive capture stress (e.g. profuse forearm and/or groin licking), as this is often an indicator of underlying physiological problems such as heat stress, dehydration or undernutrition;

½ point for subdued response to handling, lethargy, or a slow departure upon release;

½ point for evidence of diarrhoea; and

½ point for non-reproducing sexually mature females.

For example, a stressed male suffering minor mass loss, heavy parasite load, stress and general lethargy during drought would score 3 out of 5.

Dependence of condition scores on age, mass, rainfall, vegetation abundance, and vitamin E concentration were analysed by simple linear regression, and sex, release site (*P. x. celeris*), subspecies, season, and pre- and post-release changes (*P. x. celeris*) were examined by one-factor ANOVA (Zar 1984).

7.3 Results for *Petrogale xanthopus celeris* and *P. x. xanthopus*

7.3.1 Growth rates and age determination

Natural logarithms for the growth of all captive and post-release captured male and female *P. x. celeris*, and the relative accuracy of each parameter, as indicated by the percentage of measurements accounted for by R^2 , are reported (Table 7.1). For *P. x. celeris*, body mass was the best indicator of age, followed by forearm length (from bent wrist to elbow) and head length for males and the opposite for females. The difference is possibly explained by the more developed forearm and shoulder musculature of male macropods (Jarman 1989). Contrary to Poole *et al.* (1985), hindfoot and tail length were found to be less accurate predictors of age. Ear length was the least accurate age indicator for the sub-species.

For *P. x. xanthopus* captured post-release (no captive measurements were obtained), head length, tibia and forearm were the best predictors of age for males, and tibia, body mass

and forearm for females (Table 7.2). Body mass was a less accurate predictor of age for *P. x. xanthopus* (Fig. 7.2, 7.4) than *P. x. celeris* (Fig. 7.1, 7.3), probably due to the less constant environment at Aroona Sanctuary, where animals underwent fluctuations during regular dry periods. Ear length was the least accurate indicator of age for reintroduced *P. x. xanthopus*. Although a similar finding to reintroduced *P. x. celeris*, Poole *et al.* (1985) reported it to be a reasonable predictor of age in captive *P. x. xanthopus*.

Table 7.1 Logarithmic growth equations for aging captive- and wild-born *Petrogale xanthopus celeris* from mass and body measurements, where y is the body parameter, x is age in months, and R^2 is percentage of measurement variation accounted for by the relationship.

	<u>MALES</u>			<u>FEMALE</u>		
	<i>N</i>	Equation	R^2	<i>N</i>	Equation	R^2
MASS (g)	19	$y = 3213.8\text{Ln}(x) - 4301.8$	98.1	40	$y = 2466.5\text{Ln}(x) - 3063.5$	95.5
HEAD (mm)	19	$y = 30.952\text{Ln}(x) + 22.053$	94.4	40	$y = 27.102\text{Ln}(x) + 27.924$	92.8
EAR (mm)	19	$y = 12.2\text{Ln}(x) + 47.784$	63.2	40	$y = 18.544\text{Ln}(x) + 28.701$	72.1
FOREARM (mm)	19	$y = 30.55\text{Ln}(x) + 14.018$	96.0	40	$y = 25.207\text{Ln}(x) + 18.677$	89.1
TIBIA (mm)	19	$y = 53.565\text{Ln}(x) + 73.099$	84.6	40	$y = 61.896\text{Ln}(x) + 34.218$	87.0
FOOT (mm)	19	$y = 40.221\text{Ln}(x) + 42.599$	80.7	40	$y = 35.955\text{Ln}(x) + 46.42$	78.0
TAIL (mm)	19	$y = 165.8\text{Ln}(x) + 5.9181$	71.8	40	$y = 178.91\text{Ln}(x) - 17.622$	87.4

To calculate the age of male *P. x. celeris*: $\text{Age (mth)} = e^{x [(mass+4301.8)/3213.8]}$

Table 7.2 Logarithmic growth equations for aging re-introduced and wild-born *Petrogale xanthopus xanthopus* from mass and body measurements, where y is the body parameter, x is age in months, and R^2 is the measurement variation accounted for by the relationship.

	<u>MALES</u>			<u>FEMALE</u>		
	<i>N</i>	Equation	R^2	<i>N</i>	Equation	R^2
MASS (g)	27	$y = 3280.1\text{Ln}(x) - 5052.3$	91.7	30	$y = 2933.9\text{Ln}(x) - 4220.4$	92.0
HEAD (mm)	27	$y = 31.22\text{Ln}(x) + 20.531$	96.0	30	$y = 24.481\text{Ln}(x) + 38.856$	83.6
EAR (mm)	27	$y = 19.917\text{Ln}(x) + 24.045$	83.0	30	$y = 10.515\text{Ln}(x) + 50.284$	63.6
FOREARM (mm)	27	$y = 34.244\text{Ln}(x) - 1.231$	92.9	30	$y = 29.771\text{Ln}(x) + 9.212$	86.2
TIBIA (mm)	27	$y = 71.604\text{Ln}(x) + 14.371$	95.7	30	$y = 57.824\text{Ln}(x) + 53.116$	92.6
FOOT (mm)	27	$y = 49.062\text{Ln}(x) + 24.631$	92.2	30	$y = 23.367\text{Ln}(x) + 96.091$	71.4
TAIL (mm)	27	$y = 207.08\text{Ln}(x) - 36$	89.6	30	$y = 144.42\text{Ln}(x) + 136.17$	84.9

Male *P. x. celeris* (Fig. 7.1) gained mass faster than *P. x. xanthopus* (Fig. 7.2), as evident by the lower age at which each 1000 g mass was reached. Again this is likely a result of the more consistent environment at Lambert Station. Female *P. x. celeris* and *P. x. xanthopus* gained mass at similar rates until 12 months of age or 3 kg, after which gain in mass of *P. x. celeris* slowed. Therefore, growth rates were more similar between the sexes in *P. x. xanthopus* than in *P. x. celeris*, indicating greater sexual dimorphism in the latter (Sharman *et al.* 1995).

Growth equations for five head lengths of male and female reintroduced *P. x. celeris* and *P. x. xanthopus*, and captive *P. x. xanthopus* (Poole *et al.* 1985) were compared to examine differences in grow rate between the three groups. Head lengths, 80-120 mm in 10 mm increments, were chosen to represent the range of head measurements recorded for near-independent pouch young to late adulthood. Growth in head length is minimal above 120 mm for females and 130 mm for males (pers. obs.). Results indicated that captive *P. x. xanthopus* had the lower four head lengths at a younger age and thus the fastest rate of growth (Fig. 7.5). The second fastest rate of growth was in reintroduced *P. x. celeris*, followed by reintroduced *P. x. xanthopus*. These results follow the same pattern of food or vegetation abundance, which was highest in captivity and lowest at Aroona Sanctuary. Calculated ages were not significantly different between sub-species or between reintroduced and captive *P. x. xanthopus*; thus growth was relatively consistent between the two *P. xanthopus* subspecies. However, larger sample sizes for reintroduced *P. x. celeris* and *P. x. xanthopus* and from wild populations would be needed to confirm these trends.

Figure 7.1 Relationship between body mass and age for male *Petrogale xanthopus celeris*.

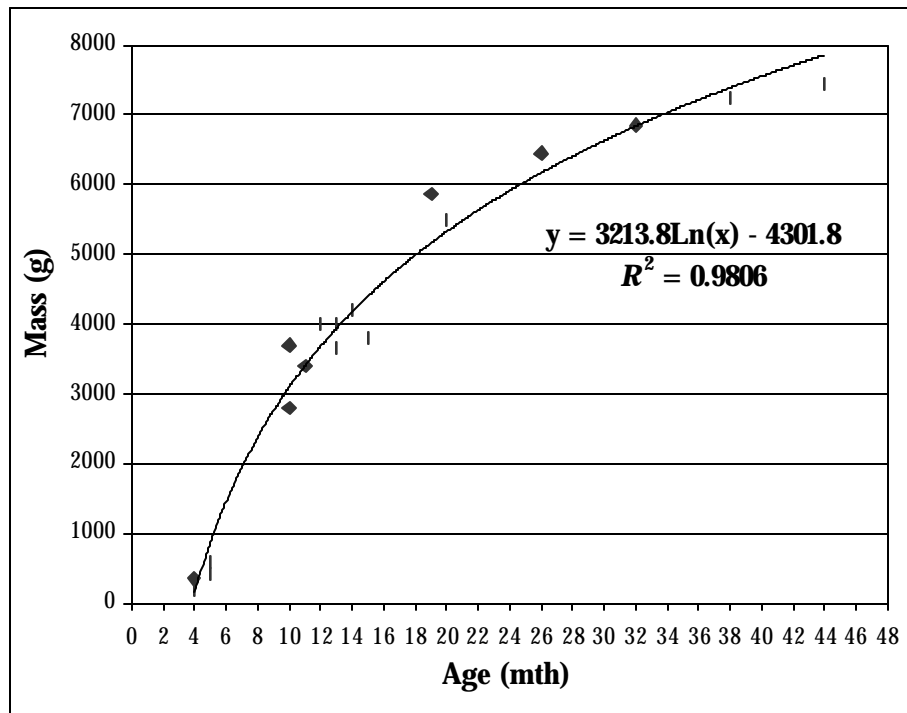


Figure 7.2 Relationship between body mass and age for male *Petrogale xanthopus xanthopus*.

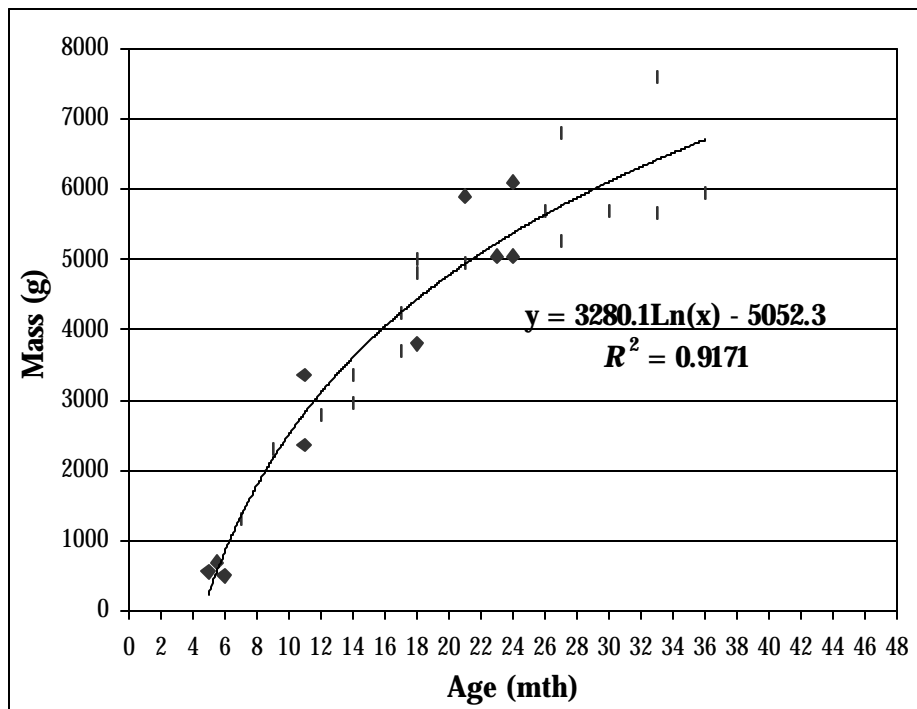


Figure 7.3 Relationship between body mass and age for female *Petrogale xanthopus celeris*

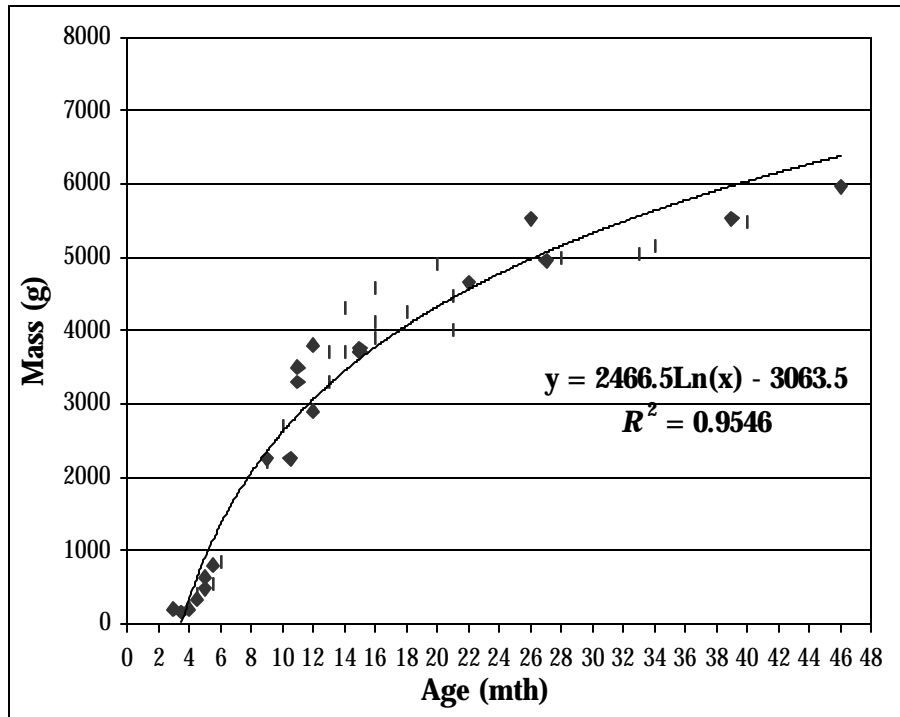


Figure 7.4 Relationship between body mass and age for female *Petrogale xanthopus xanthopus*.

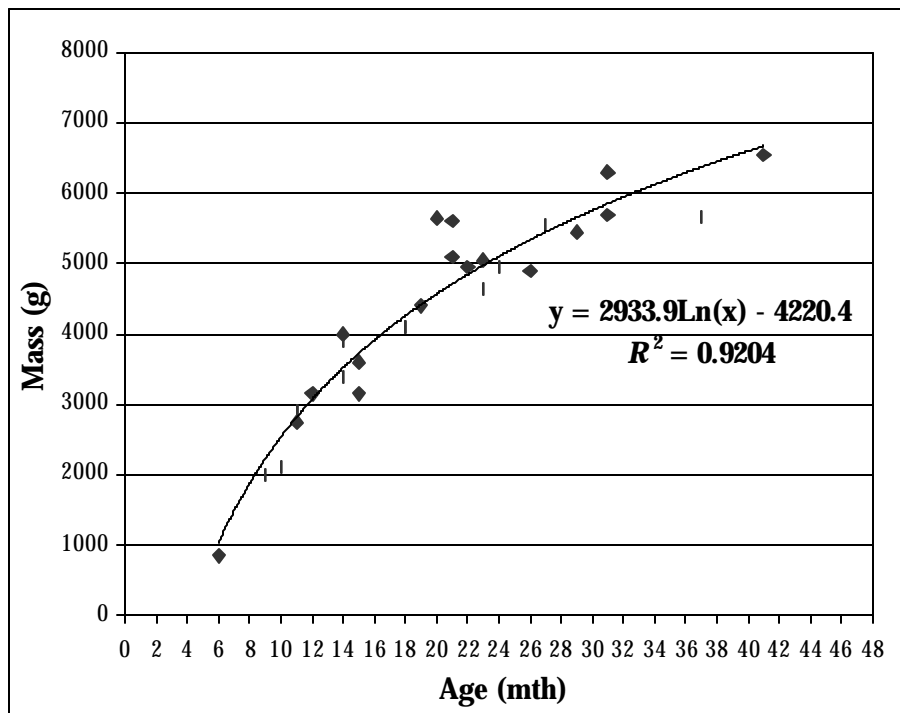
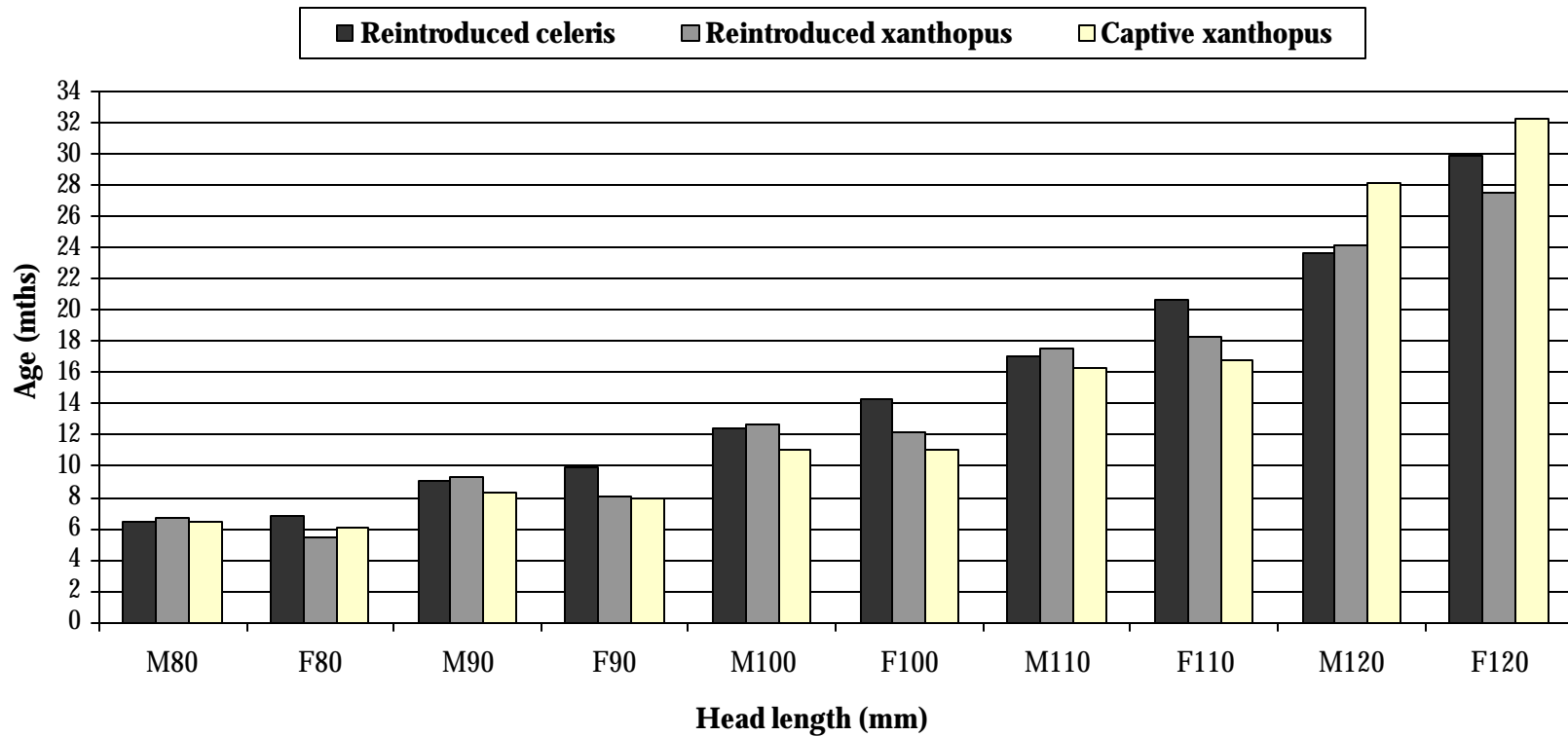


Figure 7.5 Comparison of ages obtained at five head lengths using logarithmic growth curves for re-introduced *Petrogale xanthopus celeris* and *P. x. xanthopus* (this study), and captive *P. x. xanthopus* (Poole *et al.* 1985). Head lengths range from 80 mm for pouch-young to 120 mm for adults. Key to x-axis labels: M- Male, F- Female. Captive *P. x. xanthopus* obtained the lower four head lengths (80-110 mm) at a younger age and thus had the fastest rate of growth. Ages were not however significantly different between sub-species or re-introduced and captive *P. x. xanthopus* (one-factor ANOVA, $P < 0.05$).



7.3.2 Haematology

Pre-release haematology for *P. x. celeris* (Table 3.2) and *P. x. xanthopus* (Table 3.6) are detailed in Chapter 3. Haematological analysis in this chapter refers to post-release data, except for the comparison of pre- and post-release values. Haematology results for *P. xanthopus* (Tables 7.3, 7.4) are largely similar to those of other marsupials (Bolliger and Backhouse 1960; Lewis *et al.* 1968; Parsons *et al.* 1970; Parsons *et al.* 1971a; Harrop and Barker 1972; Barnett *et al.* 1979a; Barnett *et al.* 1979b; Presidente and Correa 1981; Bradley 1990; Spencer and Speare 1992; Gaughwin *et al.* 1984). They are most similar to *P. assimilis* (Spencer and Speare 1992) and *M. eugenii* (Lewis *et al.* 1968) compared with other marsupials and the monotremes (Lewis *et al.* 1968; Parsons *et al.* 1971a; Barnett *et al.* 1979a).

Ponder *et al.* (1928) and Parsons *et al.* (1971a) reported considerable variation in white blood cell (WBC) counts for marsupials. *P. xanthopus* WBC counts are similar to those of *P. assimilis* (Spencer and Speare 1992), but higher than those of other macropods (Ponder *et al.* 1928). Spencer and Speare (1992) suggested that this was because *Petrogale* species lived in crowded colonies in the wild. Higher WBC counts post-release, where colonies were not crowded, do not support this idea. Higher WBC counts in *Petrogale* species may rather be an evolutionary adaptation to sedentary existence or isolation and may help prevent disease epidemics, which could potentially drive small isolated populations to local extinction. This alternative explanation is supported by high WBC counts for other sedentary marsupials with a restricted distributional range, such as Southern Hairy-nosed Wombats (*Lasiornhinus latifrons*), Quokkas (*Setonix brachyurus*) and Tasmanian devils (*Sarcophilus harrisi*) (Lewis *et al.* 1968; Parsons *et al.* 1971a). In contrast, species with wide distributional ranges have relatively low WBC counts (Parsons *et al.* 1971a).

Barnett *et al.* (1979a) and Spencer and Speare (1992) reported significant age and sex differences in haematological values for Common and Mountain Brushtail Possums (*Trichosurus vulpecula* and *T. caninus*) and *P. assimilis* respectively, with males and older animals generally having higher counts. Haematology results for *P. xanthopus* were not dependent on age or gender (Table 7.5). This difference may be due to Barnett *et al.* (1979a) pooling results into sub-adult and adult age classes, and Spencer and Speare (1992) showing the relationship for animals under 2.5 years of age; most *P. xanthopus* in

the current study were older than this. Haemoglobin (HGB: $F=9.89$, $F_{0.01(1)1,137}=6.82$, $P<0.01$), packed cell volume (PCV: $F=10.3$, $F_{0.01(1)1,138}=6.82$, $P<0.01$), and mean corpuscular volume (MCV: $F=9.21$, $F_{0.01(1)1,138}=6.82$, $P<0.01$) were dependent on body mass and platelet count (PLT: $F=7.01$, $F_{0.05(1)1,126}=6.85$, $P<0.01$) was inversely dependent on mass. As *P. xanthopus* mass is highly dependent on age (simple linear regression; $F=93.8$, $F_{0.0005(1)1,133}=12.7$, $P<0.0005$) this result is unexpected. A possible explanation is that age (months) was less statistically sensitive than mass (<100 g accuracy). Thus, although the same regression trends occurred for mass and age, the latter was not significant ($P<0.05$). Barnett *et al.* (1979a, b) and Spencer and Speare (1992) both aged animals in days rather than months.

HGB, PCV and MCV in marsupials have previously been reported to increase with body mass or age (Parsons *et al.* 1971a; Barnett *et al.* 1979a; Presidente and Correa 1981; Spencer and Speare 1992), and are thus consistent with the current findings. Increases are related to the greater oxygen requirements of larger animals. PLT counts remained stable in the only other report on this parameter in marsupials (Spencer and Speare 1992). In the current study PLT counts significantly decreased with increasing mass. The difference between the two studies is not readily explained.

Comparison of haematology results for *P. x. celeris* at the three Lambert Station release sites indicated only one significant difference; MCV was lower at Site 1 than Sites 2 and 3 ($F=5.54$, $F_{0.01(1)2,68}=4.92$, $P<0.01$). MCV is calculated from RBC (Feldman *et al.* 2000). Although RBC counts were not significantly different between sites, there was greater variation at Site 1 ($0.36 \times 10^{12} \text{ L}^{-1}$) than Sites 2 and 3 (0.13 and $0.28 \times 10^{12} \text{ L}^{-1}$ respectively), resulting in greater MCV variation and, by chance, statistical significance.

PCV, MCV and mean corpuscular haemoglobin (MCH) were all significantly greater in *P. x. celeris* than *P. x. xanthopus* ($F=8.24$, $F=233.3$ and $F=36.1$ respectively, $F_{0.005(1)1,138}=8.14$, $P<0.005$). PCV and MCV were also dependent on body mass ($F=10.3$ and $F=9.21$ respectively, $F_{0.005(1)1,138}=8.14$, $P<0.005$). Spencer and Speare (1992) have reported MCH is also related to mass. The mean mass of bled *P. x. celeris* was 5.9 kg (Table 7.3) and *P. x. xanthopus* was 5.1 kg (Table 7.4). Thus findings may be related to differences in mass rather than reflecting differences between the sub-species.

Table 7.3 Post-release haematology for re-introduced *Petrogale xanthopus celeris*. Key to abbreviations: WBC- white blood cells, RBC- red blood cells, HGB- haemoglobin, PCV- packed cell volume, MCV- mean corpuscular volume, MCH- mean corpuscular haemoglobin, MCHC- mean corpuscular haemoglobin concentration, and PLT- platelets. Table 3.2 contains pre-release values.

	COLONY	N	MASS	AGE	WBC	RBC	HGB	PCV	MCV	MCH	MCHC	PLT
			kg	Months	$\times 10^9 \text{ L}^{-1}$	$\times 10^{12} \text{ L}^{-1}$	g L^{-1}	LL^{-1}	fL	pg	g L^{-1}	$\times 10^9 \text{ L}^{-1}$
Trapping 1	1	3	5.8±0.8	32±2	9.8±2.0	5.3±0.3	161±10	0.488±0.03	92.0±1.2	30.5±0.1	332±3	123±65
January 1999	2	3	5.5±0.8	28±10	10.5±1.5	4.97±0.3	153±7	0.460±0.02	92.7±2.7	30.8±0.8	332±1	140±49
	3	5	5.5±0.8	28±4	7.5±0.9	4.74±0.6	145±18	0.435±0.05	91.1±3.1	30.9±0.3	340±14	92±38
	Mean	11	5.6±0.8	29±6	9.3±2.0	5.0±0.4	153±12	0.461±0.03	91.9±2.3	30.7±0.4	335±10	118±60
Trapping 2	1	6	6.3±1.0	29±11	6.4±1.7	4.97±1.07	165±11	0.448±0.096	90.4±3.9	34.8±9.1	384±93	95±47
July 1999	2	6	5.0±1.6	26±14	7.2±1.3	5.18±0.4	166±12	0.494±0.043	95.2±3.4	32.0±0.6	337±9	117±33
	3	4	5.7±0.7	34±5	6.8±2.2	5.26±0.42T	168±10	0.490±0.032	93.2±1.9	31.9±0.6	342±4	101±58
	Mean	16	5.7±1.2	29±11	6.8±1.6	5.12±0.69	166±10	0.475±0.066	92.9±3.8	33.0±5.5	356±59	105±43
Trapping 3	1	4	6.5±1.2	39±7	8.6±1.7	4.85±0.63	156±15	0.456±0.052	94.1±2.1	32.3±1.2	344±7	111±61
January 2000	2	7	5.3±1.3	35±16	7.2±1.8	4.80±0.5	153±16	0.449±0.049	93.7±5.1	31.9±1.1	341±8	41±29
	3	5	5.3±1.3	36±14	7.5±0.4	5.25±0.50	163±7	0.495±0.022	94.6±5.4	31.3±1.7	330±5	136±50
	Mean	16	5.7±1.3	36±12	7.6±1.5	4.95±0.54	157±13	0.465±0.045	94.1±4.4	31.8±1.3	338±8	88±61
Trapping 4	1	7	6.5±1.9	36±16	7.5±1.7	5.15±0.35	157±10	0.447±0.034	86.9±3.1	30.5±0.9	351±8	85±48
July 2000	2	7	5.3±1.1	29±12	7.0±2.7	5.07±0.23	162±7	0.471±0.023	92.9±2.7	32.0±0.8	345±6	78±49
	3	4	5.7±1.1	40±14	6.7±1.7	5.36±0.46	142±53	0.489±0.025	91.5±3.7	26.6±9.9	290±106	126±72
	Mean	18	5.9±1.4	35±14	7.1±2.0	5.17±0.34	156±25	0.466±0.031	90.2±4.0	30.2±4.7	335±51	91±54
Trapping 5	1	6	7.6±1.5	42±14	6.3±1.6	5.00±0.35	151±10	0.468±0.041	93.7±3.9	30.2±1.7	323±19	123±64
January 2001	2	5	6.2±0.8	40±11	5.2±2.2	4.99±0.2	160±12	0.495±0.031	99.1±2.8	32.1±1.6	324±13	130±53
	3	1	5.8	58	4.2	4.52	145	0.462	102.2	32.1	314	49
	Mean	12	6.9±1.3	42±13	5.7±1.8	4.95±0.29	154±11	0.479±0.035	96.7±4.3	31.2±1.7	323±15	120±56
MEAN		73	5.9±1.3	34±12	7.2±2.0	5.05±0.50	158±17	0.469±0.046	93.0±4.4	31.4±3.7	338±39	102±55

Table 7.4 Post-release haematology for re-introduced *Petrogale xanthopus xanthopus*. Key to abbreviations: WBC- white blood cells, RBC- red blood cells, HGB- haemoglobin, PCV- packed cell volume, MCV- mean corpuscular volume, MCH- mean corpuscular haemoglobin, MCHC- mean corpuscular haemoglobin concentration, and PLT- platelets. Table 3.6 contains pre-release values.

Trapping	Date	n	MASS kg	AGE Months	WBC x 10 ⁹ L ⁻¹	RBC x 10 ¹² L ⁻¹	HGB g L ⁻¹	PCV LL ⁻¹	MCV fL	MCH pg	MCHC g L ⁻¹	PLT x 10 ⁹ L ⁻¹
1	April 1998	7	4.5±1.7	36±12	5.5±2.1	4.66±0.32	131±20	0.401±0.029	85.9±2.0	28.2±3.3	328±35	-
2	July 1998	5	3.4±0.9	14±4	9.3±3.4	5.11±0.39	143±12	0.411±0.038	80.2±1.5	28±0.7	348±10	135±56
3	October 1998	4	5.2±0.9	36±17	7.4±1.9	5.24±0.93	165±8.6	0.447±0.078	85.5±1.8	28.8±0.9	337±2.9	111±47
4	January 1998	4	5.3±1.4	38±24	6.4±2.7	5.63±0.26	161±10	0.501±0.036	89±3	29.3±1.3	329±18	138±13
5	April 1999	8	5.2±1.0	34±20	8.1±1.7	5.40±0.54	155±14	0.443±0.049	81.9±1.9	28.7±0.9	351±11	106±25
6	July 1999	9	5.1±0.8	31±16	8.2±3.1	5.73±0.44	161±11	0.465±0.036	81.1±2.1	28.1±1.0	346±7	123±50
7	October 1999	6	4.8±1.6	34±24	7.7±3.0	5.00±0.72	142±23	0.410±0.063	81.9±1.6	28.4±1.2	347±10	116±53
8	January 2000	6	6.1±0.4	41±20	8.9±1.6	5.63±0.29	163±5	0.465±0.021	82.5±2.1	28.9±0.7	350±10	110±35
9	April 2000	7	5.4±1.1	38±23	9.5±3.5	5.48±0.27	157±9	0.462±0.032	84.3±3.0	28.6±0.5	339±14	112±37
10	July 2000	7	5.6±1.1	38±25	6.6±1.0	5.74±0.31	163±9	0.477±0.031	83.0±1.7	28.4±0.8	342±9	122±30
11	October 2000	2	6.4±1.0	59±31	7.5±0.4	5.43	156±6	0.450	82.9±0.3	28.7±1.2	346±13	119±85
12	January 2001	5	6.0±2.2	48±30	5.8±1.5	5.20±0.37	154±10	0.440±0.030	85.0±3.0	29.7±1.4	350±5	-
Mean		70	5.1±1.4	35±22	7.7±2.7	5.4±0.6	154±16	0.446±0.055	83.3±3.0	28.5±1.4	343±16	118±41

Table 7.5 Significant haematological changes in *Petrogale xanthopus* associated with physical and environmental variables and reintroduction. All statistics, except captive versus reintroduced, are based on post-release values of both sub-species. Key to abbreviations: WBC- white blood cells, RBC- red blood cells, HGB- haemoglobin, PCV- packed cell volume, MCV- mean corpuscular volume, MCH- mean corpuscular haemoglobin, MCHC- mean corpuscular haemoglobin concentration, and PLT- platelets. - indicates no significant effect, ρ indicates a significantly higher mean and * indicates no baseline data.

Variable	Sub-species	Stat. test	d.f.	WBC	RBC	HGB	PCV	MCV	MCH	MCHC	PLT
Age	pooled	Regression	-	-	-	-	-	-	-	-	-
Mass	pooled	Regression	1,57-138	-	-	Positive $P<0.005$	Positive $P<0.005$	Positive $P<0.005$	-	-	Negative $P<0.05$
Sex	pooled	ANOVA	-	-	-	-	-	-	-	-	-
Lambert Sites	<i>celeris</i>	ANOVA	2,68	-	-	-	-	ρ Site 2-3 $P<0.01$	-	-	-
Sub-species	compared	ANOVA	1,126-138	-	ρ <i>xanthopus</i> $P<0.001$	-	ρ <i>celeris</i> $P<0.005$	ρ <i>celeris</i> $P<0.005$	ρ <i>celeris</i> $P<0.005$	-	-
Season- summer v winter	<i>celeris</i>	ANOVA	1,69	-	-	-	-	ρ summer $P<0.005$	-	-	-
- all	<i>xanthopus</i>	ANOVA	3,65	-	ρ summer and winter $P<0.05$	-	-	ρ summer $P<0.01$	-	-	-
Captive v reintroduced	<i>celeris</i>	ANOVA	1,135	-	ρ captivity $P<0.05$	ρ captivity $P<0.05$	ρ captivity $P<0.05$	-	-	-	ρ captivity $P<0.001$
	<i>xanthopus</i>	trend	-	ρ wild	ρ captivity	ρ captivity	ρ captivity	ρ captivity	-	-	*

Table 7.6 Post-release biochemistry for re-introduced *Petrogale xanthopus celeris*. Pre-release values shown in Table 3.3.

COLONY		n	MASS	AGE	Protein	Albumin	Cholesterol	Triglycerides	Creatinine	Urea	Globulins	Vitamin E
			kg	Months	g L ⁻¹	g L ⁻¹	mmol L ⁻¹	mmol L ⁻¹	μmol L ⁻¹	mmol L ⁻¹	g L ⁻¹	μmol L ⁻¹
Trapping 1	1	3	5.8±0.8	32±2	76.1±6.5	37.9±2.5	-	0.79±0.39	114±12	10.1±0.9	38.2±9.0	20±5
	January 1999	2	5.5±0.8	28±10	70.7±2.0	39.3±1.3	-	0.51±0.12	107±14	9.2±1.0	32.2±3.2	23±3
		3	5.5±0.8	28±4	74.0±7.6	39.1±3.2	-	0.43±0.12	115±18	11.4±0.7	34.9±6.8	20±6
	Mean	11	5.6±0.8	29±6	73.7±6.5	38.8±2.7	-	0.55±0.27	113±16	10.5±1.3	35.1±7.1	21±5
Trapping 2	1	6	6.3±1.0	29±11	63.1±4.4	43.6±1.3	3.23±0.85	0.33±0.20	111±10	9.17±1.05	19.6±3.3	23±4
	July 1999	2	5.0±1.6	26±14	62.4±7.3	41.8±3.3	3.78±0.84	0.60±0.35	99±14	8.43±1.66	19.6±5.5	26±8
		3	5.7±0.7	34±5	57.3±12.5	44.1±1.7	3.46±0.59	0.28±0.06	116±12	9.71±1.58	15.8±7.2	27±2
	Mean	16	5.7±1.2	29±11	61.4±7.8	43.0±2.4	3.49±0.78	0.42±0.28	108±13	9.02±1.44	18.6±5.2	25±6
Trapping 3	1	4	6.5±1.2	39±7	74.5±12.8	48.5±7.7	3.00±1.49	0.49±0.17	141±8	10.88±1.56	26.0±7.5	19±5
	January 2000	2	5.3±1.3	35±16	82.8±17.0	50.7±15.1	5.15±2.01	1.01±0.45	148±14	12.78±2.10	32.1±5.8	28±4
		3	5.3±1.3	36±14	81.2±9.8	50.1±4.5	3.50±2.12	0.39±0.18	138±15	11.58±1.07	31.1±5.4	23±4
	Mean	16	5.7±1.3	36±12	80.2±13.7	50.0±10.5	4.10±2.05	0.69±0.43	143±13	11.93±1.80	30.3±6.3	24±5
Trapping 4	1	7	6.5±1.9	36±16	88.1±19.4	51.9±9.8	4.13±1.26	0.42±0.27	98.4±12.3	10.7±1.4	36.2±11.3	26±2
	July 2000	2	5.3±1.1	29±12	97.4±14.0	55.0±7.8	5.72±2.22	0.65±0.34	105±8	10.9±2.3	42.4±8.3	34±8
		3	5.7±1.1	40±14	134.0±11.0	76.9±8.8	8.28±1.60	0.60±0.19	113.3±8.0	11.6±2.2	57.2±2.9	44±12
	Mean	18	5.9±1.4	35±14	101.9±23.5	58.6±13.1	5.67±2.31	0.55±0.29	104±11	10.99±1.87	43.3±11.7	33±10
Trapping 5	1	6	7.6±1.5	42±14	70.3±6.2	41.8±1.9	2.38±0.77	0.54±0.36	118±10	9.75±0.88	28.5±5.0	27±4
	January 2001	2	6.2±0.8	40±11	66.9±5.8	42.3±2.4	3.18±0.80	0.54±0.25	106±9	10.59±1.45	25.4±5.2	32±6
		3	5.8	58	55.4	37.0	2.00	0.76	120	11.72	18.4	19
	Mean	12	6.9±1.3	42±13	67.6±6.6	41.6±2.3	2.68±0.80	0.55±0.28	113±10	10.27±1.18	26.4±5.2	29±6
MEAN		73	5.9±1.4	34±12	78.4±20.5	47.5±11.0	4.12±2.00	0.55±0.33	116±19	10.57±1.85	31.0±11.7	27±8

Table 7.7 Post-release biochemistry for reintroduced *Petrogale xanthopus xanthopus*. Pre-release values shown in Table 3.6.

Trapping	Month	n	MASS kg	AGE Months	Protein g L ⁻¹	Albumin g L ⁻¹	Cholesterol mmol L ⁻¹	Triglycerides mmol L ⁻¹	Creatinine μmol L ⁻¹	Urea mmol L ⁻¹	Globulins g L ⁻¹	Vitamin E μmol L ⁻¹
2	July 1998	5	3.4±0.9	14±4	54.9±3.8	36.3±2.9	3.42±0.48	0.43±0.05	84±8	10.4±0.6	18.6±2.6	22±7
3	October 1998	4	5.2±0.9	36±17	57.3±3.0	36.3±5.5	3.22±0.34	0.22±0.06	70±7	12.53±2.1	21.1±5.0	21±7
4	January 1998	4	5.3±1.4	38±24	84.2±14.6	57.1±6.8	-	0.76±0.26	136±28	16.32±3.32	27.0±12.4	18±3
5	April 1999	8	5.2±1.0	34±20	64.4±5.3	42.1±2.7	2.50±0.59	0.41±0.11	93±10	12.58±1.33	22.3±5.9	23±5
6	July 1999	9	5.1±0.8	31±16	62.4±4.7	40.8±3.3	3.08±0.74	0.29±0.13	96±7	12.35±2.34	21.6±3.6	19±6
7	October 1999	6	4.8±1.6	34±24	68.3±3.6	39.1±17.1	2.64±1.18	0.49±0.15	94±19	11.49±2.43	22.8±1.9	21±5
8	January 2000	6	6.1±0.4	41±20	61.5±8.6	47.6±2.6	3.10±0.97	0.40±0.36	187±25	11.63±1.89	14.2±8.0	19±4
9	April 2000	7	5.4±1.1	38±23	67.4±5.7	46.1±2.2	3.18±0.83	0.27±0.17	126±20	10.11±1.93	21.3±5.5	22±3
10	July 2000	7	5.6±1.1	38±25	73.3±10.2	42.9±2.8	3.14±0.37	0.48±0.18	103±16	10.57±2.00	30.4±8.6	20±7
11	October 2000	2	6.4±1.0	59±31	56.9±6.5	36.8±4.6	3.18±0.52	0.58±0.11	79±4	10.87±1.80	20.1±2.0	31±9
12	January 2001	5	6.0±2.2	48±30	58.1±3.4	36.1±3.3	2.32±0.36	0.91±0.30	103±7	12.42±2.04	22.1±1.3	24±3
Mean		63	5.3±1.2	36±22	64.3±8.5	42.3±8.0	2.97±0.75	0.44±0.25	108±34	11.84±2.45	22.1±7.2	21±6

Table 7.8 Significant biochemical changes in *Petrogale xanthopus* associated with physical and environmental variables and re-introduction. All statistics, except captive versus reintroduced, are based on post-release values of both sub-species. Key: - indicates no significant effect, ρ indicates a significantly higher mean and * indicates no baseline data.

Variable	Sub-species	Stat. test	d.f.	Protein	Albumin	Cholesterol	Triglycerides	Creatinine	Urea	Globulins	Vitamin E
Age	pooled	Regression	-	-	-	-	-	-	-	-	Positive P<0.02
Mass	pooled	Regression	1,133	-	-	-	-	Positive P<0.005	-	-	-
Sex	pooled	ANOVA	1,133	-	-	-	ρ females P<0.05	-	-	-	-
Lambert Sites	<i>celeris</i>	ANOVA	2,59-70	-	-	ρ Site 2-3 P<0.05	ρ Site 2 P<0.025	-	-	-	ρ Site 2-3 P<0.05
Sub-species	compared	ANOVA	1,117-133	ρ <i>celeris</i> P<0.001	ρ <i>celeris</i> P<0.025	ρ <i>celeris</i> P<0.001	ρ <i>celeris</i> P<0.05	-	ρ <i>xanthopus</i> P<0.001	ρ <i>celeris</i> P<0.001	ρ <i>celeris</i> P<0.001
Season- summer v winter	<i>celeris</i>	ANOVA	1,62-72	-	ρ winter P<0.01	ρ summer P<0.025	-	ρ summer P<0.001	ρ summer P<0.05	-	ρ winter P<0.05
- all	<i>xanthopus</i>	ANOVA	3,58	-	ρ summer P<0.02	-	ρ summer P<0.005	ρ summer P<0.001	ρ summer P<0.02	-	-
Captive vs reintroduced	<i>celeris</i>	ANOVA	1,80-140	ρ captivity P<0.005	ρ captivity P<0.005	*	ρ captivity P<0.001	ρ captivity P<0.001	-	ρ captivity P<0.025	ρ wild P<0.0005
	<i>xanthopus</i>	trend	-	ρ captivity	ρ wild	ρ captivity	*	ρ captivity	-	ρ wild	ρ wild

Red blood cell (RBC) counts were higher in summer and winter than autumn and spring for *P. x. xanthopus* ($F=2.82$, $F_{0.05(1)3,65}=2.75$, $P<0.05$), with an associated higher MCV in summer ($F=4.52$, $F_{0.01(1)3,65}=4.10$, $P<0.01$). The higher RBC coincided with greater temperature extremes. Sealander (1962) proposed that species that do not avoid temperature extremes by hibernating must be able to extend their tolerance limits through thermal acclimation. Thus, higher RBC counts in *P. xanthopus* in summer and winter are likely related to greater thermoregulatory demands. Similarly, mean RBC counts were higher for *P. x. xanthopus* than *P. x. celeris* ($F=12.2$, $F_{0.001(1)1,138}=5.73$, $P<0.001$), probably due to greater and more prolonged temperature extremes at Aroona Sanctuary than Lambert Station (Fig. 2.2, 2.5) and less shade. Furthermore, Barnett *et al.* (1979b) reported higher RBC counts for *T. caninus* occupying peripheral or marginal habitat than those occupying preferred habitat, thus supporting the notion that Aroona Sanctuary is more marginal habitat than Lambert Station.

Significantly greater RBC, HGB, PCV and PLT counts ($F=4.86$, $F=7.26$, $F=4.96$ and $F=13.8$ respectively, $F_{0.05(1)1,135}=3.91$, $P<0.05$) in captive than re-introduced *P. xanthopus* indicate a higher quality and more consistent diet in captivity (Gaughwin *et al.* 1984). The same difference occurred between captive and wild *S. brachyurus* (Shield 1971). Differences are possibly due to less consistency in the diet of re-introduced or wild animals, including periods of semi-starvation through drought, that cause greater fluctuations in haematological values (Shield 1971; Gaughwin *et al.* 1984). This is evident in lower standard deviations of haematological parameters for captive animals (Tables 3.2, 7.2).

7.3.3 Biochemistry

Pre-release biochemical parameters for *P. x. celeris* are shown in Table 3.3 and for *P. x. xanthopus* in Table 3.6. Biochemical analysis refers to post-release data only, except for the comparison of pre- and post-release values. As vitamin E underwent the most significant change post-release it is presented separately (7.3.4; Fig. 7.5).

Studies on broad-spectrum plasma biochemistry of marsupials are rare (Parsons *et al.* 1971b; Gaughwin *et al.* 1984), although analysis of protein or albumin is more common (Lewis *et al.* 1968; Barnett *et al.* 1979a; Barnett *et al.* 1979b; Presidente and Correa 1981;

Bradley 1990). Plasma biochemistry results for *P. xanthopus* (Table 7.6, 7.7) generally showed similar values to other marsupials except urea and cholesterol. *P. xanthopus* has the highest recorded urea and cholesterol concentrations for a macropod (Parsons *et al.* 1971b; Presidente and Correa 1981), and is more similar to *L. latifrons* (Caughwin *et al.* 1984) which occupies a similar climatic environment (Wells 1978). High cholesterol has also been reported for the marsupial families Dasyuridae and Peramelidae (Parsons *et al.* 1971b). Urea and cholesterol concentrations in both marsupials (10-15 mmol L⁻¹ and 3-4 mmol L⁻¹ respectively) are considerably greater than in eutherian mammals of similar (5-30 kg; 2-5 mmol L⁻¹ and 1-2 mmol L⁻¹) or larger (>30 kg; 312 mmol L⁻¹ and 1.5-2.5 mmol L⁻¹) body mass (Kaneko *et al.* 1997).

Presidente and Correa (1981) reported that protein and globulin concentrations in *T. vulpecula* increased linearly with age, but no such relationship was found for *P. xanthopus* (Table 7.8). The difference is likely a result of the maximum age class in Presidente and Correa (1981) of 24+ months, while the majority of animals bled in the current study were adults with a mean age greater than 27 months. Moreover, Kaneko (1997) reported that plasma proteins including albumin and globulins become more stable in adulthood. Plasma proteins (total protein, albumin, globulins) remained relatively stable post-release, indicating no obvious disease (Kaneko 1997). Protein (total protein: $F=25.1$, $F_{0.001(1)1,132}=11.3$, $P<0.001$; albumin: $F=9.84$, $F_{0.025(1)1,133}=9.50$, $P<0.0025$; globulins: $F=27.3$, $F_{0.001(1)1,133}=11.3$, $P<0.001$) and triglyceride ($F=4.62$, $F_{0.05(1)1,133}=3.91$, $P<0.05$) concentrations were higher in *P. x. celeris* than *P. x. xanthopus* post-release, and higher in captivity (total protein: $F=8.25$, $F_{0.005(1)1,140}=8.14$, $P<0.005$; albumin: $F=8.93$, $F_{0.005(1)1,140}=8.14$, $P<0.005$; globulins: $F=5.45$, $F_{0.025(1)1,140}=5.13$, $P<0.025$; triglyceride $F=30.4$, $F_{0.001(1)1,140}=11.3$, $P<0.001$). This result is directly attributable to diet quality and consistency (Kaneko 1997). Commercial pellet diets are usually high in protein content (I.D. Hume pers. comm.), which would explain the higher levels seen in captivity. Post-release plasma protein concentrations were higher in *P. x. celeris* than *P. x. xanthopus*, reflecting the greater abundance and more consistent vegetation, predominantly browse, at Lambert Station than Aroona Sanctuary (Ch. 5). Albumin concentration increased significantly ($F=8.21$, $F_{0.01(1)1,71}=7.01$, $P<0.01$) in *P. x. celeris* in winter, indicative of some degree of dehydration (Kaneko 1997). The reverse trend was observed in *P. x. xanthopus* with significantly higher albumin concentrations in summer ($F=3.98$, $F_{0.02(1)3,58}=3.54$, $P<0.02$), possibly caused by heat stress and dehydration,

which concentrates proteins in the plasma; of the plasma proteins albumin is the most prominent and thus the most likely to be affected (Kaneko 1997).

Levels of plasma cholesterol are related to diet, but only nonesterified cholesterol is absorbed. The type of triglyceride present in the diet determines cholesterol absorption (Kaneko *et al.* 1997). Cholesterol and triglycerides were significantly higher in *P. x. celeris* than *P. x. xanthopus* (cholesterol: $F=17.6$, $F_{0.001(1)1,117}=11.4$, $P<0.001$; triglycerides: $F=4.62$, $F_{0.05(1)1,133}=3.91$, $P<0.05$), indicating a diet richer in cholesterol and triglycerides in *P. x. celeris*. Similarly, cholesterol was significantly higher in summer than winter in *P. x. celeris* ($F=5.49$, $F_{0.025(1)1,60}=5.29$, $P<0.025$), which coincides with higher rainfall and greater plant growth. The significantly higher levels of cholesterol ($F=3.88$, $F_{0.05(1)2,59}=3.15$, $P<0.05$), triglycerides ($F=4.86$, $F_{0.025(1)2,70}=3.89$, $P<0.025$) and vitamin E ($F=3.19$, $F_{0.05(1)2,68}=3.13$, $P<0.05$) in *P. x. celeris* at Sites 2 and 3 indicates that Bendee (*Acacia catenulata*) may be rich in both, as the vegetation was otherwise similar between sites. The significantly higher levels of triglycerides recorded in females than males ($F=4.73$, $F_{0.05(1)1,133}=3.91$, $P<0.05$) have been linked with lactation (Schweigert 1993).

Although plasma creatinine concentration is generally consistent (approximately 100 $\mu\text{mol L}^{-1}$) between marsupials (Parsons *et al.* 1971b), it was the only biochemical parameter dependent on mass in *P. xanthopus* ($F=8.26$, $F_{0.005(1)1,133}=8.15$, $P<0.005$). Although not recorded for marsupials, Finco (1997) found creatinine to be affected by dietary intake and muscle mass. Creatinine is affected by metabolic rate (Hume 1999), with higher levels recorded in active or physical stressed animals (Finco 1997). Thus the higher creatinine concentrations found heavier animals may be related to greater physical activity or exertion or higher metabolic rate, or to the personal observation that older animals stress more during capture.

Creatinine levels increased in *L. latifrons* during drought (Gaughwin *et al.* 1984). The drought occurred in late spring and summer, and thus higher temperatures (not reported) may have confounded results. Creatinine concentration was significantly higher for both *P. x. celeris* ($F=24.0$, $F_{0.001(1)1,71}=11.8$, $P<0.001$) and *P. x. xanthopus* ($F=17.0$, $F_{0.001(1)3,58}=6.17$, $P<0.001$) during the hotter summer sampling. Results indicate that heat stress, whether natural or capture-induced, may elevate creatinine concentration.

Furthermore, creatinine concentrations of *P. x. xanthopus* remained uniformly low (<100 $\mu\text{mol L}^{-1}$) in July 1998, despite the lowest condition scores being recorded for all animals (Fig. 7.8) and the death of at least one animal due to starvation (Fig. 6.22).

Creatinine concentration was significantly higher in captivity ($F=22.3$, $F_{0.001(1)1,140}=11.3$, $P<0.001$). *P. x. celeris* handled for the first time in captivity at Charleville had the highest creatinine levels recorded for the sub-species (152 ± 30 $\mu\text{mol L}^{-1}$) throughout the three years of sampling. Creatinine levels decreased by the second capture two months later (134 ± 16 $\mu\text{mol L}^{-1}$), and continued to decrease by the third capture (122 ± 32 $\mu\text{mol L}^{-1}$). As capture procedures and climatic conditions (April to August 1998) remained relatively constant throughout baseline data collection, results indicate that the elevated creatinine levels were directly caused by capture and handling stress, for which animals gradually became accustomed. Thus a highly significant decrease post-release (113 ± 16 $\mu\text{mol L}^{-1}$ after 5 months; 108 ± 13 $\mu\text{mol L}^{-1}$ after 12 months) indicates that *P. x. celeris* showed more evidence of stress in captivity. As the recapture process did not overtly stress *P. x. celeris*, the other haematological and biochemical values obtained were likely to be valid. The same result occurred for *P. x. xanthopus*, creatinine concentrations were 122 ± 22 $\mu\text{mol L}^{-1}$ in captive animals, but averaged 108 ± 34 $\mu\text{mol L}^{-1}$ post-release.

Plasma urea concentration was significantly higher in *P. x. xanthopus* than *P. x. celeris* ($F=11.8$, $F_{0.001(1)1,133}=11.3$, $P<0.001$), despite the availability of free water to the former. Urea concentration in *P. x. celeris* was significantly higher in summer than winter ($F=5.00$, $F_{0.05(1)1,71}=3.98$, $P<0.05$). Although urea concentration was not significantly higher in summer for *P. x. xanthopus* in a four-season ANOVA, when mean summer urea concentration is compared to combined results for other seasons there is a significance difference (summer 13.2 mmol L^{-1} V. autumn to spring 11.3 - 11.7 mmol L^{-1} ; $F=5.98$, $F_{0.02(1)1,60}=5.71$, $P<0.02$). Gaughwin *et al.* (1984) reported that urea concentration increased as a result of poor nutrition in *L. latifrons* but this cannot be used to explain the trend for higher plasma urea concentrations in *P. x. xanthopus* in summer, as rainfall and vegetation abundance were lower in winter at both Lambert Station and Aroona Sanctuary (Chapter 5). Furthermore, urea concentrations in *P. x. xanthopus* (10.4 ± 0.6 $\mu\text{mol L}^{-1}$) were below average (11.84 ± 2.45 $\mu\text{mol L}^{-1}$) in July 1998, and coincided with minima in condition scores (Fig. 7.7) and vegetation abundance (Fig. 5.9), and the death

of at least one animal from starvation (Fig. 6.22). Dawson and Denny (1969) reported that arid zone kangaroos (*Macropus rufus* and *M. robustus*) concentrate urine in summer, likely as a result of thermoregulatory demands of evaporative cooling. The current findings concur, although elevated urea levels may also be biased by dehydration of the animals in the traps despite the provision of free water.

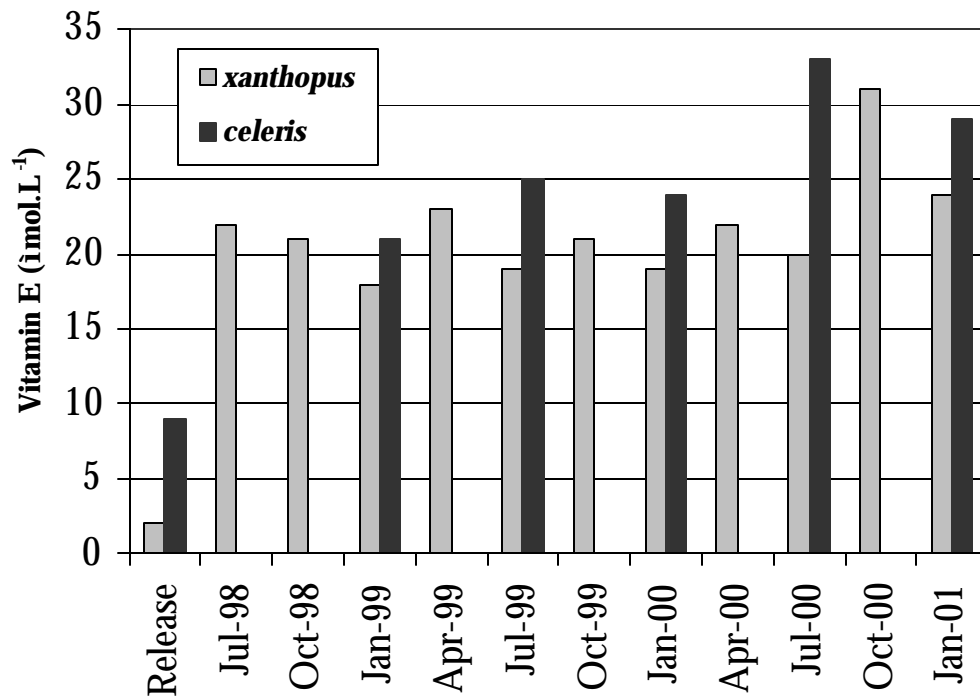
7.3.4 Vitamin E

Plasma vitamin E concentration (PVEC) underwent the most significant ($F=61.3$, $F_{0.0005(1)1,80}=13.2$, $P<0.0005$) post-release change of all blood biochemistry parameters assessed. PVEC in *P. x. celeris* more than doubled by five months post-release, and continued to increase over the following two years (Fig. 7.6). Similarly in *P. x. xanthopus*, PVEC was 10 times higher at Aroona Sanctuary 22 months post-release (2.1 to 22 $\mu\text{mol L}^{-1}$) than in captivity (Fig. 7.6). Green leaves are important sources of vitamin E (Rucker and Morris 1997). PVEC was dependent on age ($F=5.56$, $F_{0.02(1)1,130}=5.54$, $P<0.02$), indicating either that it accumulates with age or that older animals consume a diet higher in leaf content and thus vitamin E. Significantly higher PVEC was recorded for *P. x. celeris* at Sites 2 and 3 than Site 1 at Lambert Station ($F=3.19$, $F_{0.05(1)2,68}=3.13$, $P<0.05$). The higher occurrence of *A. catenulate* at Sites 2 and 3 suggests that it may be high in vitamin E, as vegetation was otherwise similar between sites, but plant levels of the vitamin were not measured. PVEC was also significantly higher in *P. x. celeris* than *P. x. xanthopus* ($F=19.9$, $F_{0.001(1)1,130}=11.3$, $P<0.001$), consistent with the higher availability of browse at Lambert Station than Aroona Sanctuary (Fig. 5.6, 5.9). Significantly higher PVEC in winter in *P. x. celeris* ($F=4.55$, $F_{0.05(1)1,69}=4.00$, $P<0.05$) but not *P. x. xanthopus* is likely related to dehydration in *P. x. celeris* concentrating the vitamin in the plasma rather than a change in dietary intake, as vitamin E availability would be expected to remain relatively constant throughout the year along with browse availability (Fig. 5.6).

PVEC was found to be independent of rainfall (*celeris*: $F=0.03$, $F_{0.05(1)1,3}=10.1$, $P>0.05$; *xanthopus*: $F=1.24$, $F_{0.05(1)1,9}=5.12$, $P>0.05$) or vegetation abundance (*celeris*: $F=0.26$, $F_{0.05(1)1,3}=10.1$, $P>0.05$; *xanthopus*: $F=1.57$, $F_{0.05(1)1,9}=5.12$, $P>0.05$) in both subspecies. Browse species, predominantly large woody bushes and trees, were the only plant group not to undergo marked seasonal fluctuations at either fieldsite (Fig. 5.6, 5.9), indicating

they were the most likely source of vitamin E, or were at least relied upon during drier periods when most other vegetation had disappeared.

Figure 7.6 Mean vitamin E concentration of re-introduced *Petrogale xanthopus celeris* at Lambert Station and *P. x. xanthopus* at Aroona Sanctuary.



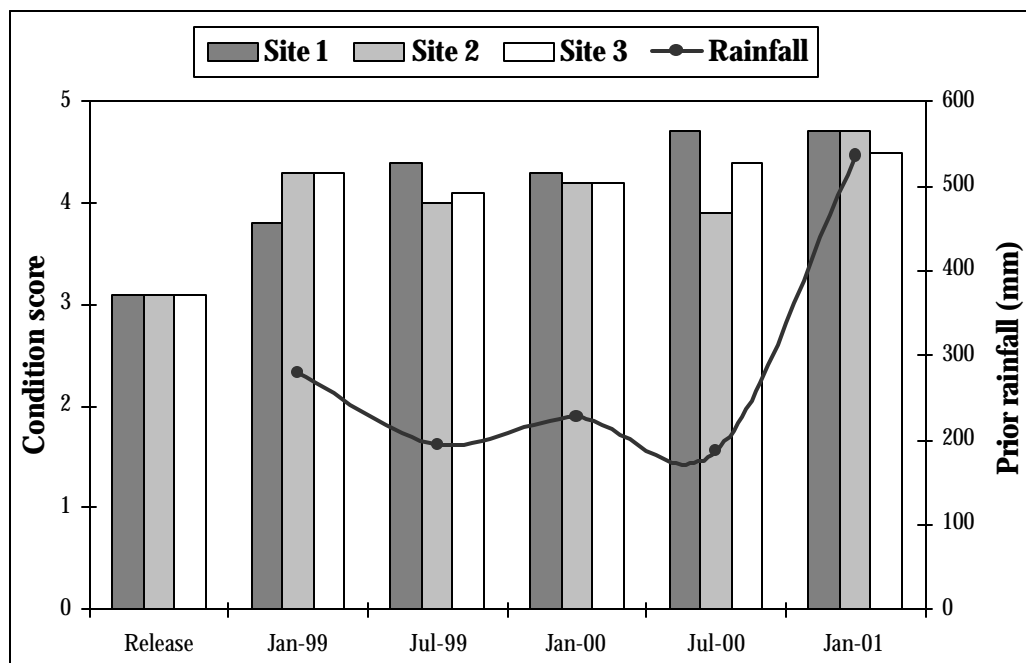
7.3.5 Condition Scores

Condition scores generally remained stable throughout the year, although there was a significant increase post-release for *P. x. celeris* ($F=82.6$, $F_{0.001(1)1,93}=11.55$, $P<0.001$). Condition scores for both sub-species were found to be independent of age and mass ($F=2.35$ and $F=1.87$ respectively, $F_{(1)1,131}=3.91$, $P>0.05$), sex ($F=0.80$, $F_{(1)1,131}=3.91$, $P>0.05$) and season (*celeris*: $F=0.27$, $F_{0.05(1)1,69}=4.00$, $P>0.05$; *xanthopus* $F=2.22$, $F_{0.05(1)3,58}=2.76$, $P>0.05$). Condition scores of *P. x. celeris* were found to be independent of prior rainfall ($F=0.03$, $F_{0.05(1)1,3}=10.1$, $P>0.05$), vegetation abundance ($F=0.03$, $F_{0.05(1)1,3}=10.1$, $P>0.05$), and release site ($F=3.06$, $F_{0.05(1)2,68}=3.13$, $P>0.05$) (Fig. 7.7). However, condition scores of *P. x. xanthopus* were dependent on rainfall between quarterly trapping sessions ($F=7.0$, $F_{0.05(1)1,9}=5.12$, $P<0.05$), and vegetation abundance ($F=15.2$, $F_{0.005(1)1,9}=13.6$, $P<0.005$) at Aroona Sanctuary (Figure 7.8). Furthermore,

condition scores of *P. x. celeris* (4.3 ± 0.5) were significantly higher than those of *P. x. xanthopus* (4.0 ± 0.6) throughout post-release monitoring ($F=9.71$, $F_{0.05(1)1,9}=8.16$, $P<0.005$).

Condition scores were significantly dependent on independently measured plasma vitamin E concentration for all *P. x. celeris* values ($F=4.59$, $F_{0.05(1)1,69}=3.98$, $P<0.05$) and mean trapping *P. x. xanthopus* values ($F=5.37$, $F_{0.05(1)1,9}=5.12$, $P<0.05$) when tested using simple linear regression.

Figure 7.7 Post-release condition scores for re-introduced *Petrogale xanthopus celeris* at Lambert Station. Condition scores did not significantly differ between sites ($F=3.06$, $F_{0.05(1)2,68}=3.13$, $P>0.05$) and were independent of prior rainfall ($F=0.03$, $F_{0.05(1)1,3}=10.1$, $P>0.05$).



Ecto-parasites were recorded on 29% (7/24) of *P. x. celeris* during final veterinary preparation before release in August 1998 (3.3.2.3). In post-release summers 62% (8/13), 53% (10/19) and 62% (8/13) of animals carried ecto-parasites in January 1999, 2000 and 2001 respectively. The number of animals carrying ecto-parasites fell to 16% (3/19) and 26% (5/19) in the winters of July 1999 and 2000 respectively. Thus, while more reintroduced *P. x. celeris* carried ecto-parasites post-release in summer, fewer did in

winter, the season in which the pre-release survey was conducted. No summer ecto-parasite survey was undertaken in captivity for comparison. Consequently, seasonal baseline data on ecto-parasites on *P. x. celeris* are insufficient to determine if ecto-parasite burdens changed significantly post-release.

As well as carrying heavy ecto-parasite loads in summer, two of three wild male *P. x. celeris* (E1M and E4M) had Macropod Pox Virus lesions on the inner thigh (Fig. 7.9). Despite 18 months co-existence with reintroduced *P. x. celeris* (E1M), no released animal showed evidence of the disease at the cessation of trapping. Macropod Pox Virus lesions were only detected in wild *P. x. celeris* males during summer, possibly related to heat stress (Speare *et al.* 1989).

Table 7.8 Post-release condition scores for re-introduced *Petrogale xanthopus* at Aroona Sanctuary. Condition scores of *P. x. xanthopus* were significantly dependent rainfall between quarterly trapping sessions ($F=7.0$, $F_{0.05(1),9}=5.12$, $P<0.05$) and vegetation abundance ($F=15.2$, $F_{0.005(1),9}=13.6$, $P<0.005$).

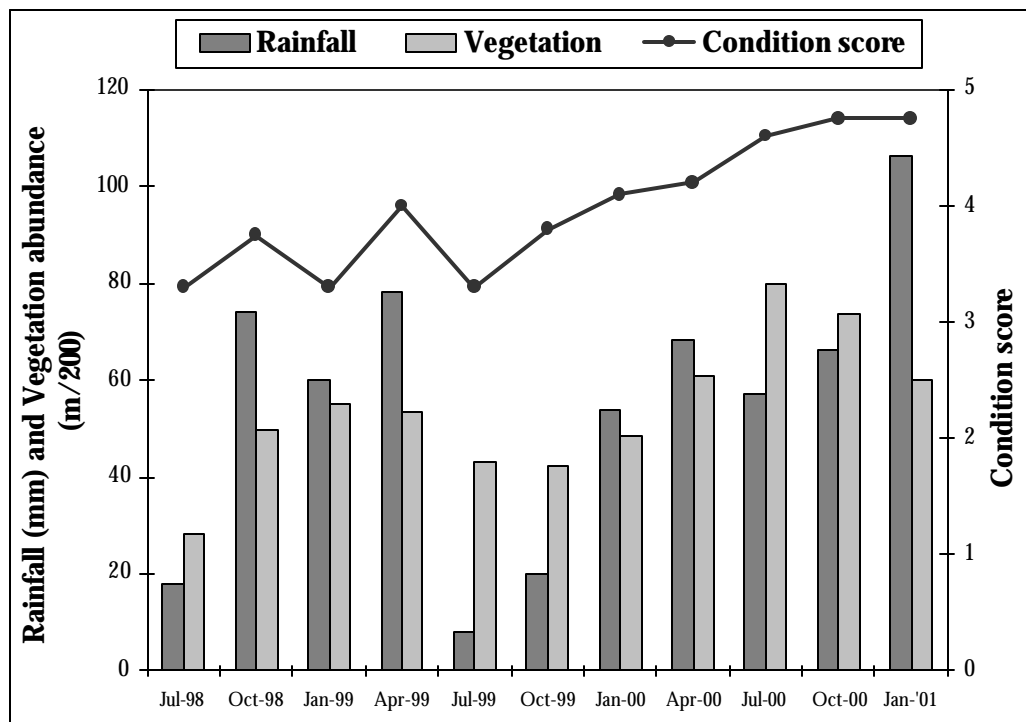


Figure 7.9 Macropod Pox Virus in the groin of wild *Petrogale xanthopus celeris* on Lambert Station, Queensland. The virus was only detected during summer, possibly related to heat stress. Virus transmission to released animals did not occur in 18 months after initial detection.



7.4 Discussion

Although sample size was small, relatively high R^2 values for growth of both re-introduced *P. x. celeris* and *P. x. xanthopus* indicate that growth rates within sexes of both sub-species was relatively consistent. The animal sample size (8 : 18) used by Poole *et al.* (1985) was similar to that of the current study, but Poole *et al.* (1985)'s study was based on 8-11 repeat measures per animal, which probably accounts for the considerably higher R^2 values. Nevertheless, results from the current study were similar to those of Poole *et al.* (1985). Mass was the best indicator of age in *P. x. celeris*, while head length in male *P. x. xanthopus* and tibia length in female *P. x. xanthopus* were the best indicators, with tibia being best overall for the sub-species. Tibia length was also the equal best indicator, along with head length, in captive female *P. x. xanthopus* (Poole *et al.* 1985). The difference between sub-species is possibly due to the more constant growing conditions at Lambert Station with its associated greater abundance of vegetation.

Comparison of growth rates of re-introduced *P. x. celeris* and *P. x. xanthopus* and captive *P. x. xanthopus* (Poole *et al.* 1985) showed no significant difference, although growth of captive *P. x. xanthopus* was slightly faster. The current study agrees with findings of Shield

and Woolley (1961), Sadleir (1963), Murphy and Smith (1970) and Delaney and De'ath (1990) that growth rates of captive and wild macropods are not significantly different. Moreover, results show that growth was not retarded by the re-introduction process and sudden change in diet. However, *P. x. xanthopus* at Aroona Sanctuary ceased breeding during dry times thus preventing possible retarded development of wild-born pouch young.

The capture and bleeding process of animals has been reported to both affect haematological and biochemical results (Parsons *et al.* 1971b) and to have no significant effect (Barnett *et al.* 1979a; Speare and Spencer 1992). The difference is possibly explained by Parsons *et al.*'s (1971b) low sample size (n=1) and physically stressing the animal through "rough handling and partial suffocation". Although Parsons *et al.* (1971b) recommended the collection of blood samples under anaesthesia, plasma glucose levels in *T. vulpecula* were elevated in anaesthetised animals, indicating a stress response (Barnett *et al.* 1979a). The low creatinine levels in the current study indicate that animals were accustomed to handling.

The vast majority of haematological and biochemical parameters observed underwent a significant decrease post-release. Haematology of re-introduced *P. xanthopus* was more consistent with wild *P. assimilis* post-release than pre-release (Spencer and Speare 1992), indicating that blood parameters are artificially high in captive *P. xanthopus*. Blood chemistry values are predominantly related to nutritional status (Kaneko *et al.* 1997; Feldman *et al.* 2000). Blood parameters changed following release to the wild, likely as the result of the adoption of an appropriate natural diet (Lapidge 2000).

Results from the current study indicate that thermoregulation, osmoregulation and stress play major roles in haematological and plasma biochemical values of *P. xanthopus* in the wild. Significant increases in the red blood cell counts during summer and winter reflect their role in thermoregulation. Changes in albumin, cholesterol, creatinine and urea concentrations were more closely related to heat stress and dehydration during hot (summer) or dry (winter) periods in *P. xanthopus* than to previously reported nutritional stress (Caughwin *et al.* 1984). Thus, findings indicate that extrapolation of plasma biochemical results obtained on captive animals to wild animals should be made with caution.

Plasma vitamin E concentration and its association with myopathy is reported to be affected by enclosure size in *S. brachyurus*, with animals housed in smaller enclosures more prone to myopathy (Kakulas 1963). Results from the current study suggest that other factors such as nutrition were involved; *P. x. xanthopus* housed in large natural enclosures at Monarto Zoological Park had a significantly lower plasma vitamin E concentration ($2.1 \text{ } \mu\text{mol L}^{-1}$) than *P. x. celeris* housed in small crowded enclosures at the Charleville Environmental Protection Agency compound ($7.8 \text{ } \mu\text{mol L}^{-1}$). Furthermore, a number of *P. x. xanthopus* at Monarto Zoological Park apparently died from myopathy (S. Conaghty pers. comm.), while there are no reported cases for *P. x. celeris*. The difference is possibly because the diet of captive *P. x. celeris* was supplemented with Mulga (*Acacia aneura*) leaves. This feature of captive *P. x. celeris* animal husbandry may be one that has prevented animal deaths from myopathy in the overcrowded enclosures. The technique is recommended for other captive herbivores that are prone to myopathy. Findings suggest that plasma vitamin E concentrations should be higher than $2.1 \text{ } \mu\text{mol L}^{-1}$ to prevent myopathy in *P. xanthopus*.

Vitamin E (PVEC) underwent the most significant change post-release of all measured biochemical parameters, doubling in five months in *P. x. celeris* and increasing ten-fold in *P. x. xanthopus* in 22 months. The control of PVEC is reported to be multi-factorial, with nutrition and stress implicated (Kakulas 1961, 1963b; Munday 1988; Speare *et al.* 1989; Rucker and Morris 1997; Hume 1999). Results from the current study indicate that browse, in particular *Acacia* species, is the most likely dietary source of the vitamin for free-living *P. x. celeris*. The diet of reintroduced *P. x. xanthopus* contained minimal browse (4%) at Monarto Zoological Park prior to release (Lapidge 2000) and low PVEC ($2.1 \text{ } \mu\text{mol L}^{-1}$) (Conaghty and Schultz 1998). One month post-release browse intake had risen to 12% of the diet, and remained similarly high at the end of the study (Lapidge 2000). A reduction in stress levels may also be involved, a suggestion supported by the highly significant decrease in plasma creatinine concentration in *P. xanthopus* post-release. Thus, results support the idea that PVEC is both diet and stress related.

Tocopherol (Vitamin E) comes from the Greek word “tokos” meaning reproduction or childbirth, as it is known to aid embryonic development (Rucker and Morris 1997). An effect of vitamin E on macropod reproduction has not been reported. Higher fecundity of *P. x. celeris* was recorded post-release (100%) than pre-release (86%), with associated

higher vitamin E levels of 21 $\mu\text{mol L}^{-1}$ and 8 $\mu\text{mol L}^{-1}$ respectively. However, Conaghty and Schultz (1998) reported a PVEC in wild macropods, likely *M. r. erubescens*, of 9 $\mu\text{mol L}^{-1}$ which was similar to that of captive *P. x. celeris*. Unfortunately fecundity rates of captive *P. x. xanthopus* with lower PVEC are not known. These limited data indicate that vitamin E may assist reproduction in *P. x. celeris*.

The condition score index developed and used throughout the study provides a simple but indicative parameter with which to compare the condition of animals across the population and over time. While less direct than body measurements previously used to assess condition in macropods (Bakker and Main 1980; Catt 1981; Moss and Croft 1999; Short and Turner 2000), the technique takes into account more factors that indicate animal health. Condition scores were significantly dependent on independently measured vitamin E levels for both sub-species, and vegetation abundance and rainfall for *P. x. xanthopus*. These findings attest to its worth. The difference between sub-species is related to the more constant environment present at Lambert Station. Furthermore, the technique was found particularly useful in conveying information on animal health throughout the study to project supporters, particularly the Royal Zoological Society of South Australia.

Condition scores for *P. x. celeris* increased significantly post-release. Animals generally gained mass faster than in captivity, males suffered less alopecia, females were more fecund, both sexes showed less evidence of stress, and coat condition was observed to improve. Although Short and Turner (2000) found that reintroduced Burrowing Bettongs (*Bettongia lesueur*) maintained similar body condition to the source population post-release, both populations were free living.

Despite great seasonal variation in rainfall and vegetation abundance at Lambert Station, there was no difference in condition scores of *P. x. celeris* between summer and winter, indicating that captive-bred animals were highly resilient to seasonal change. Physiological implications from the current study suggest that captive-bred *P. xanthopus* successfully adapt to an arid environment upon release, showing similar growth and blood parameters to other wild *Petrogale* species. Moreover, significantly lower stress indices and significantly higher overall condition indicated that the health of both captive-bred *P. xanthopus* sub-species improved upon release into the wild.

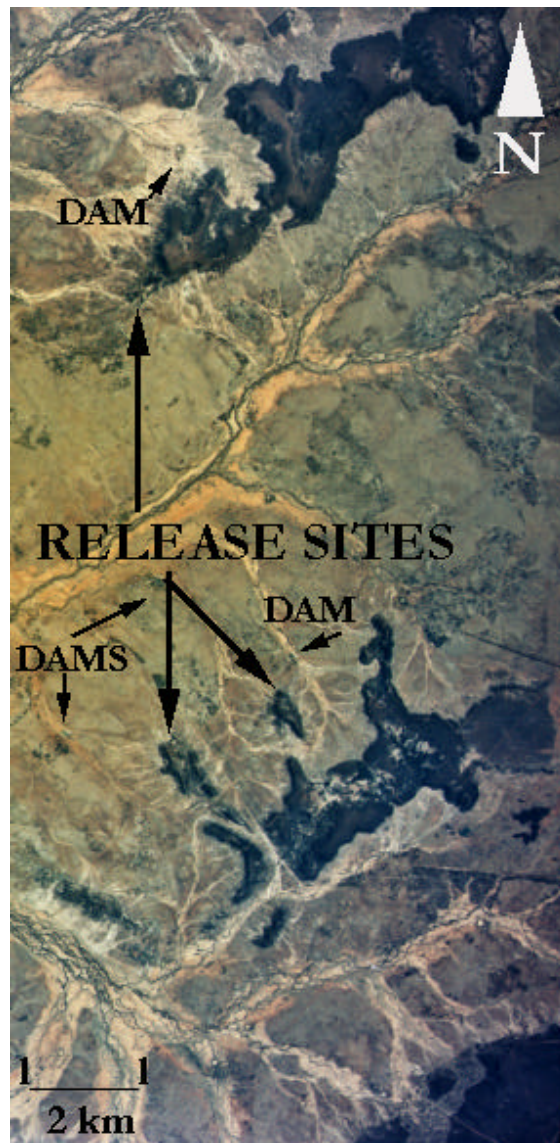
SEASONAL WATER TURNOVER, FIELD METABOLIC RATE, AND FOOD INTAKE RATE

8.1 Introduction

Adaptation of re-introduced captive-bred animals to the wild involves many factors, the foremost of which is obtaining sufficient water and energy to survive. The ability of captive-bred animals to adapt their nutritional requirements to a new environment and cope with seasonal fluctuations is little studied (Stanley Price 1989; Lapidge 2000).

Water influx in animals includes preformed water in the diet, metabolic water produced through oxidation of foodstuffs, and free water consumed through drinking and from water vapor in the air (Withers 1992). The reliance of *P. xanthopus* on free drinking water is a matter of contention. Lim (1987) reported that *P. x. xanthopus* requires fresh water to survive periods of low rainfall. Observation and radio-telemetry of free-living wallabies during drought showed that most travelled up to 2 km to drink at a permanent waterhole in summer (Lim 1987). A similar finding was reported for *P. x. xanthopus* at Coturaundee Range, New South Wales, where animals were sighted 5 km from their daytime refuge drinking at Pines Tank (Lim 1987). Allen (2001) recorded diatoms in the faeces of *P. x. celeris* at Lisburne Station after four months without rain, indicating that they traveled to a dam 600 m from their daytime refuge. Consequently, re-introduction sites on Lambert Station were chosen with proximity to a permanent source of free water in mind. From the point of release, Site 1 was 3 km from the closest dam (although only 0.6 km from the cliff edge further along the range), and Sites 2 and 3 were 0.5 km and 1.4 km respectively from the closest dam (Fig. 8.1). Although evidence suggests that *P. xanthopus* requires water during drought, many extant colonies do not occur near permanent free water, particularly in Queensland (P. McRae pers. comm.), indicating that the subspecies *P. x. celeris* must be able to survive without drinking.

Figure 8.1 Aerial photograph of *Petrogale xanthopus celeris* release sites on Lambert Station, indicating location of closest permanent free water.



To allow comparison between species of different body size, the allometric scaling exponents of 0.71 for water turnover rates (Nagy and Peterson 1988), 0.58 for field metabolic rate (Nagy 1994) and 0.75 for food intake rate (Withers 1992) have been used for field measurements throughout the chapter. An exponent of 0.80 has been used for captive water turnover rate (Hume 1999). Exponents represent the common slope of the regression equation for marsupials.

Captive *P. x. celeris* supplied with water *ad libitum* consumed 98 mL kg^{0.80} d⁻¹, although the relatively high protein content of the diet (22%) may have possibly increased water consumption above levels in the wild (Murphy 1985). The kidneys of *P. x. celeris* have a similar relative medullary thickness to those of other arid-dwelling macropods (Murphy 1985; Hume 1999; Blaney *et al.* 2000), indicating considerable ability to reabsorb filtered urea and withstand water stress (Hume 1999). Additionally, water-stressed *P. x. celeris* (50% reduction in water availability) were found to increase the digestibility of dietary energy, assisting animals to survive periods of lower nutritional value of vegetation during drought. However, a 50% reduction in water availability reduced dry matter intake by 23% and animals lost body mass (Murphy 1985). Eastern grey kangaroos (*Macropus giganteus*) similarly lost mass when experimentally deprived of free water (Blaney *et al.* 2000). The effects of water stress on *P. xanthopus* without access to free water are unknown.

Water turnover rates (WTR) of wild *P. x. xanthopus* have previously been measured at Middle Gorge, South Australia (Lim *et al.* 1987). Results showed two distinct patterns, a 'summer pattern', when a WTR of 101 mL kg^{-0.75} d⁻¹ was recorded, and a 'winter pattern' of 276 mL kg^{-0.75} d⁻¹. The higher WTR in winter was attributed to higher water content of the diet at the time, but this pattern was not repeated in the following winter, which was dry, and WTR reverted to the summer pattern. Lim *et al.* (1987) suggested that once the diet fails to provide enough preformed water, such as in the summer pattern, then *P. x. xanthopus* is reliant on free water. The summer WTR of *P. x. xanthopus* was similar to that of the wild sympatric Euros, *Macropus robustus erubescens* (104 mL kg^{-0.75} d⁻¹) and Red Kangaroos, *M. rufus* (97 mL kg^{-0.75} d⁻¹) (Dawson *et al.* 1975), and to that of the Unadorned Rock-wallaby, *P. inornata* (110 mL kg^{-0.75} d⁻¹) from the more mesic central Queensland coast (Green 1989). Rothschild's Rock-wallaby, *P. rothschildi*, which inhabits the wet/dry tropics of Western Australia at a similar latitude to *P. x. celeris* (Strahan 1995), was found to be able to withstand greater water stress than other *Petrogale* species examined. WTR at the end of the dry season (spring) was 63 mL kg^{-0.75} d⁻¹, increasing to 178 mL kg^{-0.75} d⁻¹ after rain (Green 1989). The lowest WTR recorded for a macropod is 51 mL kg^{-0.75} d⁻¹ for the Spectacled Harewallaby, *Lagorchestes conspicillatus*, at the end of the dry season in northern Australia (Bakker and Bradshaw 1983; Hume 1999).

Field metabolic rate (FMR) is the total energy cost of free existence. In addition to basal metabolism it includes the energy costs of foraging, growth, reproduction and thermoregulation (Nagy 1987). FMR is less well studied in macropods of similar size to *P. xanthopus* than WTR, possibly due to the high cost of the doubly-labelled water technique. A single measurement of field metabolic rate in *P. xanthopus*, likely at Middle Gorge, South Australia, was $622 \text{ kJ kg}^{-0.75} \text{ d}^{-1}$ (Green 1989).

The amount of dry matter consumed by an animal to meet its daily energy requirements can be estimated from its FMR (Nagy *et al.* 1999). The voluntary dry matter intake of captive *P. x. celeris* was reported to be $42 \text{ g kg}^{-0.75} \text{ d}^{-1}$ (Murphy 1985), similar to that calculated for *P. x. xanthopus* from field metabolic rate of $42.9 \text{ g kg}^{-0.75} \text{ d}^{-1}$, assuming a metabolisable energy content of an average forage diet of 10 kJ per g dry matter (Nagy *et al.* 1990a, b; Nagy *et al.* 1999). Similar dry matter intakes were reported for the Red-necked Pademelon, *T. thetis* ($40.9 \text{ g kg}^{-0.75} \text{ d}^{-1}$), Parma Wallaby, *M. parma* ($47.7 \text{ g kg}^{-0.75} \text{ d}^{-1}$), and Tammar Wallaby, *M. eugenii* ($37.2 \text{ g kg}^{-0.75} \text{ d}^{-1}$) in captivity by White, Hume and Nolan (1998).

This chapter investigates the seasonal nutritional requirements of captive-bred *P. x. celeris* re-introduced to Lambert Station through measurement of water turnover rates, field metabolic rates, and food intake rates.

8.2 Materials and Methods

8.2.1 Sampling technique

The doubly-labelled water method (Lifson and McClintock 1966) was used to determine water turnover rate and field metabolic rate following Nagy (1980, 1983, 1989). The technique is based on equilibration of ^{18}O with oxygen in the body's water and bicarbonate pools, the latter through the carbonic anhydrase reaction. Hydrogen is lost from the body in water, so the rate of loss of ^3H represents the WTR. Oxygen is lost from the body in both water and CO_2 so the difference between the rates of loss of ^{18}O and ^3H represents the rate of loss of CO_2 , or metabolic rate (Wallis and Green 1992). Potential errors in the doubly-labelled water technique have been reviewed (Nagy 1980, 1992; Nagy and Costa 1980; Green 1997; Gibson 1999).

P. x. celeris were sampled during the wet summer (January) and dry winter (July) of 2000. Trapping and bleeding procedures are outlined in Chapters 6 and 3 respectively. Traps were baited without free water. Animals were transferred from traps into large pet-packs and carried to the vehicle for transport to the field-base. Animals were placed in a darkened air-conditioned room prior to processing, during equilibration, and before being returned to the colony to minimize capture stress and dehydration. An initial 2mL blood sample was taken for measurement of background isotope levels. Blood was stored in 5-mL dried Lithium Heparin vials, further sealed with Parafilm, and frozen. Each animal was injected intramuscularly (hind-limb) with separate pre-weighed doses of H_2^{18}O and ^3HOH (tritiated water) in a 2.5 mL syringe (with 25 g needle attached). Summer dose rates were $0.5 \text{ mL } ^3\text{HOH kg}^{-1} \text{ body mass}$ (8 MBq.mL^{-1}) and $0.5 \text{ mL H}_2^{18}\text{O.kg}^{-1} \text{ body mass}$ (98 atom%). Winter dose rates were reduced to $0.1 \text{ mL } ^3\text{HOH kg}^{-1} \text{ body mass}$ (8 MBq.mL^{-1}) and $0.3 \text{ mL H}_2^{18}\text{O.kg}^{-1} \text{ body mass}$ (98 atom%) based on the summer results. The mass of the injectate was determined by weighing the injection syringe on a digital balance (0.001 g) before and after injection. Injectate mass was converted to volume by weighing 100 μL of each solution on a digital balance (0.0001 g). Animals were left for 3 to 4 h to allow the isotopes to equilibrate with the body water pool, sufficient time for a marsupial of the mass of adult *P. x. celeris* (Nagy 1983; Green 1989). A second 2-mL blood sample was taken following equilibration. Once all equilibration samples were obtained, animals were returned to the point of capture and released. Animals were recaptured between 5 and 9 days post-injection, reweighed (winter only), and a 2-mL blood sample was taken at the point of capture.

8.2.2 Sample analyses

Pure water was extracted from blood samples by microdistillation (Nagy 1983). Liquid-scintillation spectroscopy was used to determine ^3H activity. One hundred μL of distilled water was pipetted into plastic scintillation vials (Packard), to which 5 mL of scintillation fluid (Packard Ultima Gold™) was added. Specific activity of each sample, along with two vials of the standard dose and four blanks (scintillation fluid only), was determined using a Packard 1600 TR liquid scintillation counter. Samples from each collection period were analysed together to avoid bias in differential radioactive decay of tritium. Isotope Ratio Mass Spectroscopy (Speakman *et al.* 1990) was used to determine ^{18}O concentration by Metabolic Solutions Inc., Nashua, NH, U.S.A. WTR and FMR were calculated using

the equations of Nagy (1983, 1987). It was assumed that total body water did not change throughout the decay period and that any changes in total body water were linear. CO₂ production rates were converted to units of energy metabolism (kJ) using the calculated heat equivalent for a general herbivorous diet of 21.2 kJ.L⁻¹ CO₂ (Munks and Green 1995; Hume 1999).

8.2.3 Food intake rate

The rate of food intake required to balance daily energy expenditure was calculated from FMR. Nagy *et al.* (1990 a, b) and Munks (1990) calculated that the metabolisable energy content of a generalist herbivore diet was approximately 45% of gross energy. Assuming a gross energy content of 21 kJ.g⁻¹ dry matter, the metabolisable energy content of the diet consumed by wild *P. x. celeris* was approximately 10kJ.g⁻¹ dry matter. Lapidge (2000) reported that re-introduced *P. x. xanthopus* adopted a similar diet to their wild counterparts 2-3 months post-release. Allen's (2001) dietary study on wild *P. x. celeris* at nearby Lisburne Station included both dietary analysis and plant water content sampling during climatic conditions similar to those experienced on Lambert Station during FMR/WTR measurements. This allowed a mean plant water content to be used to estimate rates of dry and fresh plant matter intake and preformed water influx from the diet. Calculations are detailed as a footnote to the table of results. Mass-corrected dry matter intake rates and water influx rates through drinking were calculated to allow direct comparison between sexes and seasons.

8.2.4 Statistical analyses

Results for WTR and FMR from individual animals are included because of the unequal sample sizes in the two measurement periods. The significance of differences in WTR and FMR between sexes and seasons were tested using two-factor ANOVA for unequal sample sizes. Dependence of WTR and FMR on mass or reproductive status (joey age) was examined using simple linear regression (Zar 1984). Mean mass-corrected dry matter intake rates and water influx rates through drinking were compared between sexes and seasons by two-factor ANOVA for unequal replication (Zar 1984; SYSTAT 1998).

8.3 Results

8.3.1 Seasonal conditions

In the three months prior to the summer measurements 205 ml of rain had fallen on Lambert Station. During sampling a further 22 ml of rain fell. Daily temperatures during sampling were regularly in excess of 40°C. Radio-telemetry (Ch. 6) of animals immediately after rain indicated that they were foraging on the plain, and possibly were obtaining free water that had collected in the hoof depressions of cattle. No animal was tracked in the vicinity of dams. No rain fell during the winter measurements or during the two months prior. The last rainfall recorded was 117 mm in mid-May. Daily temperatures during measurements were generally in the mid-20°C range.

8.3.2 Animal details and body mass

Ten *P. x. celeris* were injected with doubly-labelled water ($H_2^{18}O$ and 3HOH) within the first 6 days of each measurement period, although one male (M9) was only injected with tritiated water (the cheaper isotope) in summer as the chance of recapture was deemed low. Eight animals were recaptured during both measurement periods. Four animals were sampled during both periods (M28, M24, C1M1 and F17), thus 12 animals were sampled in total (4 : 8). All females captured were lactating, with a mean age of pouch young of 105 ± 45 days (range 33-169 days).

8.3.3 Total body water and water turnover rate

Total body water (TBW) was not significantly different between males and females for either season (summer: $F=0.28$, $F_{0.05(1),1,5}=6.61$, $P>0.05$; winter: $F=0.33$, $F_{0.05(1),1,4}=7.71$, $P>0.05$), or between seasons ($F=1.43$, $F_{0.05(1),1,13}=4.67$, $P>0.05$; Table 8.1). Mean TBW for re-introduced *P. x. celeris* at Lambert Station was $73.1 \pm 5.8\%$ over the measurement periods.

Water turnover rate (Table 8.1) was not significantly different between males and females within either season (summer: $F=0.02$, $F_{0.05(1),1,7}=5.59$, $P>0.05$; winter: $F=0.12$, $F_{0.05(1),1,6}=5.99$, $P>0.05$). However, pooled summer and winter samples were significantly different ($F=206.6$, $F_{0.001(1),1,14}=17.1$, $P<0.001$), with mass-corrected

Table 8.1 Body mass (BM), body mass change (BMC), total body water, water turnover rate and field metabolic rate of re-introduced *Petrogale xanthopus celeris* at Lambert Station. Data are shown as means \pm standard deviation. † Within columns, pooled means differ significantly at $P < 0.001$. Difference was assessed by one-factor ANOVA for unequal replication.

Animal	BM	BMC	Total Body Water		Water Turnover Rate		Field Metabolic Rate		
	g	%d ⁻¹	g	%	mL d ⁻¹	mL.kg ⁻¹ .d ⁻¹	mL.CO ₂ g ⁻¹ h ⁻¹	kJ d ⁻¹	kJ.kg ⁻¹ .d ⁻¹
WET SUMMER									
M28	6750	-	4913.1	72.8	685.2	176.6	0.695	2387.8	788.9
M24	6850	-	4877.0	71.2	504.2	128.6	0.853	2974.5	974.3
M9	6900	-	-	-	704.1	178.7	-	-	-
C1M1	5300	-	4021.7	75.9	607.9	186.0	0.453	1221.3	464.2
Male sub-total	6450 \pm 666	-	4603.9 \pm 412.0	73.3 \pm 2.0	625.4 \pm 78.7	167.5 \pm 22.7	0.667 \pm 0.164	2194.5 \pm 728.7	742.5 \pm 210.8
F41	5400	-	4038.4	74.8	644.5	194.6	0.761	2089.8	785.8
F29	5150	-	3657.8	71.0	531.4	166.0	0.367	961.4	371.6
F36	5250	-	3838.6	73.1	607.1	187.0	0.297	793.6	303.3
F17	5650	-	4002.2	70.8	635.9	186.0	0.327	940.7	344.6
Female subtotal	5362 \pm 188	-	3884.3 \pm 150.9	72.4 \pm 1.6	604.7 \pm 44.5	183.4 \pm 10.6	0.438 \pm 0.188	1196.4 \pm 520.0	451.3 \pm 194.6
SUMMER TOTAL	5906 \pm 731	-	4192.7 \pm 461.1	72.8 \pm 1.8	615.0 \pm 64.8†	175.4 \pm 19.4†	0.536 \pm 0.211	1624.1 \pm 791.2	576.1 \pm 247.9
DRY WINTER									
M28	7060	-0.041	5843.2	82.8	113.1	28.2	0.685	2461.5	792.3
M24	7240	-0.141	5284.3	73.0	240.3	58.9	0.574	1445.0	458.4
C1M1	5840	-0.177	4452.1	76.2	252.4	72.1	0.392	1704.1	612.3
Male sub-total	6713 \pm 622	-0.12 \pm 0.06	5193.2 \pm 571.6	77.3 \pm 4.1	201.9 \pm 63.0	53.1 \pm 18.4	0.550 \pm 0.121	1870.2 \pm 431.2	621.0 \pm 136.5
F27	5520	-0.061	4124.2	74.7	136.4	40.6	0.895	2513.7	933.2
F17	5660	-0.121	4372.8	77.3	187.2	54.7	0.450	1296.7	474.5
15(F)	4900	0	3624.8	74.0	170.9	55.3	0.641	1599.0	636.1
F30	5680	0	4225.5	74.4	177.2	51.6	0.254	735.0	268.4
43(2F)	4580	0	3706.9	54.9	117.3	30.2	0.709	1653.0	683.9
Female subtotal	5268 \pm 446	-0.04 \pm 0.05	4010.8 \pm 293.7	71.1 \pm 8.2	157.8 \pm 26.5	46.5 \pm 9.7	0.590 \pm 0.220	1559.5 \pm 577.9	599.2 \pm 221.4
WINTER TOTAL	5810 \pm 871	-0.06 \pm 0.06	4454.2 \pm 710.0	73.4 \pm 7.6	174.3 \pm 48.8†	49.0 \pm 14.0†	0.575 \pm 0.190	1676.0 \pm 548.7	607.4 \pm 194.2

summer WTR being over 350% greater than winter WTR. WTR was independent of mass in either season (summer: $F=0.03$, $F_{0.05(1),1,6}=5.99$, $P>0.05$; winter: $F=0.41$, $F_{0.05(1),1,6}=5.99$, $P>0.05$) and for either sex (male: $F=1.77$, $F_{0.05(1),1,4}=7.71$, $P>0.05$; female: $F=1.44$, $F_{0.05(1),1,6}=5.99$, $P>0.05$). WTR was independent of joey age in lactating females ($F=0.27$, $F_{0.05(1),1,7}=5.59$, $P>0.05$). Although sample sizes were low, WTR did not vary between sites in either season (summer: $F=1.00$, $F_{0.05(1),1,7}=5.59$, $P>0.05$; winter: $F=2.50$, $F_{0.05(1),2,5}=5.79$, $P>0.05$).

8.3.4 Field metabolic rate

Field metabolic rate (Table 8.1) remained relatively constant between seasons at 592.8 ± 229.2 kJ kg^{0.58} d⁻¹. Although there was a tendency for males to have higher FMRs, there was no significant difference between the sexes within either season (summer: $F=0.81$, $F_{0.05(1),1,6}=5.99$, $P>0.05$; winter: $F=0.86$, $F_{0.05(1),1,6}=5.99$, $P>0.05$). Total FMR (kJ.day⁻¹) was dependent on body mass in summer ($F=12.7$, $F_{0.02(1),1,5}=11.3$, $P<0.02$), but not winter ($F=0.20$, $F_{0.05(1),1,6}=5.99$, $P>0.05$). FMR of lactating females was independent of joey age ($F=0.001$, $F_{0.05(1),1,7}=5.59$, $P>0.05$). Although sample sizes were low, FMR did not vary between sites (summer: $F=2.00$, $F_{0.05(1),1,6}=5.99$, $P>0.05$; winter: $F=1.00$, $F_{0.05(1),2,5}=5.79$, $P>0.05$).

8.3.5 Food intake rate

The proportions and water content of plants selected by *P. x. celeris* at Lisburne Station during similar climatic conditions to the current study are shown in Table 8.2 (Allen 2001). Results indicate that the mean water content of the diet of *P. x. celeris* was 59% in summer, but declined to 41% in winter during a year of similar rainfall.

On a metabolic body-mass basis, dry matter intake did not significantly differ between sexes or seasons (Table 8.3). Water influx from preformed water was considerably higher in summer (190.2 mL.d⁻¹) than winter (105.3 mL.d⁻¹), however metabolic water remained relatively constant between seasons (56.0 mL.d⁻¹ and 56.9 mL.d⁻¹ respectively). The greatest and most significant change in water influx occurred in free water consumption between sexes ($F=7.06$, $F_{0.025(1),1,11}=6.72$, $P<0.025$) and between seasons ($F=107.2$, $F_{0.001(1),1,11}=19.7$, $P<0.001$). Females consumed 62.4 ± 5.6 mL.kg^{0.71}d⁻¹ of free water

throughout sampling while males consumed $38.9 \pm 6.8 \text{ mL} \cdot \text{kg}^{-0.71} \text{ d}^{-1}$. For both sexes, free water consumption in summer was $96.5 \pm 6.4 \text{ mL} \cdot \text{kg}^{-0.71} \text{ d}^{-1}$ when free water was available from rainfall (Ch. 6), but this decreased to $4.7 \pm 6.1 \text{ mL} \cdot \text{kg}^{-0.71} \text{ d}^{-1}$ in winter when no free water was available within traveling a minimum distance of 0.5 km (Site 2). Free water intake in winter was likely from dew, both in the air and on foliage.

Table 8.2 Mean plant water content in the diet of *Petrogale xanthopus celeris* on Lisburne Station under similar climatic conditions to those during water turnover rate measurements on Lambert Station. *P. x. celeris* diet and plant water contents from Allen (2001).

Plant group	Species	Wet Summer			Dry Winter		
		% of diet	% H ₂ O		% of diet	% H ₂ O	
Grasses	<i>Brachiaria gilesii</i>	34.0	69.7	23.7%	12.5	41.6	5.2%
	<i>Sporobolus caroli</i>	2.7	52.0	1.4%	1.4	32.1	0.4%
Browse	<i>Acacia petraea</i>	45.7	50.0	22.9%	63.4	41.8	26.5%
	<i>Capparis spp.</i>	16.3	62.6	10.2%	21.4	38.5	8.2%
Forbs	<i>Sida spp.</i>	1.5	61.8	0.9%	1.3	36.7	0.5%
TOTAL DIET		59.1%			40.8%		

Table 8.3 Seasonal field metabolic rates, food intake rates and water influx rates of re-introduced *Petrogale xanthopus celeris* on Lambert Station. Data are means \pm S.D., with sample size indicated in parentheses next to sex. Data was analysed for significant differences between sex and season using two-factor ANOVA for unequal replication. Food intake rate was not significantly different between sexes or seasons. Water influx rate through drinking ($\text{mL} \cdot \text{kg}^{-0.71} \text{ d}^{-1}$) significantly varied between sexes (males: 38.9 ± 6.8 , females: 62.4 ± 5.6 ; $F=7.06$, $F_{0.025(1),1,11}=6.72$, $P<0.025$) and between seasons (summer: 96.5 ± 6.4 , winter: 4.7 ± 6.1 ; $F=107.2$, $F_{0.001(1),1,11}=19.7$, $P<0.001$).

Parameter		Wet Summer		Dry Winter	
		Males (3)	Females (4)	Males (3)	Females (5)
		$6.45 \pm 0.66 \text{ kg}$	$5.36 \pm 0.19 \text{ kg}$	$6.71 \pm 0.62 \text{ kg}$	$5.27 \pm 0.45 \text{ kg}$
Field metabolic rate	KJ d ⁻¹	2194.5 ± 728.7	1196.4 ± 520.0	1870.2 ± 431.2	1559.5 ± 577.9
	kJ kg ^{-0.58} d ⁻¹	742.5 ± 210.8	451.3 ± 194.6	621.0 ± 136.5	599.2 ± 221.4
Food intake rate	g DM d ⁻¹ A	219.5 ± 72.9	119.6 ± 52.0	187.0 ± 43.1	156.0 ± 57.8
	g DM kg ^{-0.75} d ⁻¹	54.1 ± 17.8	33.9 ± 16.9	45.0 ± 12.1	45.9 ± 19.8
	g fresh matter d ⁻¹ B	523.8 ± 213.1	285.6 ± 143.3	316.5 ± 89.4	263.9 ± 109.3
Water influx rate	mL day ⁻¹				
	Total (from HTO)	625.4 ± 78.7	604.7 ± 44.5	201.9 ± 63.0	157.8 ± 26.5
	Preformed in food C	243.1 ± 217.6	137.3 ± 23.5	115.8 ± 14.8	94.7 ± 26.3
	Metabolic (from FMR) D	72.4 ± 29.4	39.5 ± 19.8	63.2 ± 22.2	50.5 ± 19.6
	Drinking E	283.6 ± 111.9	428.0 ± 27.3	23.0 ± 80.8	12.6 ± 48.5
	Drinking mL kg ^{-0.71} d ⁻¹ *	71.5 ± 25.7	121.4 ± 6.8	6.2 ± 19.7	3.3 ± 14.6

Key to Table 8.3 calculations:

- A Gross energy content of green vegetation is about 21 kJ/g dry matter, equivalent to a metabolisable energy content of about 10 kJ/g DM when fed to tamar wallabies (Nagy *et al.* 1990a, b; Nagy *et al.* 1999).
- B Grams DM to grams fresh matter: $DM / (\% \text{ dry} / 100)$ ie $219.5 / 0.419$ (summer water content 59.1%).
- C Performed in food: grams fresh matter x water content - 59.1% in summer, 40.9% in winter (Table 8.2).
- D Metabolic water: oxidation of foodstuffs yields 0.0007 ml H₂O per ml⁻¹ CO₂ (Nagy and Martin 1985; Munks 1990), $FMR (\text{mL CO}_2 \text{ g}^{-1} \text{ h}^{-1}) \times \text{mass (g)} \times 24 \text{ h} \times 0.0007 (\text{H}_2\text{O per ml}^{-1} \text{ CO}_2) = \text{metabolic water (mL.day}^{-1})$.
- E Drinking (total – performed and metabolic) includes consumption of dew and free water.

8.4 Discussion

Packed cell volume (PCV) and total plasma protein (TPP) are useful indicators of dehydration in animals, with increases in either indicating a plasma volume deficit (Carlson 1997). TPP during winter sampling was significantly higher than during summer sampling (Table 7.6), and significantly higher than any other time post-release. The highest TTP recorded post-release coincided with the longest period without rainfall post-release. TTP during summer sampling was not significantly different from that of other sampling periods, thus TTP indicates that *P. x. celeris* were dehydrated during the winter sampling. However, PCV was not significantly different between sampling periods (Table 7.3), indicating hyperproteinemia caused by dehydration (Carlson 1997). However, stress indices such as creatinine did not change (Table 7.6). The difference in mean TTP concentration between the two sampling periods indicates a winter plasma volume deficit of 21%. Despite this, re-introduced *P. x. celeris* at Lambert Station maintained TBW at $73.1 \pm 5.8\%$ throughout the sampling period, with no significant difference between sexes or seasons. Results for TBW are similar to those from the sympatric *M. r. erubescens* ($73.1 \pm 0.5\%$; Dawson *et al.* 1975), *M. giganteus* ($73.5 \pm 0.4\%$; Blaney *et al.* 2000), *M. rufus* ($73.9 \pm 0.8\%$; Blaney *et al.* 2000), and also *M. eugenii* ($73.3 \pm 4.7\%$; Nagy *et al.* 1990a).

Total body water has often been used as an indicator of body condition as it varies inversely with body fat, thus a high TBW percentage indicates poor body condition (Holleman and Dieterich 1973; Bakker and Main 1980; Catt 1981; Reimer and Hindell 1996; Gibson and Hume 2000). *P. x. celeris* maintained TBW and thus body condition throughout the dry winter, consistent with the finding of no mass loss between seasons (Table 8.1), consistent general condition scores (Ch. 7) and continuous breeding by all

females throughout the year. Furthermore, TBW indicates re-introduced *P. x. celeris* remained healthy throughout the driest period recorded since release. In contrast, wild Bilbies (*Macrotis lagotis*) lost body condition in summer, possibly as a result of reduced food availability or increased thermoregulatory demands (Gibson and Hume 2000), while *M. rufus* lost body condition in winter due to reduced rainfall and pasture biomass (Moss and Croft 1999). Hence, captive-bred *P. x. celeris* retained the ability to cope with seasonal fluctuations in water and food availability, despite the absence of similar stresses in captivity.

The summer ($101 \text{ mL kg}^{-0.71} \text{ d}^{-1}$) and winter ($276 \text{ mL kg}^{-0.71} \text{ d}^{-1}$) WTR patterns reported by Lim *et al.* (1987) follow trends opposite to those found in the current study (summer $175.4 \text{ mL kg}^{-0.71} \text{ d}^{-1}$; winter $49.0 \text{ mL kg}^{-0.71} \text{ d}^{-1}$), but both follow local rainfall patterns. The higher WTR in summer in *P. x. celeris* is likely the result of increased evaporation and water required for thermoregulation (sweating) in temperatures regularly in excess of 40°C throughout the species' range (Dawson and Denny 1969; Withers 1992; BOM 2001), in addition to the higher water content of consumed forage.

The winter WTR of *P. x. celeris* in the current study ($49.0 \text{ mL kg}^{-0.71} \text{ d}^{-1}$) is among the lowest recorded for any marsupial (Hume 1999). Similarly low WTRs have been reported for dehydrated captive *L. onspicillatus* ($51 \text{ mL kg}^{-0.71} \text{ d}^{-1}$; Bakker and Bradshaw 1983), and for free-ranging *B. penicillata* and *S. brachyurus* (both $57 \text{ mL kg}^{-0.71} \text{ d}^{-1}$; Green 1989 and Nagy *et al.* 1990a respectively). There are few other reports of seasonal rates of water turnover in macropods; winter WTR in *P. x. celeris* is most akin to *P. rothschildi* ($63 \text{ mL kg}^{-0.71} \text{ d}^{-1}$ in a dry spring; Green 1989), a species occupying similar habitat and climatic niche in Western Australia, rather than to *P. x. xanthopus*. This finding supports the suggestion by Hume (1999) that nutritional requirements of macropods are more closely related to habitat than to phylogeny.

The average water intake of captive *P. x. celeris* was $98 \text{ mL kg}^{-0.80} \text{ d}^{-1}$ (Murphy 1985), equivalent to $113 \text{ mL kg}^{-0.71} \text{ d}^{-1}$, thus lower than the summer WTR of re-introduced *P. x. celeris* when free water was available ($175 \text{ mL kg}^{-0.71} \text{ d}^{-1}$). The winter WTR of wild *P. x. xanthopus* ($276 \text{ mL kg}^{-0.71} \text{ d}^{-1}$; Lim *et al.* 1997) is the highest WTR recorded for any macropod (Green 1989; Hume 1999). Lim *et al.* (1987) suggested that this high WTR came from preformed water in the diet due to recent rainfall; no animals were sighted or

radio-tracked near permanent sources of free water. However, it is difficult to explain such a high WTR without consumption of free water. From Lim *et al.* (1987) I calculated the water content of the plants consumed by *P. x. xanthopus* during winter was approximately 63% using the same approach as in Table 8.2. On this basis *P. x. xanthopus* was consuming 2.12 kg of fresh plant material per day ($150 \text{ ml kg}^{-1} \cdot \text{day}^{-1} \times 8.9 \text{ kg body mass} = 1335 \text{ ml} \times 63\% \text{ water content} = 2119 \text{ g fresh matter d}^{-1}$). This is over 20% of their body mass. Metabolic water would account for less than $100 \text{ ml} \cdot \text{d}^{-1}$ (Green 1989). Dry matter intake calculated from FMR (Nagy *et al.* 1990b) would be approximately 221 g per day, equivalent to 600 g of fresh matter. This leaves 1.4 kg of fresh matter or approximately 800 mL of water influx unaccounted for. This amount is well outside any errors inherent in the above calculations, and can only be accounted for by free water intake.

Dawson and Denny (1969) and Bakker and Bradshaw (1983) reported that plasma urea concentration increased significantly in dehydrated *M. rufus* and *M. r. erubescens*, and *L. conspicillatus* respectively. Plasma urea concentration in the current study remained constant during winter sampling despite dehydration, supporting the suggestion by Murphy (1985) of an enhanced capacity by *P. x. celeris* to reabsorb filtered urea. Furthermore, the four animals sampled in both seasons gained mass ($313 \pm 193 \text{ g}$) between summer and winter, indicating that *P. x. celeris* is well adapted to deal with water stress in its natural semi-arid environment.

Generally WTR, FMR and food intake rate in female marsupials are correlated with age of pouch young and thus lactational demands (Green 1989, 1997; Munks 1990; Hume 1999). However, in the current study there were no significant differences in WTR or FMR between males and lactating females in either season (comparison with non-lactating females was not possible as all females carried pouch young). Furthermore, joey age was independent of WTR or FMR when assessed by simple linear regression. These findings indicate no measurable water or energy demands of lactation on *P. x. celeris*. The same finding has been reported for other macropodoid marsupials including *A. rufescens* (Wallis and Green 1992), *B. penicillata* and *M. eugenii* (Green 1997). Green (1997) suggested that this was due to macropods usually having only a single pouch young and relatively low rates of milk production. A range of ages among the *P. x. celeris* pouch

young would also make it difficult to detect a significant affect of lactation on FMR and WTR, as both peak close to permanent pouch evacuation (Green 1997).

Field metabolic rate was similar between summer and winter, averaging 592.8 ± 229.2 kJ $\text{kg}^{0.58} \text{d}^{-1}$. A similar FMR ($622 \text{ kJ kg}^{-0.58} \text{d}^{-1}$) was reported by Green (1989) for one specimen of *P. x. xanthopus*. The tendency for a larger winter FMR was likely a result of higher thermoregulatory demands resulting from low winter temperatures. A similar finding was reported for *A. rufescens* (Wallis and Green 1992).

Dry matter intake (DMI) of captive *P. x. celeris* was $42 \text{ g kg}^{0.75} \text{d}^{-1}$ (Murphy 1985) and of wild *P. x. xanthopus* $42.9 \text{ g kg}^{0.75} \text{d}^{-1}$ (Nagy *et al.* 1990b). Mean DMI of re-introduced *P. x. celeris* was $44.7 \text{ g kg}^{0.75} \text{d}^{-1}$. Thus DMI varied little with season, diet or location. This is surprising given the additional energy demands of free existence compared with captivity. For instance, Dellow and Hume (1982) reported voluntary DMIs of captive *M. eugenii* of $29.4 \text{ g kg}^{0.75} \text{d}^{-1}$, much less than the $38.0 \text{ g kg}^{0.75} \text{d}^{-1}$ calculated by Nagy *et al.* (1990b) in free living *M. eugenii*.

Dry matter intake and consequently fresh food intake tended to be higher by males than females. Conversely, females drank significantly more free water than males throughout sampling ($62.4 \pm 5.6 \text{ mL.kg}^{-0.71} \text{d}^{-1}$ compared to $38.9 \pm 6.8 \text{ mL.kg}^{-0.71} \text{d}^{-1}$). The higher fresh food intake of males provided an additional 141 mL of preformed water in summer and 22 mL in winter. Thus, males possibly increased food intake to obtain preformed water on and close to their hill refuge rather than drinking free water collected on the surrounding plain, thus reducing their risk of fox (*Vulpes vulpes*) predation. All females predated (known or suspected) by *V. vulpes* (n=5) were killed during months of high rainfall (Fig. 5.3, 6.12) and were found on the plain. However, no rain fell in the months two males died of suspected fox predation, and their remains were found on the hill. Greater foraging for free water by females than males may explain the slight female bias than occurred in fox predation and why remains of each sex were found in different locations. However, much larger sample sizes would be needed to test this suggestion more rigorously. Furthermore, feral goats (*Capra hircus*) have a similar diet to *P. xanthopus*, particularly during drought (Dawson and Ellis 1979; Allen 2001). In areas where *C. hircus* numbers are not controlled they are likely to remove most of the browse on hills to a height above the reach of *P. xanthopus* (Lim 1987; Allen 2001). Thus, *P. xanthopus* would

be forced to forage on the plain or travel to free water, both of which would increase the likelihood of *V. vulpes* predation.

Current results indicate a difference in free water usage between *P. x. celeris* and *P. x. xanthopus*. Despite dehydration in winter and a mean free water influx of only 4.7 mL.kg^{0.71}d⁻¹, likely through breathing dew-laden air, re-introduced *P. x. celeris* at Lambert Station did not travel less than 500 m (Site 2) to free water, whereas *P. x. xanthopus* under similar rainfall conditions at Middle Gorge travelled up to 2 km (Lim 1987; Lim *et al.* 1987). *P. xanthopus* rely on browse species as their main food source during dry periods (Dawson and Ellis 1979; Copley and Robinson 1983; Allen 2001). Browse species retain higher water content throughout dry periods than other plant groups (Table 8.2; Allen 2001). Thus, switching their diet from grasses to browse increases water influx through preformed water. The greater occurrence of browse species at Lambert Station (Ch. 5) than Middle Gorge (Copley and Robinson 1983; Lim *et al.* 1987) should sustain *P. x. celeris* further into drought before a source of free water is required. Furthermore, the high incidence of rainfall in summer throughout the range of *P. x. celeris*, when animals are more likely to require water for thermoregulation, would reduce the sub-species' need to travel to free water. The difference in rainfall patterns and the greater occurrence of browse throughout the range of *P. x. celeris* than *P. x. xanthopus* may explain why extant *P. x. celeris* colonies do not occur near free water. Thus, it is possible that both subspecies of *P. xanthopus* may utilize free water only if it lies within range, but are not strictly reliant on free water for their survival.

The maintenance of total body water, field metabolic rate, high condition scores and fecundity during the dry winter sampling indicates that *P. x. celeris* is well adapted to cope with water stress and the seasonality of its semi-arid environment. Furthermore, captive breeding seems unlikely to have reduced the animal's ability to adapt its nutritional requirements to a new environment upon release, as suggested by Lapidge (2000). However, the possibility that the re-introduced *P. x. celeris* may need to travel to free water during future droughts cannot be ruled out.

GENERAL DISCUSSION

9.1 Key findings

The aims of the study, as detailed in Chapter 1, were to:

1. **re-introduce captive-bred *P. x. celeris* in accordance with established and original reintroduction techniques and identify successful methods of reintroduction.**

Petrogale x. celeris were re-introduced to Lambert Station after a 20 year absence on August 9, 1998, in accordance with the published guidelines of Kleiman (1989), Stanley Price (1989), Short *et al.* (1992), Kleiman *et al.* (1994) and IUCN (1998) as well as original criteria introduced in Chapters 2, 3 and 4. Success criteria proposed in Chapter 4 for the *P. x. celeris* reintroduction were that:

- i. baseline data on captive *P. x. celeris* were established,
- ii. the majority of *P. x. celeris* released on Lambert Station survived,
- iii. the cause of any *P. x. celeris* deaths was determined,
- iv. breeding occurred by the release animals and their offspring,
- v. insight into ecological, physiological and genetic adaptation of released *P. x. celeris* was obtained,
- vi. results were analysed and published in both thesis and journal format, and
- vii. a viable, free-ranging and self-sustaining meta-population of *P. x. celeris* was formed.

Extensive baseline data was established on captive *P. x. celeris* prior to release (Ch. 2-3) and a minimum of 12 of 24 *P. x. celeris* released remained alive 2.5 years post-release (Ch. 6). The cause or suspected cause of most animal deaths was determined (Ch. 6), and 33 1st generation and four 2nd generation offspring were recorded (Ch. 6). Original insight into the ecological and physiological adaptation of captive-bred *P. x. celeris* to the wild, as well as the species in the wild, occurred in Ch. 6, 7 and 8 and is summarized below.

Genetic results will be fore coming, and already indicate *P. xanthopus* colonies are not based on a strict male dominance hierarchy (as different males simultaneously sired young), wild males (confirmed through genetics) that immigrated to Site 1 have sired offspring and added to the gene pool, and the male that undertook the exploratory movement from Site 1 sired offspring at Site 2 (M. Eldridge, pers. comm.). Currently the thesis and one paper (Appendix II) have been published from the research, with at least five more papers in preparation for submission to international journals post graduation. At present, a viable and free-ranging population of *P. x. celeris* occurs on Lambert Station. Whether the population is self-sustaining in the long-term can only be demonstrated with time. Consequently, the re-introduction of *P. x. celeris* to Lambert Station has met all success criteria established at the projects outset and thus should be declared a short-term success.

However, success criteria vary between reintroduction practitioners as no formal criteria have been established by the IUCN (1998) or in the literature. Morris (2000) proposed criteria for Western Australian marsupial reintroductions, namely:

- a) initial mortality of founders < 30 per cent
- b) breeding (presence of pouch young)
- c) recruitment to population
- d) first generation breeding
- e) population expansion.

According to these criteria, the re-introductions of *P. x. celeris* and *P. x. xanthopus* are both successful, as initial mortality was 21% for *P. x. celeris* and 10% for *P. x. xanthopus*, both populations have bred and recruited young to the population, 1st generation wild-born offspring have produced 2nd generation offspring at both sites, and both populations are expanding (Fig. 6.11 and 6.21; actual population between Potential and KTBA).

The re-introductions of *P. x. celeris* and *P. x. xanthopus* used similar techniques (detailed extensively in Ch. 2 and 3), namely a 'hard' and direct release to the wild without supplementary provisioning and pre-release training limited to enhancing recapture (*P. x. celeris* only). As both reintroductions have succeeded in the short-term, re-introduction methods can be seen as successful. Furthermore, as *P. xanthopus* readily adapted to the wild in all ecological and physiological parameters measure, this re-introduction technique is ethical for the species. Moreover, as biological findings for 'adapted' *P.*

xanthopus were similar to other wild *Petrogale* species (Ch. 6-8), the technique is likely to be suitable for the more endangered Black-footed Rock-wallaby (*P. lateralis*) in South Australia and Brush-tailed Rock-wallaby (*P. penicillata*) in Victoria, Australia.

2. determine how captive-bred *P. x. celeris* adjust to the wild ecologically, specifically through studies on fecundity, survival, home range and dispersal, and physiologically, specifically through studies on growth, haematology, biochemistry, vitamin E, condition, water turnover rate, field metabolic rate and food intake rate, and genetically by directly comparing data gathered both pre- and post-release.

All ecological and physiological studies proposed were undertaken successfully and are detailed in Chapters 6, 7 and 8. Pre-release genetic diversity of release animals is detailed in Chapter 3. Unfortunately, post-release changes in genetic diversity and ascertaining paternity of all offspring has not been completed due to unforeseeable problems encountered with outsourced genetic analysis (M. Eldridge, Macquarie University). Findings will however be published once analysis is complete.

3. undertake comparable ecologically and physiologically monitoring of the previously re-introduced *P. x. xanthopus* to that of *P. x. celeris*; and

Between April 1998 and January 2001 studies on fecundity, survival, home range, dispersal, growth, haematology, biochemistry, vitamin E, and condition were undertaken on *P. x. xanthopus*. Although conducted on *P. x. celeris*, studies on water turnover rate, field metabolic rate and food intake rate were not undertaken on *P. x. xanthopus*. The principal reasons for this were the expense (\$13,000 in materials alone for *P. x. celeris*), and that water turnover and field metabolic rate had previously been determined for the sub-species at Middle Gorge (Lim *et al.* 1997).

Success criteria proposed for the re-introduction of *P. x. xanthopus* in Chapter 4 were:

- i. the establishment of a exotic predator-free sanctuary and buffer zone,
- ii. the survival of released *P. x. xanthopus*,
- iii. breeding by the release animals and their offspring,
- iv. the cause of any *P. x. xanthopus* deaths be identified,

- v. promotion of the reintroduction to the international zoo community,
- vi. a sense of project ownership by the local community, and
- vii. the formation of a viable, free-ranging and self-sustaining population of *P. x. xanthopus*.

In relation to the trapping program, additional success criteria were:

- viii. the regular capture of all remaining released and wild-born *P. x. xanthopus* for physiological monitoring,
- ix. the collection of data on population growth and seasonal health trends for *P. x. xanthopus*.

In addressing success criteria, the exotic predator-free area within and around Aroona Sanctuary has been expanded from 600 km² to over 2000 km² through the Aroona Catchment Biodiversity Enhancement Project, three of 12 (25%) founders remain five years post-release (Ch. 6), first and second generation offspring have been recorded (Ch. 6), the causes of many deaths were identified (Ch. 6), and the project has received international exposure and sponsorship (Ch. 4). Although the Royal Zoological Society of South Australia remains involved with the project, the Leigh Creek community, through the Natural Heritage Trust-funded Aroona Catchment Biodiversity Enhancement Project, has mustered 10,000 feral goats, destroyed 3,000 rabbit warrens, largely eradicated foxes and cats from the Aroona catchment, removed exotic weeds, revegetated large tracts of degraded land surrounding the sanctuary, constructed two interpretative walking trails that highlight both project successes and the unique local arid-zone biota, and actively promoted the re-introduction and their own environmental achievements through field days, newsletters, pamphlets, posters and the Internet. Future re-introduction success can no doubt be contributed to the Leigh Creek community.

A viable, free-ranging and self-sustaining (five years) population of *P. x. xanthopus* now exists in Aroona Sanctuary. Since trapping related to this project ceased in January 2001, a further six new animals have been captured and marked by RZSSA staff (D. Schultz pers. comm.). Released and wild-born *P. x. xanthopus* were captured regularly throughout the study allowing population growth and seasonal health trends to be monitored (Ch. 6 and 7). Furthermore, the reintroduction has now surpassed the five-year barrier, the minimum time proposed by Short *et al.* (1992) and Short and Turner (2000) at which success can be assessed. Thus, the re-introduction and monitoring of *P. x. xanthopus* has also achieved its objectives and is a success.

4. compare post-release results from the two re-introductions to pre-release data for each sub-species, to each other, to previous research on both sub-species in the wild, and to findings for other re-introduced animals to examine the applicability of re-introduction as a conservation technique for *P. xanthopus*.

Post-release results for both sub-species have been compared to each other, wild-born *P. xanthopus*, and other re-introduced and wild marsupials and eutherian mammals in Chapters 5-8 and below. The applicability of re-introduction as a conservation technique will be discussed later in this chapter. Following is a summary of the key findings from the study.

Lambert Station in Queensland received on average 250% greater monthly rainfall than Aroona Sanctuary in South Australia since the respective releases of *P. xanthopus* occurred in August 1998 and September 1996. Furthermore, release sites at Lambert Station contained on average 156% greater coverage of vegetation throughout the study than the release site at Aroona Sanctuary, principally in the form of *Acacia* trees.

Trap training captive *P. xanthopus* prior to release ensured a significantly higher recapture rate post-release, enabling greater ability to study individual animal adaptation to the wild. The technique is recommended to facilitate post-release monitoring for future reintroductions of species not easily recaptured in the wild.

Reintroduced *P. x. xanthopus* (n=12) were generally older and larger than reintroduced *P. x. celeris* (n=24). Age, mass, sex or condition at the time of release did not significantly affect short-term survival post-release. A similar finding was reported for translocated Brush-tail possums (*Trichosurus vulpecula*) in New Zealand. However, this is in contrast to Golden-lion tamarins (*Leontopithecus rosalia*) and South Island saddlebacks (*Philesturnus carunculatus carunculatus*) where juveniles survived better (Kleiman *et al.* 1996 and Pierre 1999 respectively), Bighorn Sheep (*Ovis canadensis*) and Greater Bilbies (*Macrotis lagotis*) where adults survived better (Ostermann *et al.* 2001 and Southgate *et al.* 2000 respectively), Quokkas (*Setonix brachyurus*) and *P. penicillata* where males survived better (Short *et al.* 1992), and *O. canadensis* where females survived better (Ostermann *et al.* 2001). This comparison indicates survivorship characteristics of re-introduced animals vary between species and cannot be generalized.

Survival of re-introduced *P. x. celeris* was 63% after 1 year, 54% after 2 years, and 50% after 2.5 years (the cessation of trapping). Survival of re-introduced *P. x. xanthopus* was 67% after 1 year, 42% after 2 years, 33% after 2.5 years, and is currently 25% 5 years post-release. Thus, mortality was higher for *P. x. xanthopus* at 2.5 years post-release, likely from natural attrition of older animals; only 10% of *P. x. xanthopus* normally survive past 6 years of age (Lim 1987). Thus, the release of younger, sexually mature adults is recommended for longevity of founding populations.

First-year survival of reintroduced *P. xanthopus* was higher than that of *L. rosalia* (<59%; Beck *et al.* 1991), Black-footed ferrets, *Mustela nigripes* (0-45% after 9 months dependent on rearing method; Biggins *et al.* 1998), *P. c. carunculatus* (50% at 8 months; Pierre 1999), Swift Foxes, *Vulpes velox* (15%, Ginsberg 1994), *O. canadensis* (60%; Ostermann *et al.* 2001), Parma wallabies, *Macropus parma* (0%, Short *et al.* 1992), *B. lesueur* and Golden Bandicoots, *Isodon auratus* (0% by 3 months; Christensen and Burrows 1994), and Eastern Barred Bandicoots, *Perameles gunnii* (8%; Dufty *et al.* 1994). Longer-term survival of captive-bred animals was however higher for Arabian Oryx, *Oryx leucomyx* (71% after 4 years; Stanley Price 1989), and similar for Red Wolf, *Canis rufus* (45% after 3 years; Phillips 1990) and *L. rosalia* (34% after 4+ years; Beck *et al.* 1991). Thus, survival of re-introduced *P. xanthopus* was more similar to that of larger eutherian mammals, which are reported to have a greater re-introduction success rate (Griffith *et al.* 1989; Stanley Price 1989).

Survival of re-introduced *P. x. celeris* did not vary significantly from captive *P. x. celeris*. To my knowledge, this is previously unreported for a captive-bred marsupial. Pouch-young survival was high in re-introduced females, with only two known losses. It is however possible more pouch young were aborted immediately prior to full-term, thus not affecting the 190-day gestation cycle of the dam on which survival was assessed. Juvenile survival was the lowest, with a minimum of 30% of *P. x. celeris* and 20% of *P. x. xanthopus* surviving to adulthood. No juvenile carcasses were found during trapping and other surveys of the release sites. Juveniles were consequently present and not trapped, predated and devoured or removed from the release site or not found, or dispersed from the release site. The low trappability of trap-shy juvenile animals, many of which were seen but not trapped and tagged, indicates percentages are underestimates. Survival of juveniles could not be monitored remotely through mortality radio-collars, as per adults,

due to animals being of insufficient mass to radio-collar. Thus, unless a juvenile was trapped during the last session they were considered dead and not included in KTBA estimates. Ideally, an expandable collar or ear-tag transmitter should be fitted to juveniles to obtain a more accurate estimate of juvenile *P. xanthopus* survival. Survival of sub-adults (mean 12 months old and 2.8 kg) was much higher at 73%. This is higher than wild *P. x. xanthopus* (58%) of the same age (Lim 1987), but similar to *M. lagotis* (Southgate and Possingham 1995)

Wild-born juvenile survival rates were less than that of reintroduced *L. rosalia* (67%; Beck *et al.* 1991), *C. rufus* (100%; Phillips 1990), and *O. leucoryx* (76%; Stanley Price 1989), similar to re-introduced Bilbies, *Macrotis lagotis* (25%; Southgate *et al.* 2000), but greater than wild Allied Rock-wallabies during drought, *P. assimilis* (12%; Delaney 1997). Results indicate juvenile survival in eutherian mammals may possibly be higher than that of marsupials. Southgate *et al.* (2000) and Delaney (1997) suggest low juvenile survival of marsupials is due to losses during weaning, and inability to obtain food and avoid predation. Survival of wild-born sub-adults (73%) was higher than that reported for wild *P. x. xanthopus* (58%; Lim 1987). This finding indicates first generation wild-born offspring of re-introduced captive-bred *P. xanthopus* are unlikely to be at a disadvantage compared to their always-wild counterparts.

In contrast to previous studies on captive and wild *P. x. xanthopus* in which male-biased sex ratios have been reported (Poole *et al.* 1985; Rix 1978; Lim 1987; Robinson *et al.* 1994), re-introduced *P. x. celeris* produced a female-biased sex ratio of offspring, and *P. x. xanthopus* a sex ratio close to unity. The studies on *P. x. xanthopus* were all on high-density colonies, which were possibly limited by resources. All re-introduced colonies were low-density, with *P. x. celeris* showing no evidence of resource limitation (100% fecundity, no deaths from starvation), while *P. x. xanthopus* was resource limited during drier times (as evident by the cessation of breeding and deaths from starvation). Thus results from the current study support previous suggestions (Clark 1978; Johnson 1989; Stuart-dick and Higginbottom 1989; Cockburn 1990; Fisher 1999) that sex determination is population density and resource dependent, with females produced to remain with the natal colony and facilitate population growth when resources are not limited.

Fecundity was related to vegetation abundance for both sub-species, but only dependent on rainfall for *P. x. xanthopus*. Similarly, vegetation abundance was only dependent on rainfall at Aroona Sanctuary and not Lambert Station. This is possibly because of the greater abundance of browse species at Lambert Station, which are less affected by short-term rainfall. Furthermore, *P. xanthopus* is known to rely on browse species in their diet during drier periods (Dawson and Ellis 1979; Copley and Robinson 1983; Allen 2001). Hence, population trends are only likely to be dependent on rainfall with where browse species form a small percentage of total vegetation available, as at Aroona Sanctuary.

Throughout its range, *P. xanthopus* has significantly higher fecundity in autumn and spring (Lim 1987; Robinson *et al.* 1994; Chapter 6), and so the species should be described as a semi-seasonal breeder. As temperature seasonality is the only consistent variable throughout the species' range, it is thought that females avoid the peak thermoregulatory demands of summer and winter. That these demands are significant is evident through significantly higher red blood counts in these seasons. Pouch life and growth rates of reintroduced *P. xanthopus* were not significantly different between sub-species or to those previously reported for the species (Poole *et al.* 1985). However, the greater sexual dimorphism seen in *P. x. celeris* is consistent with recent reports (Sharman *et al.* 1995). Fecundity was higher and sexual maturity earlier for re-introduced *P. x. celeris* than captive *P. x. celeris*. The same has previously been reported for reintroduced *M. lagotis* (Southgate and Possingham 1995; Southgate *et al.* 2000) and Burrowing Bettongs, *Bettongia lesueur* (Short and Turner 2000). The higher fecundity and earlier sexual maturity in reintroduced populations allow rapid population growth.

Home range of re-introduced *P. x. celeris* peaked at 12 months (15.9 ± 7.1 ha) post-release before slightly decreasing by 24 months (14.8 ± 6.6 ha). Core area continued to increase throughout the two-year tracking period (6.0 ± 3.6 ha 24 months post-release). Both home range and core area did not significantly differ from that of wild counterparts on Lambert Station by 12 months. Home range was also similar to that previously reported for the sub-species in similar habitat (20-40 ha; Sharp 1994), but significantly less than of wild *P. x. xanthopus* (130-210 ha; Lim 1987). Current findings support the suggestion by Sharp (1994) that of the two sub-species, *P. x. xanthopus* occupies more marginal habitat, thus requiring a larger home range to meet ecological and physiological needs. Home range of

re-introduced *P. x. celeris* was also similar to that reported for other *Petrogale* species and *Wallabia bicolor* (Short 1980; Troy and Coulson 1993; Horsup 1994).

Captive-bred Przewalski's horses (*Equus przewalskii*) re-introduced to semi-wild life in desert conditions similarly expanded their herd home range until 12 months (Pereladova *et al.* 1999). Home range establishment in other re-introduced captive-bred animals (*O. leucoryx* and *M. lagotis*) was reported to continue expanding from the release site for a number of years (Stanley Price 1986; Southgate and Possingham 1995). However, home range size was not analysed in discrete time-units in the latter studies, and therefore it is unknown whether it became asymptotic during this time.

Male-instigated overlap of female home ranges was significantly higher than that of female overlap of male ranges, male overlap of male ranges, or female overlap of female ranges. Male overlap of female home ranges averaged 64% throughout sampling and did not differ between Period's 1 and 5, therefore indicating overlap structure was likely established upon release and prior to home range expansion. Comparison of percentage overlaps between the same sex interactions versus opposite sex interactions, and between percentage overlaps within each sex indicated no significant difference. Both findings indicate males and females are distributed relatively evenly throughout the colony. Core areas formed by re-introduced *P. x. celeris* two-years post-release were 57% free of overlap from conspecifics and home ranges 46% free. This finding supports the suggestion by Lim (1987) that *P. xanthopus* are predominantly asocial within the colony. A similar finding has also been reported for *P. penicillata* (Jarman and Bayne 1997).

Dispersal of re-introduced *P. x. celeris* was the furthest recorded for any *Petrogale* species. One male dispersed 7.3 km to join another re-introduced colony, and another male undertook a minimum exploratory movement of 27 km over 12 months before returning to the same site. Initial genetic results indicate the male, from Site 1, sired young at another re-introduced colony (Site 2) where he spent several months (M. Eldridge pers. comm.). Long-distance dispersal has previously been reported for other reintroduced macropods (Short and Turner 2000; G. Lundie-Jenkins pers. comm.). A minimum of three wild male *P. x. celeris* immigrated to Site 1 during the study. Initial genetic results have confirmed that each male was not sired by re-introduced *P. x. celeris* (M. Eldridge pers. comm.). The closest known colony to the site is 17.2 km distant. Results indicate

that *P. x. celeris* has the ability to undertake long-distance dispersal, and is possibly more likely to do so in non-contiguous habitat. Moreover, dispersal did not occur between the closest colonies, indicating that male *P. x. celeris* showed preference for a site or possibly members of the other colony. Male dispersal between colonies on Lambert Station and breeding has resulted in the formation of a meta-population.

Most haematological and plasma biochemical parameters underwent significant decreases post-release, likely due to adoption of an appropriate natural diet (Lapidge 2000) and lower levels of stress. Haematology of re-introduced *P. xanthopus* was more similar to that of wild *P. assimilis* (Spencer and Speare 1992) than to captive *P. xanthopus*, indicating that the haematology of *P. xanthopus* is environmentally rather than phylogenetically based. Furthermore, most blood parameters post-release were related to thermoregulation, osmoregulation and heat stress rather than to the dietary influences previously reported for marsupials (Caughwin *et al.* 1984). Findings indicate that extrapolation of blood results obtained on captive animals to wild animals should be made with caution. Previous reports of blood chemistry changes in re-introduced captive-bred animals could not be found in the literature for comparison.

Plasma vitamin E concentration (PVEC) underwent the most significant change post-release of all measured blood parameters, and was the only parameter to increase upon re-introduction. Higher pre-release PVEC for over-crowded *P. x. celeris* than *P. x. xanthopus* is likely a result of dietary supplementation with *Acacia* leaves, a management technique recommended for other macropods and eutherian herbivores prone to capture myopathy. Higher PVEC in re-introduced *P. xanthopus* is likely related to both change in diet and reduction in stress, as indicated by other blood parameters. Increased fecundity in *P. x. celeris* post-release may also be PVEC-related.

Post-release water consumption has previously been monitored in re-introduced captive-bred *O. leucoryx* and *E. przewalskii* (Stanley Price 1986 and Pereladova *et al.* 1999 respectively). Both studies reported significant decreases in water consumption post-release as animals adapted to their new arid environment. For example, *O. leucoryx* decreased their water intake from 5 L d⁻¹ at release in 1980, to 2.5 L d⁻¹ by 1982 and preformed dietary water only by 1983 (Stanley Price 1986). However, neither study determined changes in total body water, field metabolic rate or food intake rate during

the same time period and thus the ability of captive-bred animals to adapt such parameters to the wild was previously unreported.

Re-introduced *P. x. celeris* maintained total body water ($73.1 \pm 5.8\%$) throughout a wet summer and a dry winter despite being dehydrated in the latter season. Water turnover rate of *P. x. celeris* was over 350% greater when free water was available after summer rain ($175.4 \text{ mL kg}^{-0.75} \text{ d}^{-1}$) than during winter drought ($49.0 \text{ mL kg}^{-0.75} \text{ d}^{-1}$). Mean water influx from drinking was $96.5 \pm 6.4 \text{ mL kg}^{-0.75} \text{ d}^{-1}$ in summer and $4.7 \pm 6.1 \text{ mL kg}^{-0.75} \text{ d}^{-1}$ in winter. Although the findings are opposite to the seasonal pattern reported for *P. x. xanthopus* at Middle Gorge, South Australia, they are consistent with rainfall patterns. The winter water turnover rate was among the lowest recorded for marsupials (Hume 1999), and eutherian mammals (Nagy and Perterson, 1988; Nagy 1994), with calculated rates of water influx indicating that animals relied on preformed and metabolic water and not free drinking water. Despite dehydration, *P. x. celeris* did not travel to a permanent free water source. Results indicate that *P. x. celeris* has an innate ability to cope with water stress. Greater consumption of browse species, with associated higher water content, during drier times (Dawson and Ellis 1979; Lim *et al.* 1987; Allen 2001) may alleviate the need to drink free water. Greater abundance of browse species throughout the range of *P. x. celeris* (Lim 1987; Lim *et al.* 1987; Chapter 5) may also decrease the sub-species reliance on drinking free water. Rates of water turnover in reintroduced *P. x. celeris* were most similar to wild Rothschild's Rock-wallabies, *P. rothschildi* (Green 1989) in a similar climatic zone and habitat in Western Australia, supporting the suggestion by Hume (1999) that macropod nutritive requirements are related to habitat rather than phylogeny.

Re-introduced *P. x. celeris* maintained field metabolic rate throughout summer and winter sampling ($592.8 \pm 229.2 \text{ kJ kg}^{-0.58} \text{ d}^{-1}$), with findings similar to the single wild *P. x. xanthopus* measured by Green (1989) ($622 \text{ kJ kg}^{-0.58} \text{ d}^{-1}$). Dry matter intake was consistent for re-introduced *P. x. celeris* ($44.5 \text{ g kg}^{-0.75} \text{ d}^{-1}$), captive *P. x. celeris* ($42 \text{ g kg}^{-0.75} \text{ d}^{-1}$; Murphy 1985) and wild *P. x. xanthopus* ($42.9 \text{ g kg}^{-0.75} \text{ d}^{-1}$; Green 1989; Nagy *et al.* 1990b) despite dietary and environmental differences. Fresh and consequently dry matter intake was significantly higher for males than females, and drinking of free water was significantly higher for females than males. This finding indicates a difference in foraging practices between the sexes.

There was no significant difference in water turnover rates or field metabolic rates between males and lactating females. Furthermore, water turnover rates and field metabolic rates were not correlated with age of pouch young for female *P. x. celeris*, suggesting no measurable additional water or energy demands for lactation in *P. x. celeris*. Similar results have previously been reported for other macropods (Wallis and Green 1992; Green 1997), and are possibly related to dams normally feeding only one pouch young and having relatively low rates of milk production (Green 1997). Alternatively, any increased demands of lactation may not be seen unless all pouch young are close to permanent pouch evacuation. In the present study the age of pouch young ranged from 33 to 169 days, reflecting the fact that breeding was continuous. Thus many pouch young were not close to pouch evacuation. Similar details on pouch young ages are not available from the studies of Wallis and Green (1992) and Green (1997).

The condition score index developed for the current study was directly related to independently measured PVEC, total body water, vegetation abundance and rainfall, suggesting it may be very useful as an indicator of animal health in future re-introduction programs. General condition of *P. x. celeris* improved significantly and stress indices were lower in all animals post-release. Despite seasonality in temperature, rainfall and vegetation abundance, reintroduced *P. x. celeris* maintained condition post-release, indicating an innate ability to cope with seasonal fluctuations in their unpredictable semi-arid environment. Published reports of this occurring for other captive-bred species could not be found.

9.2 Future monitoring of *P. x. celeris* and *P. x. xanthopus* re-introductions.

All radio-collared *P. x. celeris* captured by the end of the January 2001 trapping had their collar removed, after which traps and any other physical evidence of the study was removed from each site. Despite extensive predator control throughout the study, the most common cause of death for reintroduced *P. x. celeris* on Lambert Station was predation by introduced European Foxes (*Vulpes vulpes*) (Chapter 6; Lapidge and Henshall 2001; Appendix II). Future control of *V. vulpes* is essential if re-introduced colonies are to survive long-term. Management will consist of bi-annual (minimum) aerial

1080 baiting by the Lambert Station pastoralists and the Queensland Environmental Protection Agency. Pastoralists and field macropod harvesters will continue to eradicate predators sighted during their daily routines. Staff of the Queensland Environmental Protection Agency will visit each colony annually to search for evidence of continued occupation (sightings and fresh scats) by *P. x. celeris*. It is hoped the population can be recaptured at regular, long-term intervals to examine population and genetic changes associated with reintroduction.

At Aroona Sanctuary, all radio collars were left on animals and all traps were left in position at the cessation of trapping in January 2001. Future monitoring of re-introduced *P. x. xanthopus* will remain with the Royal Zoological Society of South Australia. Radio signals continue to be checked daily by the Aroona Sanctuary caretaker to ensure the survival of collared animals. Trapping sessions will continue on a bi-annual basis, predominantly to monitor survival through telemetry. A commitment has been made by the RZSSA and NRG Flinders to continue feral animal control indefinitely. Plans are currently being made to expand the Automated Tracking System to ensure greater accuracy and to allow automatic downloading of data at Leigh Creek Area School through a modem connected to each data-logger. It has also been proposed to include the Aroona Sanctuary *P. x. xanthopus* colony in the annual helicopter surveys of the species undertaken by the South Australian Department of Environment and Heritage (A. Emmerich pers. comm.).

9.3 Research implications and recommendations

Results from the current and previous study (Lapidge 2000) indicate that captive-bred *P. xanthopus* adapt readily to the wild upon release, with most ecological and physiological parameters measured indicating that the health of animals improved in the wild. Most biological parameters addressed in the current study of *P. x. celeris*, such as fecundity, survival, core area, home range, haematology and biochemistry, changed post-release and became similar to wild counterparts or other *Petrogale* species within five (most parameters by first re-capture session) to 12 months (home range). Similar trends were also observed for re-introduced *P. x. xanthopus* (Lapidge 2000). Thus, it is concluded that

re-introduction of *P. xanthopus* is an appropriate and ethical conservation technique when conducted in accordance with procedures detailed in the current study.

The key to success of future *Petrogale* and other macropod re-introductions lies in effective control of exotic predators (Short *et al.* 1992; Morris 2000). Until *V. vulpes* and feral cats (*Felis catus*) are eradicated from Australia all current and future reintroductions will remain dependent on continuous feral animal control. Thus, reintroduction should only be used as a conservation technique when sufficient long-term project management and funding are assured.

A recurring, and often justified, criticism of re-introduction programs is their expense (MacKinnon and MacKinnon 1991; Lindburg 1992; Sunquist 1993), even though long-term funding has been found to significantly correlate with success (Beck *et al.* 1994). The cost of the current study was approximately AUS\$175,000. Although a considerable outlay, the project was undertaken at a significantly lower cost than those of previously published reintroductions (Chapter 4; Fischer and Lindenmayer 2000). The principal benefits of the current study are the information that has been gained on adaptation of captive-bred *P. xanthopus* to the wild upon re-introduction, the new ecological and physiological information gathered on *P. xanthopus*, the protection of habitat for other species at Lambert Station and Aroona Sanctuary, and community education through publications, seminars and volunteer training.

The major expense in the current study, besides salary (\$59,000 over 4 years), was transport (\$43,600). In many ways this was unavoidable in monitoring two re-introduction projects in two different states and residing in a third. Re-introductions should occur into optimal habitat for the species concerned, wherever that may be, and thus to reduce costs biologists need to reside at the release site. The current project also conducted numerous expensive studies that may otherwise not be essential in future reintroductions. For example, water turnover and field metabolic rate analysis (\$13,000 min.) and blood chemistry analysis (\$2,800). Both these studies had not previously been undertaken on re-introduced captive-bred animals (or have not been reported) and were therefore considered essential for this project. The findings, that water turnover, field metabolic and food intake rates, biochemistry, and haematology adapted to the wild and became similar to wild-born counterparts, are encouraging. Unless such studies form the

objectives of future reintroductions, they could be replaced by condition scoring indices. The condition score index used in the current study was significantly dependent on plasma vitamin E concentration, vegetation abundance, rainfall, and total body water (depending on release site) and therefore highly indicative of general health. However, the applicability of the index for non-macropodid marsupials and eutherian mammals is uncertain and more appropriate indices may already exist for specific species. Thus, the technique should only be adopted with caution and will need refining to suit the target species. In my opinion, the current poor success rate of re-introductions, and the finding that success has not increased in the last two decades (Fischer and Lindenmayer 2000), indicates that little is actually known about effectively re-establishing a species in the wild. Consequently, every effort should be made by re-introduction practitioners to intricately examine and publish all biological aspects of returning animals to the wild. Accordingly, expense should remain high until more is known (although non-detrimental cost-savings can no doubt occur for some reintroductions; Chapter 4) and success rate has increased, and projects with insufficient resources should not occur.

Community participation in the re-introductions of *P. x. celeris* and *P. x. xanthopus* has provided a vital contribution to their current success. In both Queensland and South Australia pastoralists and volunteers have undertaken the majority of feral animal control, and continue to do so. Appendix II demonstrates the level of cooperation that has occurred in Queensland between the author and S. Henshall (field macropod harvester for the region), in both eradicating exotic predators from the release sites as well as detailing their diet. The formation of the Aroona Catchment Biodiversity Enhancement Project in South Australia by Leigh Creek pastoralists, mine workers and residents would have been unlikely to occur without the re-introduction project acting as a catalyst. However, environmental works completed or underway in the Aroona catchment far surpasses what the reintroduction project had ever hoped to achieve. This strong level of community support for both projects will hopefully help to ensure their future success.

No other known reintroductions in Australia have been monitored using such a wide range of ecological and physiological measurements of adaptation of captive-bred animals to the wild. The applicability of findings from the current study to other species, particularly within the genus *Petrogale*, is unknown. However, as the biology of the genus is predominantly similar among species (Short 1980; Lim 1987; Green 1989; Horsup

1994; Delaney 1997), findings should be useful in providing a basis on which to plan re-introduction programs for other *Petrogale* species. In addition, current findings are similar to those reported from re-introduced captive-bred *O. leucoryx* (Stanley Price 1986, 1989), *L. rosalia* (Kleiman 1989; Kleiman *et al.* 1986; Beck *et al.* 1991), *C. rufus* (Phillips 1990), *E. przewalskii* (Pereladova *et al.* 1999), *O. canadensis* (Ostermann *et al.* 2001) and *M. lagotis* (Southgate and Possingham 1995; Southgate *et al.* 2000), and re-introduced wild-born *B. lesueur* (Short and Turner 2000). Wider applications of current findings to include re-introduced captive-bred and wild-born eutherian mammals can therefore be anticipated.

The current reintroductions of *P. xanthopus* form a small part of the conservation of the species in the wild. The management practice of large-scale *V. vulpes* baiting in South Australia is contributing greatly to the survival of wild *P. x. xanthopus* colonies in some areas (Alexander and Copley 1999). The current study has demonstrated that re-introduction can be an effective and ethical means of re-establishing colonies of *P. xanthopus*, and potentially other *Petrogale* species, in the short-term. However, longer-term monitoring will be required to follow the ultimate fates of the colonies.

Implications from this project, that captive-bred *P. x. celeris* may in fact be healthier in the wild, are in contrast to the previously held view of “diminished health and welfare of reintroduced captive-bred animals, compared to their previously sheltered lives” (B. Beck, *L. rosalia* reintroduction, pers. comm.). While this is encouraging for re-introduction projects of captive-bred animals, it should not detract from captive breeding programs but rather provide evidence that appropriate captive breeding may not diminish the ability of an animal to survive in the wild as previously reported (Ralls *et al.* 1988; Kleiman 1989; Stanley Price 1989; Leberg 1990; Stanley Price 1991; Frankham 1994; Kleiman *et al.* 1994; Frankham 1995; Snyder *et al.* 1996; Lacy 1997; Miller *et al.* 1999). Zoological institutions, and their associated captive breeding programs and animal exhibition, provide an essential role in educating the public to the plight of endangered species and soliciting support for their conservation. Without such publicity, there may not be the financial backing required to undertake large-scale re-introduction programs as highlighted throughout this thesis.

Re-introduction is now a commonly used conservation technique in Australia, with 59 separate releases occurring in Western Australia alone since 1971 (Morris 2000).

However, the practice is less successful on marsupials than eutherian mammals and birds (Fischer and Lindenmayer 2000). Furthermore, re-introduction success was found not to have increased in the last two decades (Fischer and Lindenmayer 2000). Thus, re-introduction needs refining and a great deal more research before the technique can be relied on to curb the present rate of animal extinctions, particularly in Australia. Hopefully this study has taken a small step in the right direction.

REFERENCES

- Alexander, P. and Copley, P. (1999). Threatened Species: Yellow-footed Rock-wallaby. (http://www.DEH.sa.gov.au/biodiversity/yellow_foot.html). Department of Environment and Heritage, Adelaide.
- Allen, C. (1990). The diet and habitat requirements of the Yellow-footed Rock-wallaby *Petrogale xanthopus (celeris)* in southwestern Queensland. In *Proceedings of the Yellow-footed Rock-wallaby Management Workshop*. Copley, P. (Ed.). Wildlife Survey and Research Section. National Parks and Wildlife Service: Norwood, SA.
- Allen, C.B. (2001). Analysis of dietary competition between three sympatric herbivores in semi-arid west Queensland. Ph.D. Thesis, University of Sydney, Australia.
- Anderegg, R., Frey, H. and Muller, H.U. (1983). Reintroduction of the Bearded vulture or Lammergeier, *Gypaetus barbatus aureus*, to the Alps. *International Zoo Yearbook* **23**: 35-41.
- Arkell, B. (1999). *Flinders Ranges Bounceback*. Department of Environment, Heritage and Aboriginal Affairs, South Australia.
- Armstrong, D.P., Lovegrove, T.G., Allan, D.G. and Craig J.L. (1994). Composition of founder groups for bird translocations: does familiarity matter?. In *Reintroduction biology of Australian and New Zealand Fauna*: 105-11. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Armstrong, D.P., Soderquist, T. and Southgate R. (1994). Designing experimental reintroductions as experiments. In *Reintroduction biology of Australian and New Zealand Fauna*: 27-29. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Bach, C. (Ed) (1998). *Birth data determination in Australian Marsupials* Australia Regional Association of Zoological Parks and Aquaria, Mosman, Australia. Pp 64, 137.
- Bakker, H.R. and Main, A.R. (1980). Condition, body composition and total body water estimation in the quokka, *Setonix brachyurus* (Macropodidae). *Australian Journal of Zoology* **28**: 395-406.
- Bakker, H.R. and Bradshaw, S.D. (1983). Renal function in the Spectacled Hare-wallaby, *Lagorchestes conspicillatus* Effects of dehydration and protein deficiency. *Australian Journal of Zoology* **31**: 101-108.
- Batcheloe, T.A. (1980). The social organization of the brush-tailed rock-wallaby (*Petrogale penicillata penicillata*) on Motutapu Island. M.Sc. Thesis, Department of Zoology, Auckland University, New Zealand.
- Bangs, E.E. and Fritts, S.H. (1996). Reintroducing the gray wolf to central Idaho and Yellowstone National Park. *Wildlife Society Bulletin* **24**: 402-413.
- Banks, P.B., Newsome, A.E. and Dickman, C.R. (2000). Predation by red foxes limits recruitment in populations of eastern grey kangaroos. *Austral Ecology* **25**: 283-291.

- Barlow S. (1999) Reintroduction of the Yellow-footed Rock-wallaby (*Petrogale xanthopus xanthopus*) to Aroona Sanctuary, northern Flinders Ranges, South Australia. *Thylacinus* **23**: 12-16.
- Barnett, J.L., How, R.A. and Humphreys, W.F. (1979a). Blood parameters in natural populations of *Trichosurus* species (Marsupialia: Phalangeridae). I. Age, sex and seasonal variation in *T. caninus* and *T. vulpecula*. *Australian Journal of Zoology* **27**: 913-926.
- Barnett, J.L., How, R.A. and Humphreys, W.F. (1979b). Blood parameters in natural populations of *Trichosurus* species (Marsupialia: Phalangeridae). II. Influence of habitat and population strategies of *T. caninus* and *T. vulpecula*. *Australian Journal of Zoology* **27**: 927-938.
- Beck, B.B., Kleiman, D.G., Dietz, J.M., Castro, I., Carvalho, C., Martins, A. and Rettberg-Beck, B. (1991). Losses and reproduction in reintroduced golden-lion tamarins *Leontopithecus rosalia*. *Dodo, Journal of the Jersey Wildlife Preservation Trust* **27**: 50-61.
- Beck, B.B., Rapaport, L.G., Stanley Price M.R. and Wilson A.C. (1994). Reintroduction of captive-born animals. In *Creative conservation: Interactive management of wild and captive animals*: 265-286. Onley, P.J.S., Mace, G.M. and Feistner, A.T.C. (Eds). London: Chapman and Hall.
- Begon, M., Harper, J.L. and Townsend, C.R. (1990). *Ecology: Individuals, populations and communities* 2nd Ed. Blackwell Scientific Publications, USA.
- Biggins, D.E., Godbey, J.L., Hanebury, L.R., Luce, B., Marinari, P.E., Matchett, M.R., and Vargas, A. (1998). The effect of rearing methods on survival of reintroduced Black-footed Ferrets. *Journal of Wildlife Management* **62**: 643-653.
- Blaney, C.E., Dawson, T.J., McCarron, H.C.K., Buffenstein, R. and Krockenberger, A.K. (2000). Water metabolism and renal function and structure in Eastern Grey Kangaroos (*Macropus giganteus*): responses to water deprivation. *Australian Journal of Zoology* **48**: 335-345.
- Bolliger, A. and Backhouse, T.C. (1960). The blood of the koala, *Phascolarctos cinereus*. *Australian Journal of Zoology* **8**: 363-370.
- Booth, W. (1988). Reintroducing a political animal. *Science* **241**: 156-158.
- Borner, M. (1985). The rehabilitated chimpanzees of Rubondo Island. *Oryx* **19**: 151-154.
- Box, H.O. (1991). Training for life after release: simian primates as examples. In *Beyond captive breeding: re-introducing endangered mammals to the wild*: 111-123. Gipps, J. H. W. (Ed.). Oxford, Clarendon Press.
- Bradley, A.J. (1990). Seasonal effects on the haematology and blood chemistry in the red-tailed phascogale, *Phascogale calura* (Marsupialia: Dasyuridea). *Australian Journal of Zoology* **37**: 533-543.

- Brandle, R. (1998). *A biological survey of the north west Flinders Ranges, South Australia*. Biological Survey and Research, Department for Environment, Heritage and Aboriginal Affairs, South Australia.
- Breitenmoser, U. (1998). Large predators in the Alps: The fall and rise of man's competitors. *Biological Conservation* **83**: 279-289.
- Brower, J.E., Zar, J.H. and von Ende, C.N. (1990) *Field and Laboratory Methods for General Ecology*, 3rd Ed. Brown Publishers, USA.
- Bruce, R. (1902). *Reminiscences of an Old Squatter*. W. K. Thomas, Adelaide.
- Bruford, M.W., Cheesman, D.J., Coote, T., Green, H.A.A., Haines, S.A., O'Ryan, C., Williams, T. (1996). Microsatellites and their application to conservation genetics. In *Molecular genetic approaches in conservation*. 278-297. Smith, T.B., Wayne, R.K, (Eds). Oxford University Press, New York.
- Bureau of Meteorology (2001). Climate averages: tables of averages for specific locations. (<http://www.bom.gov.au/climate/averages>). Bureau of Meteorology, Australia.
- Burgman, M., Ferson, S. and Lindenmayer, D. (1994). The effect of the initial age-class distribution on extinction risks: implications for the reintroduction of Leadbeater's Possum. In *Reintroduction biology of Australian and New Zealand Fauna*: 15-19. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Burgman, M.A. and Lindenmayer, D.B. (1998). *Conservation biology of the Australian environment*. Surrey Beatty and Sons, Chipping Norton. 380pp.
- Carlson, G.P. (1997). Fluid, electrolyte, and acid-base balance. In *Clinical Biochemistry of domestic animals, 5th Ed* pp 485-516.
- Catt, D.C. (1981). Growth and condition of Bennetts wallaby (*Macropus rufogriseus fruticis*) in South Canterbury, New Zealand. *New Zealand Journal of Zoology* **8**: 295-300.
- Craig, J.L. (1994). Meta-populations: is management as flexible as nature? In *Creative conservation: Interactive management of wild and captive animals*. 50-66. Onley, P.J.S., Mace, G.M. and Feistner, A.T.C. (Eds). London: Chapman and Hall.
- Chivers, D.J. (1991). Guidelines for re-introductions: procedures and problems. In *Beyond captive breeding: re-introducing endangered mammals to the wild*: 89-99. Gipps, J. H. W. (Ed.). Oxford, Clarendon Press.
- Christensen, P. and Burrows, N. (1994). Project desert dreaming: experimental reintroduction of mammals to the Gibson Desert, Western Australia. In *Reintroduction biology of Australian and New Zealand Fauna*: 27-29. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Clancy, T.F. and Close, R.L. (1997). Queensland rock-wallabies- An overview of their conservation status, threats and management. *Australian Mammalogy* **19**: 169- 174.

- Clark, A.B. (1978). Sex ratio and local resources competition in a prosimian primate. *Science* **201**: 163-165.
- Close, R.L. and Bell, J.N. (1990). Age estimation of pouch young of the Allied Rock-wallaby (*Petrogale assimilis*) in captivity. *Australian Wildlife Research* **17**: 359-367.
- Close, R.L. (1997). Foreword: Orparinna in perspective. *Australian Mammalogy* **19**: 110-112.
- Cockburn, A. (1990). Sex ratio variation in marsupials. *Australian Journal of Zoology* **37**: 467-479.
- Cole, J.R., Langford, D.G. and Gibson, D.F. (1994). Capture myopathy in *Lagorchestes hirsutus* (Marsupialia: Macropodidae). *Australian Mammalogy* **17**: 137-138.
- Conaghty S. and Schultz D. (1998) Veterinary investigation of the yellow-footed rock-wallaby (*Petrogale xanthopus xanthopus*) for reintroduction. *American Association of Zoo Veterinarians Conference Proceedings, Nebraska*: 294-300.
- Cohn, J.P. (1993). The flight of the Californian Condor. *BioScience* **43**: 206-209.
- Copley, P.B. (1983). Studies on the yellow-footed rock-wallaby, *Petrogale xanthopus* Gray (Marsupialia: Macropodidae) I. Distribution in South Australia. *Australian Wildlife Research* **10**: 47-63.
- Copley, P.B. (Ed) (1990). *Proceedings of the Yellow-footed Rock-wallaby management workshop, Pt Augusta*. Wildlife Survey and Research Section. National Parks and Wildlife Service: Norwood, SA.
- Copley, P.B. (1994). Translocations of native vertebrates in South Australia: a review. In *Reintroduction biology of Australian and New Zealand Fauna*: 35-42. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Copley, P.B. and Robinson, A.C. (1983). Studies on the yellow-footed rock-wallaby, *Petrogale xanthopus* Gray (Marsupialia: Macropodidae) II. Diet. *Australian Wildlife Research* **10**: 63-76.
- Copley, P.B. and Alexander, P.J. (1997). Overview of the status of rock-wallabies in South Australia. *Australian Mammalogy* **19**: 153-162.
- Corbett, D. (1980). *A field guide to the Flinders Ranges*. Rigby, Adelaide.
- Croft, D.B. (1985). Inter- and intraspecific conflict between arid-zone kangaroos at watering points. *Australian Wildlife Research* **12**: 337-348.
- Cunningham, A.A. (1996). Disease risks of wildlife translocations. *Conservation Biology* **10**: 349-353.
- Dawson, T.J. and Denny, M.J.S. (1969). Seasonal variation in the plasma and urine electrolyte concentration of the arid zone kangaroos *Megaleia rufa* and *Macropus robustus*. *Australian Journal of Zoology* **17**: 777-784.

- Dawson, T.J., Denny, M.J.S., Russell, E.M. and Ellis, B.A. (1975). Water usage and diet preferences of free ranging kangaroos, sheep and feral goats in the Australian arid zone during summer. *Journal of Zoology (London)* **177**: 1-23.
- Dawson, T.J. and Ellis, B.A. (1979). Comparison of the diets of yellow-footed rock-wallabies and sympatric herbivores in western New South Wales. *Australian Wildlife Research* **6**: 245-54.
- Delaney, R. (1997). Population dynamics of the Allied Rock-wallaby *P. assimilis*. Implications for conservation. *Australian Mammalogy* **19**: 199-207.
- Delaney, R.M. and De'ath, G. (1990). Age estimation and growth rates of captive and wild pouch young of *Petrogale assimilis*. *Australian Wildlife Research* **17**: 491-499.
- Dellow, D.W. and Hume, I.D. (1982). Studies on the nutrition of macropodine marsupials. IV. Digestion in the stomach and the intestine of *Macropus giganteus*, *Thylogale thetis* and *Macropus eugenii*. *Australian Journal of Zoology* **30**: 767-777.
- Delroy, L.B., Earl, J., Radbone, A.C., Robinson, A.C. and Hewett, M. (1986). The breeding and re-establishment of the brush-tailed bettong, *Bettongia penicillata*, in South Australia. *Australian Wildlife Research* **13**: 387-396.
- Dempster, J.P. and Hall, M.L. (1980). An attempt ay re-establishing the swallowtail butterfly at Wicken Fen. *Ecological Entomology* **5**: 327-24.
- Denny M.J.S. and Dawson, T.J. (1975). Comparative metabolism of tritiated water by macropodid marsupials. *American Journal of Physiology* **228**: 1794-1799.
- Dietz, J.M., Dietz, L.A. and Nagagata, E.Y. (1994). The effective use of flagship species for conservation of biodiversity: the example of lion tamarins in Brazil. In *Creative conservation: Interactive management of wild and captive animals*: 32-49. Onley, P.J.S., Mace, G.M. and Feistner, A.T.C. (Eds). London: Chapman and Hall.
- Dovey, L. Wong, V. and Bayne, P. (1997). An overview of the status and management of rock-wallabies (*Petrogale*) in New South Wales. *Australian Mammalogy* **19**: 163-168.
- Duffey, E. (1977). The re-establishment of the large copper butterfly, *Lycena dispar batava* Obth., on Woodwalton Fen National Nature Reserve, Cambridgeshire, England, 1969-73. *Biological Conservation* **12**: 143-58.
- Dufty, A.C., Seebeack, J.H., McKay, J. and Watson, A.J. (1994). Reintroduction of the Eastern Barred Bandicoot *Perameles gunnii* at Gellibrand Hill Park, Victoria. In *Reintroduction biology of Australian and New Zealand Fauna*: 219-225. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Ealey, E.H.M. (1967). Ecology of the Euro, *Macropus robustus* (Gould), in north-west Australia. II. Behaviour, movements and drinking patterns. *CSIRO Wildlife Research* **12**: 27-51.

- Ealey, E.H.M. and Mains, A.P. (1967). Ecology of the euro, *Macropus robustus* (Gould), in north-western Australia. III. Age and growth. *CSIRO Wildlife Research* **12**: 67-80.
- Eldridge, M.D.B. (1997a). Taxonomy of rock-wallabies, *Petrogale* II. An historical review. *Australian Mammalogy* **19**: 113-122.
- Eldridge, M.D.B. (1997b). Rock-wallaby conservation: Essential data and management priorities. Proceedings of the 1994 National Rock-wallaby Symposium Workshops. *Australian Mammalogy* **19**: 325-330.
- Eldridge, M.D.B. (1997c). Restriction analysis of mitochondrial DNA from the Yellow-footed Rock-wallaby, *Petrogale xanthopus*. Implications for management. *Wildlife Research* **24**: 289-294.
- Eldridge, M.D.B. and Close, R.L. (1997). Chromosomes and evolution in rock-wallabies, *Petrogale*. *Australian Mammalogy* **19**: 123-135.
- Eldridge, M.D.B. and Kinnear, J.E. (1999). How far can a wallaby hop? Gene flow and dispersal amongst rock-wallaby populations in the West Australian wheatbelt. *Australian Mammal Society Conference Proceedings*: 40. Sydney, NSW.
- Ellis, B.A., Russell, E.M., Dawson, T.J. and Harrop, C.J.F. (1977). Seasonal changes in diet preferences of free-ranging red kangaroos, euros and sheep in western New South Wales. *Australian Wildlife Research* **4**: 127-44.
- Ellis, W.A.H., White, N.A., Kunst, N.D. and Carrick, F.N. (1990). Response of Koalas (*Pascualarctos cinereus*) to re-introduction to the wild after rehabilitation. *Australia Wildlife Research* **17**: 421-426.
- Estes, J.A. (1998). Concerns about rehabilitation of oiled wildlife. *Conservation Biology* **12**: 1156-1157.
- Evans, M. (1996). Home range and movement schedules of sympatric Bridled Nailtail and Black-striped Wallabies. *Wildlife Research* **23**: 547-556.
- Evans, M. (2000). 'Tracker v.1' (radio-telemetry analysis program). University of New England, Australia.
- Eyre, E.J. (1841). *Journals of expeditions of discovery into Central Australia and overland from Adelaide to King George's Sound in the years 1840-41*. Boone, London.
- Feldman, B.F., Zinkl, J.G. and Jain, N.C. (2000). *Schalm's Veterinary Hematology, 5th Ed* Lippincott, Williams and Wilkins, USA.
- Ferris, J. and Macdonald, C. (1995) *Yellow-footed Rock-wallaby Reintroduction Project Plan*. Royal Zoological Society of South Australia Publication.
- Finco D.R. (1997). Kidney function. In *Clinical biochemistry of domestic animals, 5th Ed*: 441-484. Kaneko, J.J., Harvey, J.W. and Bruss, M.L. (Eds.). Academic Press, San Diego.

- Fischer, J. and Linder Mayer, D.B. (2000). An assessment of the published results of animal relocations. *Biological Conservation* **96**: 1-11.
- Fisher, D.O. (1999). Offspring sex ratio variation in the Bridled Naitail Wallaby, *Onychogalea fraenata*. *Behavioural Ecology and Sociobiology* **45**: 411-419.
- Fitzsimmons, N.N., Buskirk, S.W. and Smith, M.W. (1997). Genetic changes in reintroduced rocky mountain bighorn sheep populations. *Journal of Wildlife Management* **61**: 863-872.
- Flannery, T.F. (1994). *The Future Eaters*. Reed, Melbourne.
- Food and Agricultural Organization of the United Nations (1986). The Przewalski horse and restoration to its natural habitat in Mongolia. Animal production and health paper 61. FAO, Rome.
- Forbes, S.H. and Boyd, D.K. (1997). Genetic structure and migration in native and reintroduced Rocky Mountain wolf populations. *Conservation Biology* **11**: 1226-1234.
- Fox, A. (1966). Going... Going... Go... *Wildlife Service* **3** (7), 6.
- Frankham, R. (1994). Genetic management of captive populations for reintroduction. In *Reintroduction biology of Australian and New Zealand Fauna*: 31-34. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Frankham, R. (1995). Conservation genetics. *Annual Review Genetics* **29**: 305-337.
- Fritts, S.H., Paul, W.J. and Mech, L.D. (1984). Movements of translocated wolves in Minnesota. *Journal of Wildlife Management* **48**: 709-721.
- Gaughwin, M.D., Judson, G.J., Macfarlane, W.V. and Siebert, B.D. (1984). Effect of drought on the health of wild Hairy-nosed Wombats, *Lasiorninus latifrons*. *Australian Wildlife Research* **11**: 455-463.
- Gibson, D.F., Johnson, K.A., Langford, D.G., Cole, J.R., Clarke, D.E. and Willowra Community (1994). The Rufous Hare-wallaby *Lagorchestes hirsutus*: a history of experimental reintroduction in the Tanami Desert, Northern Territory. In *Reintroduction biology of Australian and New Zealand Fauna*: 171-176. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Gibson, L.A. (1999). Nutritional ecology of the Greater Bilby, *Macrotis lagotis*, in far south-western Queensland. PhD Thesis, University of Sydney, Australia.
- Gibson, L.A. and Hume, I.D. (2000). Seasonal field energetics and water influx rates of the Greater Bilby (*Macrotis lagotis*). *Australian Journal of Zoology* **48**: 225-239.
- Ginsberg, J.R. (1994). Captive breeding, reintroduction, and the conservation of canids. In *Creative conservation: Interactive management of wild and captive animals*: 365-383. Onley, P.J.S., Mace, G.M. and Feistner, A.T.C. (Eds). London: Chapman and Hall.

- Gordon, G., McGreevey, D.G., and Lawrie, B.C. (1978). The yellow-footed rock-wallaby, *Petrogale xanthopus* Gray (Macropodidae), in Queensland. *Australian Wildlife Research* **5**: 295-297.
- Gordon, G., McRae, P., Lim, L., Reimer, D., and Porter, G. (1993). The conservation status of the yellow-footed rock-wallaby in Queensland. *Oryx* **27**: 159-168.
- Gorman, M. (1999). Oryx go back to the brink. *Science* **398**: 190.
- Gray, J.E. (1854). Description of a new species of *Petrogale*. *Proc. Zool. Soc. Lond.* **22**, 249.
- Green, B. (1989). Water and energy turnover in free-living macropodoids. In *Kangaroos, Wallabies and Rat Kangaroos* 223-229. Grigg, G., Hume, I. and Jarman, P. (Eds.). Surrey Beatty and Sons, New South Wales, Australia.
- Green, B. (1997). Field energetics and water fluxes in marsupials. In *Marsupial Biology: Recent research, new perspectives*: pp 143-162. Saunders, N.R. and Hinds, L.A. (Eds.). University of New South Wales Press: Sydney.
- Green, J., Green, R. and Jefferies, D.J. (1984). A radio-tracking survey of otters, *Lutra lutra* on a Perthshire river system. *Lutra* **27**: 85-145.
- Griffin, A.S., Blumstein, D.T. and Evans, C.S. (2000). Training captive-bred or translocated animals to avoid predators. *Conservation Biology* **14**: 1317-1326.
- Griffith, B., Scott, J. M., Carpenter, J. W. and Reed, C. (1989). Translocation as a species conservation tool: status and strategy. *Science* **245**: 477-480.
- Harrop, C.J.F. and Barker, S. (1972). Blood chemistry and gastro-intestinal changes in the developing red kangaroo (*Megaleia rufa*, Desmarest). *Australian Journal of Experimental Biology and Medical Science* **50**: 245-249.
- Haig, S.M., Ballou, J.D. and Derrickson, S.R. (1990). Management options for preserving genetic diversity: reintroduction of Guam rails to the wild. *Conservation Biology* **4**: 290-300.
- Hayne, D.W. (1949). Calculation of size of home range. *Journal of Mammalogy* **30**: 1-18.
- Henzell, R. (1990). Feral goats and yellow-footed rock-wallabies. In *Proceedings of the Yellow-footed Rock-wallaby Management Workshop*. Copley, P. (ed.). Wildlife Survey and Research Section. National Parks and Wildlife Service: Norwood, SA.
- Hickling G.J. (1991). The ecology of brush-tailed possum populations infested with tuberculosis. In *Proceedings of the Symposium on Tuberculosis. Publication 132* (Ed. R. Jackson). Massey University, Palmerston North, New Zealand.
- Holleman, D.F and Dieterich, R.A. (1973). Body water content and turnover in several species of rodents as evaluated by the tritiated water method. *Journal of Mammalogy* **54**: 456-465.

- Hornsby, P.E. (1978). A note on the pouch life of rock wallabies. *Victorian Naturalist* **95**: 108-111.
- Hornsby, P.E. (1980). A history of rock-wallabies, *Petrogale* spp., in the Adelaide Zoo. *International Zoo Yearbook* **20**: 154-160.
- Hornsby, P. (1997). Possible causes of mortality in the yellow-footed rock-wallaby, *Petrogale xanthopus* Gray (Marsupialia: Macropodidae). *Australian Mammalogy* **19**: 245-248.
- Horsup, A. (1994). Home ranges of the Allies Rock-wallaby, *Petrogale assimilis*. *Wildlife Research* **21**: 65-84.
- Hume, I.D. (1999). *Marsupial nutrition*. Cambridge University Press, UK.
- Inns, R.W. (1982). Age determination in the Kangaroo Island Wallaby, *Macropus eugenii* (Desmarest). *Australian Wildlife Research* **9**: 213-220.
- IUCN (1987). *The IUCN position statement on translocation of living organisms: introductions, re-introductions and re-stocking*. Gland: IUCN.
- IUCN (1998). *Guidelines for Re-introductions*. Prepared by the IUCN/SSC Re-introduction Specialist Group. IUCN, Gland, Switzerland and Cambridge, U.K. 10pp
- IUCN (2000). *Red List of Threatened Species* (<http://www.redlist.org>). International Union for Conservation of Nature and Natural Resources, Gland Switzerland.
- Jarman, P.J. (1989). Sexual dimorphism in macropodoidea. In *Kangaroos, Wallabies and Rat Kangaroos*: 433-447. Grigg, G., Hume, I. and Jarman, P. (Eds.). Surrey Beatty and Sons, New South Wales, Australia.
- Jarman, P.J. and Bayne, P. (1997). Behavioural ecology of *Petrogale penicillata* in relation to conservation. *Australian Mammalogy* **19**: 219-228.
- Jarman, P.J. and Capararo, S.M. (1997). Use of rock-wallaby faecal pellets for detecting and monitoring populations and examining habitat use. *Australian Mammalogy* **19**: 257-264.
- Johnson, C.N. (1989). Dispersal and philopatry in the macropodoids. In *Kangaroos, Wallabies and Rat Kangaroos* 593-601. Grigg, G., Hume, I. and Jarman, P. (Eds.). Surrey Beatty and Sons, New South Wales, Australia.
- Johnson, K.A., Burbidge, A.A. and McKenzie, N.L. (1989). Australian Macropodoidea: Status, causes of decline and future research and management. In *Kangaroos, Wallabies and Rat Kangaroos* 641-657. Grigg, G., Hume, I. and Jarman, P. (Eds.). Surrey Beatty and Sons, New South Wales, Australia.
- Kakulas, B.A. (1961). Myopathy affecting the Rottneest Quokka (*Setonix brachyurus*) reversed by α -Tocopherol. *Nature* **191**: 402-3.

- Kakulas, B.A. (1963a). Trace quantities of selenium ineffective in the prevention of nutritional myopathy in the Rottnest Quokka (*Setonix brachyurus*). *Australian Journal of Science* **25**: 313-14.
- Kakulas, B.A. (1963b). Influence of the size of the enclosure on the development of myopathy in the captive Rottnest Quokka. *Nature* **198**: 673-4.
- Kaneko, J.J. (1997). Serum proteins and dysproteinemias. In *Clinical biochemistry of domestic animals*, 5th Ed: 117-138. Kaneko, J.J., Harvey, J.W. and Bruss, M.L. (Eds.). Academic Press, San Diego.
- Kaneko, J.J., Harvey, J.W. and Bruss, M.L. (1997). *Clinical biochemistry of domestic animals*, 5th Academic Press, San Diego.
- Kaufmann, J.H. (1974). Social ethology of the whiptail wallaby, *Macropus parryi*, in north-eastern New South Wales, Australia. *Animal Behaviour* **22**: 281-369.
- Kennedy, M. (1992). *Australasian marsupials and monotremes: An action plan for their conservation*. IUCN/SSC Australasian Marsupial and Monotreme Group. IUCN, Gland, Switzerland.
- Kenward, R.E. and Hodder, K.H. (1996). *RANGES V: An analysis system for biological location data*. Institute of Terrestrial Ecology, Furzebrook Research Station, Dorset, U.K.
- Kinnear, J.E., Onus, M.L. and Bromilow, P.N. (1984). Foxes, feral cats and rock-wallabys. *Swans* **14**: 3-8.
- Kinnear, J.E., Onus, M.L. and Bromilow, P.N. (1988). Fox control and rock-wallaby population dynamics. *Australian Wildlife Research* **15**: 435-50.
- Kinnear, J.E., Onus, M.L. and Sumner, N.R. (1998). Fox control and rock-wallaby population dynamics- II. An update. *Wildlife Research* **25**: 81-88.
- Kleiman, D. G. (1989). Reintroductions of captive mammals for conservation: guidelines for reintroducing endangered species into the wild. *BioScience* **39**: 152-161.
- Kleiman, D.G., Beck, B.B., Dietz, J.M. and Dietz, L.A., Ballou, J.D. and Coimbra-Filho, A.F. (1986). Conservation program for the golden lion tamarin: captive research and management, ecological studies, educational strategies, and reintroduction. In *Primates; the road to self-sustaining populations*: 959-979. Benerschke, K. (Ed.). New York: Springer-Verlag.
- Kleiman, D.G., Beck, B.B., Dietz, J.M. and Dietz, L.A. (1991). Costs of a re-introduction and criteria for success: accounting and accountability in the golden lion tamarin conservation program. In *Beyond captive breeding: re-introducing endangered mammals to the wild*: 125-142. Gipps, J. H. W. (Ed.). Oxford, Clarendon Press.
- Kleiman, D.G., Stanley Price, M.R., Beck, B.B. (1994). Criteria for reintroductions. In *Creative conservation: Interactive management of wild and captive animals*: 287-303. Onley, P.J.S., Mace, G.M. and Feistner, A.T.C. (Eds). London: Chapman and Hall.

- Kloor, K. (1999). Lynx and biologists try to recover after disastrous start. *Science* **285**: 320.
- Konstant, W.R. and Mittermeier, R.A. (1982). Introduction, reintroduction and translocation of Neotropical primates: past experiences and future possibilities. *International Zoo Yearbook* **22**: 69-77.
- Krebs, C.J. (1978). *Ecology: The experimental analysis of distribution and abundance*, 2nd ed. Harpers and Row, New York.
- Lacy, R.C. (1993). VORTEX: a computer simulation model for population viability analysis. *Wildlife Research* **20**: 45-65.
- Lacy, R.C. (1997). Importance of genetic variation to the viability of mammalian populations. *Journal of Mammalogy* **78**: 320-335.
- Langford, D. and Burbidge, A. (2001). Translocations of mala from the Tanami Desert, Northern Territory to Trimouille Island, Western Australia. *Australian Mammalogy* in press.
- Lapidge, S.J. (1997). A dietary study on reintroduced yellow-footed rock-wallabies, *Petrogale xanthopus xanthopus* (Marsupialia: Macropodidae), in the northern Flinders Ranges, South Australia. B.Sc.(Hons) Thesis, Flinders University of South Australia.
- Lapidge, S.J. (2000). Dietary adaptation of reintroduced yellow-footed rock-wallabies, *Petrogale xanthopus xanthopus* (Marsupialia: Macropodidae), in the northern Flinders Ranges, South Australia. *Wildlife Research* **27**: 195-201.
- Lapidge, S.J. and Henshall, S. (2001). Diet of foxes and cats, with evidence of predation on yellow-footed rock-wallabies (*Petrogale xanthopus celeris*) by foxes, in southwestern Queensland. *Australian Mammalogy* **23**: 47-51.
- Leberg, P.L. (1990). Genetic considerations in the design of introduction programs. *Transactions North American Wildlife National Research Conference* **55**: 609-619.
- Leberg, P.L. (1992). Effects of population bottlenecks on genetic diversity as measured by allozyme electrophoresis. *Evolution* **46**: 477-494.
- Lenth, R.V. (1981). On finding the source of a signal. *Technometrics* **23**: 149-154.
- Le Souef, A.S. (1924). Notes on some rock wallabies, genus *Petrogale*, with descriptions of two new species. *Aust. Zool.* **3**: 272-6.
- Lewis, J.H., Phillips, L.L and Hann, C. (1968). Coagulation and hematological studies in primitive Australian mammals. *Comparative Biochemistry and Physiology* **25**: 1129-1135.
- Lifson, N. and McClintock, R. (1966). Theory of use of the turnover rates of body water for measuring energy and material balance. *Journal of Theoretical Biology* **12**: 46-74.

- Lim, L., Mahood, I. and Siepen, G. (1980). Yellow-footed rock-wallaby, *Petrogale xanthopus*. *Endangered animals of New South Wales*: 27-32. National Parks and Wildlife Service: Sydney.
- Lim, T.L. (1987). Ecology and management of the rare yellow-footed rock-wallaby *Petrogale xanthopus* Gray (Macropodidae). Ph.D. thesis, Macquarie University, NSW.
- Lim, T.L. and Giles, J.R. (1987). Studies on the yellow-footed rock-wallaby, *Petrogale xanthopus* Gray (Marsupialia: Macropodidae) III. Distribution and management in western New South Wales. *Australian Wildlife Research* **14**: 147-161.
- Lim, T.L., Robinson, A.C., Copley, P.B., Gordon, G., Canty, P.D. and Reimer, D. (1987). *The Conservation and Management of the Yellow-footed Rock-wallaby Petrogale xanthopus Gray, 1854*. Special Publication No. 4. SA National Parks and Wildlife Service and SA Department of Environment and Planning, Adelaide.
- Lim, L., Sheppard, N., Smith, P. and Smith J. (1992). The biology and management of the yellow-footed rock-wallaby (*Petrogale xanthopus*) in NSW. (Species management report No. 10), NSW National Parks and Wildlife Service: Hurstville, NSW.
- Lindenmayer, D.B. (1994). Some ecological considerations and computer-based approaches for the identification of potentially suitable release sites for reintroduction programs. In *Reintroduction biology of Australian and New Zealand Fauna*: 1-5. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Lindburg, D.G. (1992). Are wildlife reintroductions worth the cost? *Zoo Biology* **11**: 1-2.
- MacKinnon, K and MacKinnon, J. (1991). Habitat protection and reintroduction programs. In *Beyond captive breeding: re-introducing endangered mammals to the wild*: 173-198. Gipps, J. H. W. (Ed.). Oxford, Clarendon Press.
- Maxwell, S., Burbidge, A.A. and Morris, K.D. (1996). *The 1996 action plan for Australian marsupials and monotremes*. Wildlife Australia, Canberra.
- May, R.M. (1991). The role of ecological theory in planning re-introduction of endangered species. In *Beyond captive breeding: re-introducing endangered mammals to the wild*: 145-163. Gipps, J. H. W. (Ed.). Oxford, Clarendon Press.
- Maynes, G.M. (1989). Zoogeography of the macropodoidea. In *Kangaroos, Wallabies and Rat Kangaroos*: 47-66. Grigg, G., Hume, I. and Jarman, P. (Eds.). Surrey Beatty and Sons, New South Wales, Australia.
- Margan, S.H., Nurthen, R.K., Montgomery, M.E., Woodworth, L.M., Lowe, E.H., Briscoe, D.A. and Frankham, R. (1998). Single large or several small? Population fragmentation in the captive management of endangered species. *Zoo Biology* **17**: 467-480.
- McCallum, H. (1994). Modelling translocation strategies for the Bridled Naitail Wallaby *Onychogalea fraenata* Gould, 1840. In *Reintroduction biology of Australian and New Zealand Fauna*: 7-14. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.

- McCallum, H., Timmers, P. and Hoyle, S. (1995). Modelling the impact of predation on reintroductions of Bridled Naitail wallabies. *Wildlife Research* **22**: 163-171.
- McCarthy, M.A. (1994). Population viability analysis of the Helmeted Honeyeater: risk assessment of captive management and reintroduction. In *Reintroduction biology of Australian and New Zealand Fauna*: 7-14. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- McIlroy, J.C. (1986). The sensitivity of Australian animals to 1080 poison IX.*Comparisons between the major groups of animals, and the potential danger non-target species face from 1080-poisoning campaigns. *Australian Wildlife Research* **13**: 39-48.
- McLean, I.G., Lundie-Jenkins, G. and Jarman, P.J. (1994). Training captive Rufous Hare-wallabies to recognize predators. . In *Reintroduction biology of Australian and New Zealand Fauna*: 177-182. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- McLean, I.G., Lundie-Jenkins, G. and Jarman, P.J. (1996). Teaching an endangered mammal to recognise predators. *Biological Conservation* **75**: 51-62.
- McNab, J. (1983). Wildlife management as scientific experimentation. *Wildlife Society Bulletin* **11**: 397-401.
- Miller, B., Ralls, K., Reading, R. P., Scott, J. M. and Estes, J. (1999). Biological and technical considerations of carnivore translocations: a review. *Animal Conservation* **2**: 59-68.
- Miller, S.D. and Ballard, W.B. (1982). Homing of transplanted Alaskan brown bears. *Journal of Wildlife Management* **46**: 869-876.
- Micham, H. (1983). *The story of the Flinders Ranges*. Rigby, Australia.
- Moore, D.E. and Smith, R. (1991). The red wolf as a model for carnivore re-introductions. In *Beyond captive breeding: re-introducing endangered mammals to the wild*: 263-278. Gipps, J. H. W. (Ed.). Oxford, Clarendon Press.
- Morris, K. (2000). Fauna translocations in Western Australia 1971-1999: an overview. In *Biodiversity and the reintroduction of native fauna at Uluru-Kata Tjuta National Park*. Gillen, J.S., Hamilton, R., Low, W.A. and Creagh, C. (eds), Bureau of Rural Sciences, Australia.
- Moss, G.L. and Croft, D.B. (1999). Body condition of the red kangaroo (*Macropus rufus*) in arid Australia: The effect of environmental condition, sex and reproduction. *Australian Journal of Ecology* **24**: 97-109.
- Munday, B.L. (1972). Myonecrosis in free-living and recently captured macropods. *Journal of Wildlife Diseases* **8**: 191-192
- Munday, B.L. (1988). Marsupial diseases. In *Australian Wildlife, Proceedings No. 104*. Post Graduate Committee in Veterinary Science, University of Sydney: Sydney.

- Munks, S.A. (1990). Ecological energetics and reproduction in the Common Ringtail Possum, *Pseudocheirus peregrinus*. PhD Thesis, University of Tasmania, Hobart.
- Munks, S.A. and Green, B. (1995). Energy allocation for reproduction in a marsupial arboreal folivore, the common ringtail possum (*Pseudocheirus peregrinus*). *Oecologia* **101**: 94-104.
- Murphy, C.R. and Smith, R.S. (1970). Age determination of pouch young and juvenile Kangaroo Island wallabies. *Transactions of the Royal Society of South Australia* **94**: 15-20.
- Murphy, K.J. (1985). Some aspects of the ecology, digestive physiology and nutrition of the Yellow-footed Rock-wallaby, *P. xanthopus*. B. Nat. Res. Thesis, University of New England, New South Wales.
- Nagy, K.A. (1980). CO₂ production in animals: analysis of potential errors in the doubly labeled water method. *American Journal of Physiology* **238**: R466-473.
- Nagy, K.A. (1983). The doubly labeled water (³HH¹⁸O) method: a guide to its use. University of California, Los Angeles, Publication No. 12-1417.
- Nagy, K.A. (1987). Field metabolic rate and food requirement scaling in mammals and birds. *Ecological Monographs* **57**: 111-128.
- Nagy, K.A. and Peterson, C.C. (1988). Scaling of water flux rate in animals. University of California Publications in Zoology No. 120.
- Nagy, K.A. (1989). Doubly-labeled water studies of vertebrate physiological ecology. In *Stable Isotopes in Ecological Research*. pp 270-287. Rundel, P.W., Ehleringer, J.R. and Nagy, K.A. (Eds.). Springer-Verlag, New York.
- Nagy, K.A. (1992). The doubly-labeled water method in ecological energetics studies of terrestrial vertebrates. *Bulletin of Society of Ecophysicists, Supplement* **XVII**: 9-14.
- Nagy, K.A. (1994). Seasonal water, energy and food use by free-living, arid-habitat mammals. *Australian Journal of Zoology* **42**: 55-63.
- Nagy, K.A., Girand, I.A., and Brown, T.K. (1999). Energetics of free-ranging mammals, reptiles, and birds. *Annual Review of Nutrition* **19**: 247-277.
- Nagy, K.A. and Costa, D.P. (1980). Water flux in animals: analysis of potential errors in the tritiated water method. *American Journal of Physiology* **238**: R454-465.
- Nagy, K.A. and Martin, R.W. (1985). Field metabolic rate, water flux, food composition and time budget of Koalas, *Phascolarctos cinereus* (Marsupialia: Phascolarctidae) in Victoria. *Australian Journal of Zoology* **33**: 655-665.
- Nagy, K.A., Bradley, A.J. and Morris, K.D. (1990a). Field metabolic rates, water fluxes and feeding rates of Quokkas, *Setonix brachyurus*, and Tammars, *Macropus eugenii*, in Western Australia. *Australian Journal of Zoology* **37**: 553-560.

- Nagy, K.A., Sanson, G.D. and Jacobsen, N.K. (1990b). Comparative field energetics of two macropod marsupials and a ruminant. *Australian Wildlife Research* **17**: 591-199.
- Nedler, V.J. (1984). Vegetation survey of Queensland: south central Queensland. *Queensland Department of Primary Industries Botany Bulletin No.3*.
- Nei, M. (1978). Estimation of average heterozygosity and genetic distance from a small number of individuals. *Genetics* **89**: 583-590.
- Nichols, J.D. and Pollock, K.H. (1983). Estimation methodology in contemporary small mammal capture-recapture studies. *Journal of Mammalogy* **64**: 253-260.
- Nicholls, C. (1972). A study of the behaviour and social organization of the Yellow-footed Rock-wallaby (*Petrogale xanthopus*). BA (Hons) Thesis, Psychology Department, Adelaide University.
- Ostermann, S.D., Deforge, J.R. and Edge, W.D. (2001). Captive breeding and reintroduction evaluation criteria: a case study of Peninsular Bighorn Sheep. *Conservation Biology* **15**: 749-760.
- Parker, B. (2000). Diet of breeding wedge-tailed eagles (*Aquila audax*) near Charleville, southwest Queensland. Honours thesis, University of Sydney.
- Parsons, R.S., Heddle, R.W.L., Flux, W.G. and Guiler, E.R. (1970). Studies on the blood of the Tasmanian devil. *Comparative Biochemistry and Physiology* **32**: 345-351.
- Parsons, R.S., Atwood, J., Guiler, E.R. and Heddle, R.W.L. (1971a). Comparative studies on the blood of monotremes and marsupials. I. Haematology. *Comparative Biochemistry and Physiology* **39B**: 203-208.
- Parsons, R.S., Guiler, E.R. and Heddle, R.W.L. (1971b). Comparative studies on the blood of monotremes and marsupials. II. Electrolyte organic constituents, proteins, gas analysis and enzymes. *Comparative Biochemistry and Physiology* **39B**: 209-217.
- Perelodova, O.B., Sempere, A.J., Soldatova, N.V., Dutov, V.U., Fisenko, G. and Flint, V.E. (1999). Przewalshi's horse- adaptation to semi-wild life in desert conditions. *Oryx* **33**: 47-58.
- Phillips, M.K. (1990). The red wolf: Recovery of an endangered species. *Endangered Species Update* **8**: 79-81.
- Pierre, J.P. (1999). Reintroduction of the South Island saddleback (*Philesturnus carunculatus carunculatus*): dispersal, social organization and survival. *Biological Conservation* **89**: 153-159.
- Pietsch, R.S. (1994). The fate of urban Common Brushtail Possums translocated to sclerophyll forests. . In *Reintroduction biology of Australian and New Zealand Fauna*: 239-246. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Ponder, E. Yeager, J.F. and Charipper, H.A. (1928). Studies in comparative haematology. *Quarterly Journal of Experimental Physiology* **19**: 273-283.

- Poole, W.E., Merchant, J.C., Carpenter, S.M., Calaby, J.H. (1985). Reproduction, growth and age determination in the yellow-footed rock-wallaby *Petrogale xanthopus* Gray, in captivity. *Australian Wildlife Research* **12**: 127-136.
- Poole, W.E., Carpenter, S.M., Wood, J.T. (1982). Growth of Grey Kangaroos and the reliability of age determination from body measurements I. The Eastern Grey Kangaroo, *Macropus giganteus*. *Australian Wildlife Research* **9**: 9-20.
- Pope, L. C., Sharp, A. and Moritz, C. (1996). Population structure of the yellow-footed rock-wallaby *Petrogale xanthopus* (Gray, 1854) inferred from mtDNA sequences and microsatellite loci. *Molecular Ecology* **5**: 629-640.
- Possingham, H.P. and Davies, I. (1995). ALEX: a model for the viability of spatially structured populations. *Biological Conservation* **73**: 143-50.
- Perelodova, O.B., Sempéré, A.J., Soldatova, N.V., Dutov, V.U., Fisenko, G. and Flint, V.E. (1999). Przewalski's horses- adaptation to semi-wild life in desert conditions. *Oryx* **33**: 47-58.
- Presidente, P.J.A. and Correa, J. (1981). Haematology, plasma electrolytes and serum biochemical values of *Trichosurus vulpecula* (Marsupialia: Phalangeridae). *Australian Journal of Zoology* **29**: 507-517.
- Priddel, D. and Wheeler, R. (1994). Mortality of captive-raised Malleefowl, *Leipoa ocellata*, released into a mallee remnant within the wheat-belt of New South Wales. *Wildlife Research* **21**: 543-552.
- Queller, D.C. and Goodnight, K.F. (1989). Estimating relatedness using genetic markers. *Evolution* **43**: 258-275.
- Ralls, K., Ballou J. D. and Templeton, A. R. (1988). Estimates of lethal equivalents and the cost of inbreeding in mammals. *Conservation Biology* **2**: 185-193.
- Raymond, M. and Rousset, F. (1995). GENEPOP (Version 1.2): Population genetics software for exact tests and ecumenicism. *Journal of Heredity* **86**: 248-249.
- Reading, R.P., Clark, T.W., Seebeck, J.H. and Pearce, J. (1996). Habitat Suitability Index Model for the Eastern Barred Bandicoot, *Perameles gunnii*. *Wildlife Research* **23**: 221-35.
- Reuters (2000). Exxon Mobil Q1 profit doubles. *The Times of India*: <http://www.timesofindia.com/260400/26inte7.htm>.
- Reimer, A.B. and Hindell, M.A. (1996). Variation in body condition and diet of the Eastern Barred Bandicoot (*Perameles gunnii*) during the breeding season. *Australian Mammalogy* **19**: 47-52.
- Risbey, D.A., Calver, M. and Short, J. (1997). Control of feral cats for nature conservation. I. Field tests of four baiting methods. *Wildlife Research* **24**: 319-326.

- Risbey, D.A., Calver, M. and Short, J. (1997). The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. I. Exploring potential impact using diet analysis. *Wildlife Research* **26**: 621-630.
- Rix, C. (1978). *The Royal Zoological Society of South Australia 1878-1978*. Griffin Press, Adelaide.
- Robinson, A.C. (1980). Notes on the mammals and reptiles of Pearson, Dorothee and Greenly Islands, South Australia. *Transactions of the Royal Society of South Australia* **104**: 93-100.
- Robinson, A.C., Lim, L., Canty, P.D., Jenkins, R.B. and Macdonald, C.A. (1994). Studies of the yellow-footed rock-wallaby, *Petrogale xanthopus* Gray (Marsupialia: Macropodidae). Population studies at Middle Gorge, South Australia. *Wildlife Research* **21**: 473-481.
- Rodriguez, A., Barrios, L. and Miguel, D. (1995). Experimental release of an Iberian lynx (*Lynx pardinus*). *Biodiversity and Conservation* **4**: 382-394.
- Rucker, R.B. and Morris, J.G. (1997). The vitamins. In *Clinical biochemistry of domestic animals, 5th Ed*: 703-739. Kaneko, J.J., Harvey, J.W. and Bruss, M.L. (Eds.). Academic Press, San Diego.
- Sadleir, R.M.F.S. (1963). Age estimation by measurement of joeys of the euro, *Macropus robustus* Gould, in Western Australia. *Australian Journal of Zoology* **11**: 241-249.
- Sanson, G.D. (1978). The evolution and significance of mastication in the Macropodae. *Australian Mammalogy* **2**: 23-28.
- Sanson, G.D. (1989). Morphological adaptations of teeth to diets and feeding in the Macropodoidea. In *Kangaroos, Wallabies and Rat-kangaroos* 151-168. Grigg, G., Hume, I. And Jarman, P. (Eds.). Surrey Beatty & Sons: New South Wales.
- Sarrazin, F. and Barbault, R. (1996). Reintroduction: challenges and lessons for basic ecology. *Trends in Ecology and Evolution* **11**: 474-478.
- Schultze, M.O. (1960). Selenium and Vitamin E. *Annual Review of Biochemistry* **29**: 391-412.
- Schweigert F.J. (1993). Effects of energy mobilization during fasting and lactation on plasma metabolites in the grey seal (*Halichoerus grypus*). *Comparative Biochemistry and Physiology A-Comparative Physiology* **105**: 347-352.
- Sealander, J.A. (1962). Seasonal changes in blood values of Deer Mice and other small mammals. *Ecology* **43**: 107-119.
- Seddon, P.J. (1999). Persistence without intervention: assessing success in wildlife reintroductions. *Trends in Ecology and Evolution* **14**: 503.
- Serena, M. and Williams, G.A. (1994). Wildlife conservation and reintroduction: an Australian perspective. In *Reintroduction biology of Australian and New Zealand Fauna*: 247-252. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.

- Sharman, G.B., Maynes, G.M., Eldridge, M.D.B. and Close, R.L. (1995). Yellow-footed Rock-wallaby *Petrogale xanthopus*. In *The Mammals of Australia* 391-393. Strahan, R. (Ed.). Reed Books, Sydney.
- Sharp, A. (1994). Yellow-footed rock Weasels? *Wildlife Australia* Spring Ed.: 4-7.
- Sharp, A. (1997a). Insights into the dispersal patterns of yellow-footed rock-wallabies, *Petrogale xanthopus*. *Australian Mammalogy* **19**: 229-238.
- Sharp, A. (1997b). Notes on the breeding season diet of the wedge-tailed eagle *Aquila audax* in Idalia National Park, south-central Queensland. *Sunbird* **27**(4): 105-108.
- Sharp, A. and Norton, M. (2000). Dynamics of the New South Wales Yellow-footed Rock-wallaby population, in relation to rainfall patterns. *Australian Mammalogy* **22**: 71-79.
- Shepherd, N.C. (1982). Capture myopathy in the Red Kangaroo (*Macropus rufus*). In *Wildlife Diseases of the Pacific Basin and other countries*: 239-247. Fowler, M.E. (ed). Proceedings of the 4th International Conference Wildlife Disease Association, Sydney.
- Shield, J.W. and Woolley, P. (1961). Age estimation by measurement of pouch young of the quokka (*Setonix brachyurus*). *Australian Journal of Zoology* **9**: 14-23.
- Shield, J. (1971). A seasonal change in blood cell volume of the Rottneest Island quokka, *Setonix brachyurus*. *Journal of Zoology (London)* **165**: 343-54.
- Short, J.C. (1980). Ecology of the brush-tailed rock-wallaby (*Petrogale penicillata*). M.Sc. Thesis, Department of Zoology, University of Sydney.
- Short, J., Bradshaw, S.D., Giles, J., Prince, R.I.T., and Wilson, G.R. (1992). Reintroduction of macropods (Marsupialia: macropodidae) in Australia- a review. *Biological Conservation* **62**: 189-204.
- Short, J. and Turner, B. (2000). Reintroduction of the burrowing bettong *Bettongia lesueur* (Marsupialia: Potoroidae) to mainland Australia. *Biological Conservation* **96**: 185-196.
- Sinclair, A.R.E., Pech, R.P., Dickman, C.R., Hik, D., Mahon P. And Newsome A.E. (1998). Predicting effects of predation on conservation of endangered prey. *Conservation Biology* **12**: 564-575.
- Slater, G. (2001). *Monotreme and Marsupial Taxon Advisory Group: Progress Reports* (<http://www.arazpa.org.au/monotr t.htm>). Australasian Regional Association of Zoological Parks and Aquaria (ARAZPA).
- Slough, B.G. (1989). Movements and habitat use by transplanted marten in the Yukon Territory. *Journal of Wildlife Management* **53**: 991-997.
- Snyder, N.R.F., Derrickson, S.R., Beissinger, S.R., Wiley, J.W., Smith, T.B., Toone, W.D. and Miller, B. (1996). Limitations of captive breeding in endangered species recovery. *Conservation Biology* **10**: 338-348.

- Soderquist, T.R. (1994). The importance of hypothesis testing in reintroduction biology: examples from the reintroduction of the carnivorous marsupial *Phascogale tapoatafa*. In *Reintroduction biology of Australian and New Zealand Fauna*: 159-64. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Soderquist, T.R. and Serena, M. (1994). An experimental reintroduction programme for brush-tailed phascogales (*Phascogale tapoatafa*): the interface between captivity and the wild. In *Creative conservation: Interactive management of wild and captive animals* 431-438. Onley, P.J.S., Mace, G.M. and Feistner, A.T.C. (Eds). London: Chapman and Hall.
- Southgate, R.I. (1994). Why reintroduce the Bilby? In *Reintroduction biology of Australian and New Zealand Fauna*: 165-170. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Southgate, R and Possingham, H. (1995). Modelling the reintroduction of the Greater Bilby *Macrotis lagotis* using the metapopulation model analysis of the likelihood of extinction (ALEX). *Biological Conservation* **73**: 151-160.
- Southgate, R.I., Christie, P. and Bellchambers, K. (2000). Breeding biology of captive, reintroduced and wild greater bilbies, *Macrotis lagotis* (Marsupialia: Peramelidae). *Wildlife Research* **27**: 621-628.
- Southwell, C. (1989). Techniques for monitoring the abundance of kangaroo and wallaby populations. In *Kangaroos, Wallabies and Rat Kangaroos*: 659-693. Grigg, G., Hume, I. and Jarman, P. (Eds.). Surrey Beatty and Sons, New South Wales, Australia.
- Speakman, J.R., Nagy, K.A., Masman, D., Mook, W.G., Poppitt, S.D., Strathearn, G.E. and Racey, P.A. (1990). Interlaboratory comparison of different analytical techniques for the determination of oxygen-18 abundance. *Analytical Chemistry* **62**: 703-707.
- Speare, R., Donovan, J.A., Thomas, A.D. and Speare, P.J. (1989). Diseases of free-ranging macropodoidea. In *Kangaroos, Wallabies and Rat Kangaroos*: 705-34. Grigg, G., Hume, I. and Jarman, P. (Eds.). Surrey Beatty and Sons, New South Wales, Australia.
- Spencer P.B.S. (1991). Evidence of predation by a feral cat, *Felis catus* (Carnivora: Felidae) on an isolated rock-wallaby population in tropical Queensland. *Australian Mammalogy* **14**: 143-4.
- Spencer P.B.S. and Speare R. (1992). Haematology of wild allied rock-wallabies, *Petrogale assimilis* Ramsay, 1877 (Marsupialia: Macropodidae), in North Queensland. *Australian Journal of Zoology* **40**: 355-364.
- Spencer, P.B.S., Odorico, M.D., Jones, S.J., Marsh, H., and Miller D.J. (1995). Highly variable microsatellites in isolated colonies of the rock-wallaby *Petrogale assimilis*. *Molecular Ecology* **4**: 523-525.
- Stanley Price, M.R. (1986). The reintroduction of the Arabian oryx, *Oryx leucoryx* into Oman. *International Zoo Yearbook* **24/25**: 179-188.
- Stanley Price, M.R. (1989). *Animal reintroductions: the Arabian oryx in Oman*. New York: Cambridge University Press.

- Stanley Price, M.R. (1991). A review of mammal re-introductions, and the role of the Re-introduction Specialist Group of IUCN/SSC. In *Beyond captive breeding: re-introducing endangered mammals to the wild*: 9-25. Gipps, J. H. W. (Ed.). Oxford, Clarendon Press.
- Stanley Price, M.R. and Fairclough, A. (1997). Translocation of wildlife: the IUCN position statement and general considerations on behavioural constraints to release. *Suppl. Ric. Biol. Selvaggina*. XXVII 25-38.
- Strahan, R. (1995). *The Mammals of Australian*. Reed Books, Sydney.
- Stuart-dick, R.I. and Higginbottom, K.B. (1989). Strategies of parental investment in Macropodoids. In *Kangaroos, Wallabies and Rat Kangaroos*: 571-592. Grigg, G., Hume, I. and Jarman, P. (Eds.). Surrey Beatty and Sons, New South Wales, Australia.
- Sunquist, F. (1993). Should we put them all back? *International Wildlife* **22**: 35-40.
- Sunnucks, P. (2000). Efficient genetic markers for population biology. *Trends in Ecology and Evolution* **15**: 199-203.
- Sunnucks, P. and Hales, D.F. (1996). Numerous transposed sequences of mitochondrial cytochrome oxidase I-II in aphids of the genus *Sitobion* (Hemiptera: Aphididae). *Molecular Biology and Evolution* **13**: 510-524.
- Swihart, R.K. and Slade, N.A. (1985). Testing for independence of observations in animal movements. *Ecology* **166**: 1176-1184.
- Swofford, D.L. and Selander, R.B. (1981). BIOSYS-1: a FORTRAN program for the comprehensive analysis of electrophoretic data in population genetics and systematics. *Journal of Heredity* **72**: 281-283.
- SYSTAT (1998). SYSTAT Version 8.0. Copyright SPSS Inc.
- Taylor, R.J. and Rose, R.W. (1987). Comparison of growth of pouch young of the Tasmanian bettong, *Bettongia gaimardi*, in captivity and the wild. *Australian Wildlife Research* **14**: 239-246.
- Torr, G. and Beaton, E. (2000). Yellow-footed Rock-wallabies: between a rock and a hard place. *Australia Geographic* **60**: 88-101.
- Troy, S. and Coulson, G. (1993). Home range of the swamp wallaby, *Wallabia bicolor*. *Wildlife Research* **20**: 571-577.
- Tunbridge, D. (1991). *The Story of the Flinders Ranges Mammals*. Kangaroo Press, New South Wales.
- Underwood, A. (1995). Where have all the malas gone? *International Wildlife* **25**: 14
- Van Dierendonck, M. and Wallis De Vries, M.F. (1996). Ungulate reintroductions: experiences with the Takhi or Rzewalski Horse (*Equus ferus przewalskii*) in Mongolia. *Conservation Biology* **10**: 728-740.

- Vargas, A. and Anderson, S.H. (1998). Ontogeny of black-footed ferret predatory behaviour towards prairie dogs. *Canadian Journal of Zoology* **76**: 1696-1704.
- Vargas, A. and Anderson, S.H. (1999). Effects of experience and cage enrichment on predatory skills of black-footed ferrets (*Mustela nigripes*). *Journal of Mammalogy* **80**: 263-269.
- Veitch, C.R. (1994). Habitat repair: a necessary prerequisite to translocation of threatened birds. In *Reintroduction biology of Australian and New Zealand Fauna*: 97-104. Serena, M. (Ed). Surrey Beatty & Sons, Chipping Norton.
- Viggers, K.L., Lindenmayer, D.B., Spratt, D.M. (1993). The importance of diseases in reintroduction programs. *Wildlife Research* **20**: 687-698.
- Wallis, I.R. and Green, B. (1992). Seasonal field energetics of the Rufous Rat-kangaroo *Aepyprymnus rufescens*. *Australian Journal of Zoology* **40**: 279-290.
- Waples, K.A. and Stagoll, C.S. (1997). Ethical issues in the release of animals from captivity. *BioScience* **47**: 115-121.
- Wells, R.T. (1978). Thermoregulation and activity rhythms in the Hairy-nosed Wombat, *Lasiorhinus latifrons* (Owen), (Vombatidae). *Australian Journal of Zoology* **26**: 639-651.
- White, R.G., Hume, I.D. and Nolan, J.V. (1988). Energy expenditure and protein turnover in three species of wallabies (Maesupialia: Macropodidae). *Journal of Comparative Physiology B* **158**: 237-246.
- Wilson, G., Dexter, N., O'Brien, P. and Bomford, M. (1992). *Pest animals in Australia. A survey of introduced wild animals*. Kenthurst: Bureau of Rural Resources and Kangaroo Press.
- Withers, P.C. (1992). *Comparative Animal Physiology*. Saunders College Publishing, U.S.A.
- Wood, J.T., Carpenter, S.M., Poole, W.E. (1981). Confidence intervals for ages of marsupials determined from body measurements. *Australian Wildlife Research* **8**: 269-74.
- Woodford, M.H. and Kock, R.A. (1991). Veterinary considerations in re-introduction and translocation projects. In *Beyond captive breeding: re-introducing endangered mammals to the wild*: 101-110. Gipps, J.H.W. (Ed.). Oxford, Clarendon Press.
- Woodford, M.H. and Rossiter P.B. (1994). Disease risks associated with wildlife translocation projects. In *Creative conservation: Interactive management of wild and captive animals*: 178-200. Onley, P.J.S., Mace, G.M. and Feistner, A.T.C. (Eds). London: Chapman and Hall.
- Wolf, C.M., Griffith, B., Reed, C. and Temple, S.A. (1996). Avian and mammalian translocations: update and reanalysis of the 1987 survey data. *Conservation Biology* **10**: 1142-1154.
- Zar, J.H. (1984). *Biostatistical analysis*, 2nd Ed. Prentice-Hall, New Jersey.