

An independent review
of the evidence under-pinning the

Rewilding of Southern Yorke Peninsula

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Executive Summary

Southern Yorke Peninsula has changed markedly since European colonisation of the area in the 1830s. Temperatures have increased, rainfall has decreased, over half of the native vegetation has been replaced by agriculture, exotic plants and animals have been introduced, and 24 (80%) of the 30 non-volant native mammals that once occurred there are now locally extinct. These fundamental changes to the ecology of the Yorke Peninsula affect the sustainability of remaining native biodiversity and the livelihoods of people living there.

The rewilding of southern Yorke Peninsula project aims to re-establish lost ecological processes on Yorke Peninsula by controlling populations of key exotic animals (foxes, cats and mice) and reintroducing locally extinct native mammals to re-establish their ecological roles. Locally extinct mammals have been chosen to re-establish soil-engineers to improve water-permeability, mycorrhizal fungi and soil fertility (southern brown bandicoot and brush-tailed bettong), improve seed dispersal and recruitment of native plants (native rodents) and re-establish mammalian predators (red-tailed phascogale, western quoll and possibly Tasmanian devil). This report summarises the current state of scientific knowledge as it relates to each of these key components of the project.

Introduced mammalian predators have been responsible for biodiversity declines, species extinctions, reduced ecosystem services, economic losses, and health impacts on humans around the world. In Australia predation by introduced foxes and cats has resulted in population declines and extinction of numerous native mammals, including more than half of the species that are now locally extinct on Yorke Peninsula. Several Australian native mammal species currently persist only in cat and fox free refuges (islands and fenced areas on the mainland). Foxes and feral cats are recognised as key threatening processes under the Australian Environment Protection and Biodiversity Conservation Act 1999. Foxes incur significant costs to agriculture as vectors for invasive weeds and by preying on lambs. Cats incur significant costs to agriculture and human health as vectors of disease (e.g. toxoplasmosis and sarcosporidiosis). The presence of foxes commonly suppresses the density of cats, and fox control can result in an increase in the density of cats. Therefore, both species should be managed in concert. Both foxes and cats can be effectively controlled over large areas using sodium fluoroacetate (1080) meat baits of an appropriate type for each species. Following baiting, foxes and cats from surrounding areas commonly disperse into baited areas. Unless it can be curtailed, this dispersal reduces the effectiveness and increases the cost of baiting programs.

Barrier fences are widely used to limit dispersal of cats and foxes and provide safe havens for native mammals. Foxes and cats can scale netting fences over 2m high, leap over 1.4m high fences and chew through soft wire netting. Effective barrier fences should be made of hard wire netting fences at least 180 cm high with a foot apron and a 60 cm-wide curved overhang ("floppy top"). The use of metal (rather than wooden) fence posts greatly reduces the ability of cats and foxes to climb a fence. A "floppy top" stops animals from climbing over the fence. Monitoring and maintenance of barrier fences are important for long-term limitation of cat and fox dispersal. Barrier fences have the disadvantage of limiting movement of non-target wildlife, such as kangaroos

and emus. Gaps in the barrier fences across Yorke Peninsula will allow some movement of cats and foxes and native wildlife where the fences cross roads and meet the coast. Nevertheless, the fences will greatly curtail dispersal of foxes and cats into the project area, enhancing the effectiveness and reducing the cost of fox and cat control.

Introduced house mice are one of the most important agricultural pests globally and incur significant costs to agriculture in the southern grain growing regions of Australia. Mice damage crops in the field, consume grain, and foul harvested grain in storage. Mice tend to be less common in intact native vegetation ($< 6 \text{ mice.ha}^{-1}$) than in agricultural land. Mice use farm buildings and remnant vegetation as refuges when fields lie fallow but move into fields and breed-up as crops grow. Most agricultural costs occur during erratic mouse plagues, which occur when high food availability following rains allows population densities to reach $>2,500 \text{ mice.ha}^{-1}$. Mouse plagues occur on average every seven years in South Australia. Currently mice are controlled using costly, toxic baits laid during plagues to reduce the magnitude of peak population density. Like baiting, predators are unlikely to stop plagues from occurring, but enhancing the density of predators should limit the magnitude of plagues at considerably less cost compared to pesticides. Small raptorial birds (barn owls, kestrels and black-shouldered kites) are the primary predators of mice in Australia, but their populations are limited in agricultural areas due to a lack of nesting sites. Provision of artificial nesting sites (nest boxes) and perches from which to hunt have been used in many parts of the world for over fifty years to artificially increase the density of avian predators to control rodents. Nocturnal barn owls have been the focus of this approach. Artificially increased densities of avian predators have resulted in reduced rodent abundance and increased crop productivity.

Ecosystem engineers are organisms that control the availability of resources to other organisms by making physical changes to the environment. Soil engineers dig burrows for shelter and turn over soil as they forage. The importance and effectiveness of each species of soil engineer can vary, suggesting that some redundancy is required to realise the full advantages of soil engineers in a landscape. Of the seven mammalian soil engineers that once occurred on southern Yorke Peninsula, only the echidna remains. Introduced rabbits do not provide the ecosystem services provided by native soil engineers. Southern brown bandicoots and brush-tailed bettongs have been chosen for reintroduction to Yorke Peninsula because they act as soil engineers that will help restore landscape function. Establishing an additional population will also assist established recovery programs for both species. Less than 2000 brush-tailed bettongs remain in the wild and they are one of the most threatened marsupials. Neither bandicoots or bettongs burrow, but individuals displace 4-5 tonnes of soil per year while foraging. The foraging pits they produce increased water infiltration, soil fertility, communities of mycorrhizal fungi and enhanced seedling recruitment and growth of native vegetation.

Remnant native vegetation on Yorke Peninsula is in gradual decline. Dispersal of seeds is important for recruitment in existing plant populations and allows plants to recolonise disturbed areas. Globally, rodents play an important role in seed dispersal and the dynamics of plant communities. While rodents consume seeds, they also disperse undigested seeds in their faeces and bury seed in caches where some of the seeds germinate. Seeds handled by rodents are more likely to germinate than seeds that have not been handled by them. The importance of rodents in plant population dynamics in Australia has been underestimated due to historical rodent extinctions and population declines. All eight native rodent species that once occurred on Yorke Peninsula are locally extinct. In addition to assisting dispersal and recruitment of declining plants, these missing rodents appear to have played a role in suppressing native, invasive woody shrubs.

Predators play a key role in maintaining species diversity and ecosystems globally. Dramatic re-organisation of ecosystems and biodiversity losses frequently follow the extinction of predators in marine, aquatic and terrestrial environments. Restoring and maintaining the ecological function of predators is a critical component of restoration ecology in general and of rewilding southern Yorke Peninsula in particular. All ten species of native predators on Yorke Peninsula are locally extinct. The red-tailed phascogale and western quoll are currently being considered for translocation to Yorke Peninsula because they are likely to control mouse and rabbit populations, and possibly feral cats. An additional benefit is the establishment of additional populations of these species for conservation. There are good grounds for going ahead with planned translocation of quolls and phascogales. Both are active, opportunistic, generalist predators. Historical accounts suggest that quolls were, at least partly, responsible for the failure of nearly 300 independent releases of rabbits in Australia between 1788 and 1900, suggesting they can regulate rabbit populations. Mice are a consistent feature of the diet of both western quolls and red-tailed phascogales, and make-up a large proportion of prey items consumed during mouse plagues by both species. Some preliminary discussions have also begun regarding the potential to translocate Tasmanian Devils to southern Yorke Peninsula in 20 years' time. Tasmanian devils are the world's largest extant carnivorous marsupial and regulate the abundance of smaller native predators. The disappearance of Devils in some areas of Tasmania, due to facial tumour disease, has been accompanied by increases in the abundance of feral cats, leading to the suggestion that Devils may regulate cat populations. However, the critical information required to objectively assess this hypothesis are not yet available. The impact of devils on sheep is a point of social controversy in Tasmania, which has not been clarified in the literature available to us. Should the impact of devils on sheep be sufficiently low and the impact of devils on feral cats be confirmed, then translocating them to Yorke Peninsula may provide benefits to agriculture by reducing the incidence of feral cat and cat-borne diseases. The intervening 20 years prior to the potential introduction of the devils will allow for the collation of additional studies and information, enabling a considered evaluation of the risks involved to be made.

All species of mammals chosen for translocation to southern Yorke Peninsula have a naturally high tolerance to 1080 and are known to survive in areas where 1080 baits are used to control foxes and cats.

Key knowledge gaps are identified for each component of the project to guide research, monitoring and evaluation to aid adaptive management to maximise the project's chances of success. The southern Yorke Peninsula rewilding project presents an opportunity to (1) increase our understanding of Australian ecosystems, (2) understand how their historical decline has affected human endeavour, and (3) demonstrate rewilding as means to re-establish lost ecological processes on Yorke Peninsula with far-reaching implications for future of biodiversity and human society.

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Figure 1.1: Map of the southern Yorke Peninsula from Taggart (2014).

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1 Rewilding southern Yorke Peninsula

1.1 What is rewilding?

Rewilding is an approach to restoration ecology that assists natural processes to repair damaged ecosystems and restore degraded landscapes (Sandom et al. 2013, Bakker and Svenning 2018, Gilby et al. 2018). Traditional restoration programmes have aimed to re-establish viable populations of specific threatened species and in doing so to conserve biodiversity (Miller and Hobbs 2007, Soorae 2008, Soorae 2010). They require ongoing management to ensure conservation goals are maintained. In contrast, rewilding seeks to restore ecosystem functions by reintroducing keystone species to regionally connected networks of protected areas (Soulé and Terborgh 1999, Terborgh and Estes 2010, Estes et al. 2011, Seddon et al. 2014). Keystone species have a disproportionately large impact on ecosystem processes in proportion to their biomass (Paine 1969). Rewilding programmes aim to conserve and restore biodiversity, by rebuilding ecological processes so that more functional, natural communities are better able to maintain themselves and continue to provide ecosystem services upon which humanity depends (Millennium Ecosystem Assessment 2005).

Over the past two decades rewilding programmes have been instigated on all continents and vary in geographic extent and complexity (Fraser 2009). The introduction of Aldabran giant tortoises (*Aldabrachelys gigantea*) to a 25 Ha island near Mauritius in the western Indian Ocean is an example of a rewilding project effecting a small geographic area and involving reintroduction of a single species to replace lost ecosystem processes (Griffiths et al. 2011, Falcón and Hansen 2018). On Ile aux Aigrettes hunting resulted in the extinction of all large-bodied frugivores in the 16th century which limited seed dispersal and germination in the critically endangered, endemic ebony tree (*Diospyros eggettatum*). Ebony forests had once covered the island, holding the soil together with their roots and providing habitat for other organisms. Endemic giant tortoises had once eaten ebony fruits and dispersed their seeds in their faeces, aiding recruitment of ebony trees. After the island was largely cleared of ebony for firewood, the remaining ebony trees were unable to recover in the absence of the native giant tortoises. The ecological process which the tortoises provided was absent and limited ecological recovery on Ile aux Aigrettes. Replacing the extinct endemic giant tortoises with closely related Aldabran tortoises allowed the ebony seeds to germinate and disperse, allowing recovery of the ebony forests, limiting erosion and invasion by alien plants (Griffiths et al. 2010, Falcón and Hansen 2018).

The Pleistocene Park project is an example of a rewilding project over a larger geographic area and involving the reintroduction of multiple species (Stone 1998, Zimov 2005, Olofsson and Post 2018). This project is focussed on the 160 km² area of the Siberian Republic of Yakutia but has implications for the future of Arctic tundra globally. During the Pleistocene (2 million - 10,000 years ago) this area was a seasonally freeze-thawing steppe maintained by herbivores such as mammoths, woolly rhinoceroses, bison, horses, reindeer, musk oxen, elk, moose, saiga antelope, and yaks. Many of these large herbivores were hunted to extinction about ten thousand years ago and steppe replaced by mossy and forest tundra. The frozen Siberian soils lock up over twice as much organic carbon as the world's rainforests (Zimov 2005). Increased vegetation cover in these areas is causing

the permafrost to melt, releasing carbon into the atmosphere and exacerbating climate change. While mammoths and woolly rhinoceroses are globally extinct, remaining herbivores that had become locally extinct are being reintroduced to restore the steppe ecosystem, prevent thawing of the permafrost by increasing soil stability and increasing landscape surface albedo to combat climate change (Zimov 2005). Reintroduced species include musk ox (*Ovibos moschatus*), and European bison (*B. bonasus*) (Nicholls 2006). This ecological restoration project has the potential to reinstate ecosystem processes that maintained the "mammoth steppe", once one of the world's most extensive ecosystems (Olofsson and Post 2018), and in so doing influence processes that influence global climate (Zimov 2005).

The Yellowstone to Yukon project is probably the largest attempt at rewilding. It aspires to restore ecological processes over a 1.3 million km² area in north-west North America by connecting established national parks and reintroducing multiple keystone species for biodiversity and community benefits. The reintroduction of wolves to alleviate elevated grazing pressure in Yellowstone National Park has been at the forefront of developing the rewilding approach to biodiversity conservation (Smith et al. 2003, White and Garrott 2005, Beyer et al. 2007, Ripple and Beschta 2007). Several established ecological restoration projects in Australia are based on the rewilding approach (Australian Rewilding Network 2018), including 'Gondwanalink' in Western Australia (Jonson 2010) which incorporates a matrix of remnant vegetation and agricultural land.

Rewilding projects are necessarily adaptive in their management approach because our knowledge of ecological processes is limited. For this reason monitoring should follow each management action in order to assess whether it achieved the desired outcomes, and to develop better understanding of the mechanisms behind expected and unexpected responses to management actions (Torres et al. 2018). This document summarises the current state of scientific knowledge underpinning the proposed rewilding of southern Yorke Peninsula (Sharp 2018), with a view to identifying knowledge gaps and minimising the risk of undesirable outcomes.

1.2 Southern Yorke Peninsula

The southern Yorke Peninsula covers 360,000 hectares from Innes National Park on the south-western tip of the peninsula to Curramulka on central Yorke Peninsula. The boundary also extends to the Point Pearce community and Aboriginal Lands near Port Victoria in the north-west (Figure 1.1).

Southern Yorke Peninsula is a highly modified agricultural landscape, with significant areas of remnant vegetation. The southern Yorke Peninsula rewilding project will (1) manage overabundant pest species that erode the integrity of natural ecosystem processes and agricultural productivity, and (2) re-introduce native mammals that have become locally extinct (Taggart 2014) to re-establish natural ecosystem processes. The Eyre and Yorke Peninsula IBRA region has been identified as a priority site for mammal conservation (Ringma et al. 2019). Southern Yorke Peninsula is also recognised as an Important Biodiversity Area that supports breeding malleefowl (*Leipoa ocellata*) and mallee whipbirds (*Psephodes leucogaster*) (Birdlife International 2018). The southern Yorke Peninsula rewilding project will attempt to restore lost ecological processes to revitalise environment, economy & society.

1.3 Environment

1.3.1 Climate

Yorke Peninsula has a Mediterranean climate, with mild wet winters and hot dry summers (Figure 1.2). Rainfall is highest on the SW 'toe' of the peninsula with (mean annual rainfall of 447 mm at Stenhouse Bay) and becomes drier further east (381mm at Edithburgh) and north (425mm at Curramulka; Australian Bureau of Meteorology 2018).

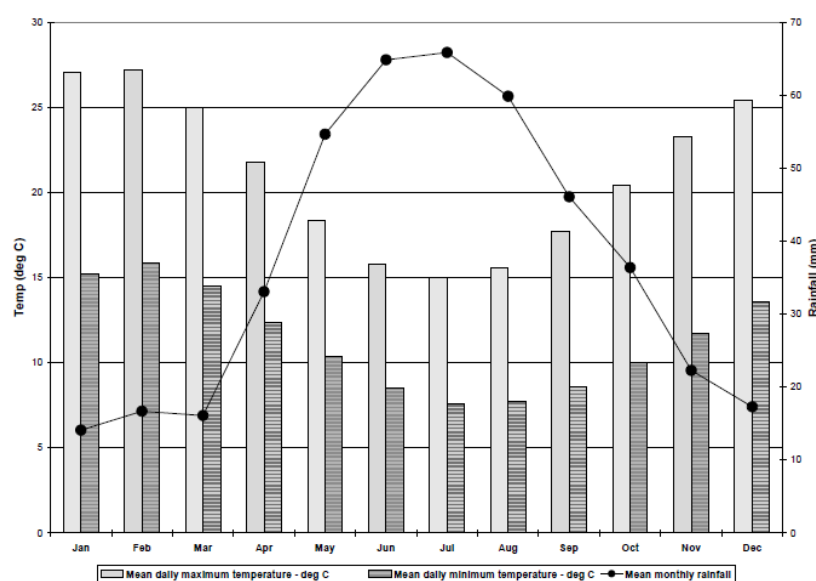


Figure 1.2: Mean daily maximum and minimum temperature (°C) and mean monthly rainfall (mm) at Warooka on southern Yorke Peninsula. From Neagle (2008).

The climate of Yorke Peninsula is changing in response to natural factors and human-induced increases in atmospheric pollution (Environment Protection Authority 2018, International Intergovernmental Panel on Climate Change 2018). Overall the Northern and Yorke Natural Resources Management Region projected temperatures are expected to increase by up to 3.7°C and rainfall is expected to decline by up to 27% by 2090 (Environment Protection Authority 2018). More specifically, rainfall at Warooka on southern Yorke Peninsula shows considerable variation between years, but an overall declining trend (Figure 1.3). Temperatures at Warooka show an increasing trend over time (Figure 1.4). Potential impacts of predicted sea level rises have been described elsewhere (Australian Water Environments 2015).

1.3.2 Geology, Geomorphology and Soils

The geology and geomorphology of southern Yorke Peninsula has been described in detail elsewhere in detail (Crawford and Ludbrook 1965, Corbett and Scrymgeour 1973, Field Geology Club of South Australia 1997, Pan et al. 2018). What follows is a brief summary derived from those sources.

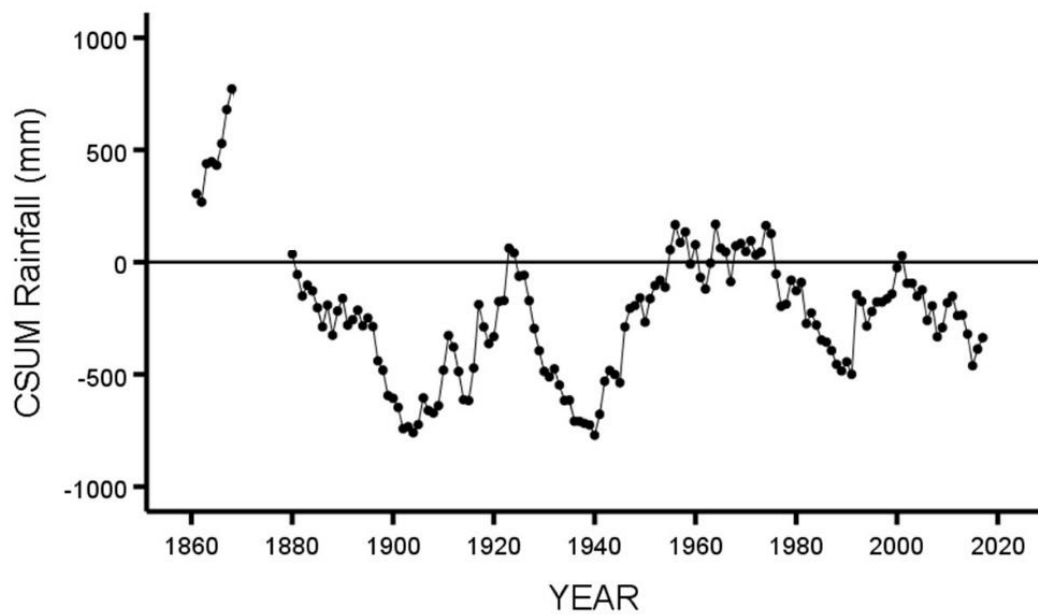


Figure 1.3: The cumulative sum of deviations (CUSUM) from the mean rainfall (mm) at Warooka between 1861 and 2017. The zero value (and horizontal line) on the y-axis represents the average rainfall. Years with below average rainfall drive the CUSUM value down, whereas years above average rainfall drive the CUSUM value up. Wet periods are indicated by values above the horizontal line, dry periods are indicated by values below the horizontal line. No rainfall records were kept between 1869 and 1879.

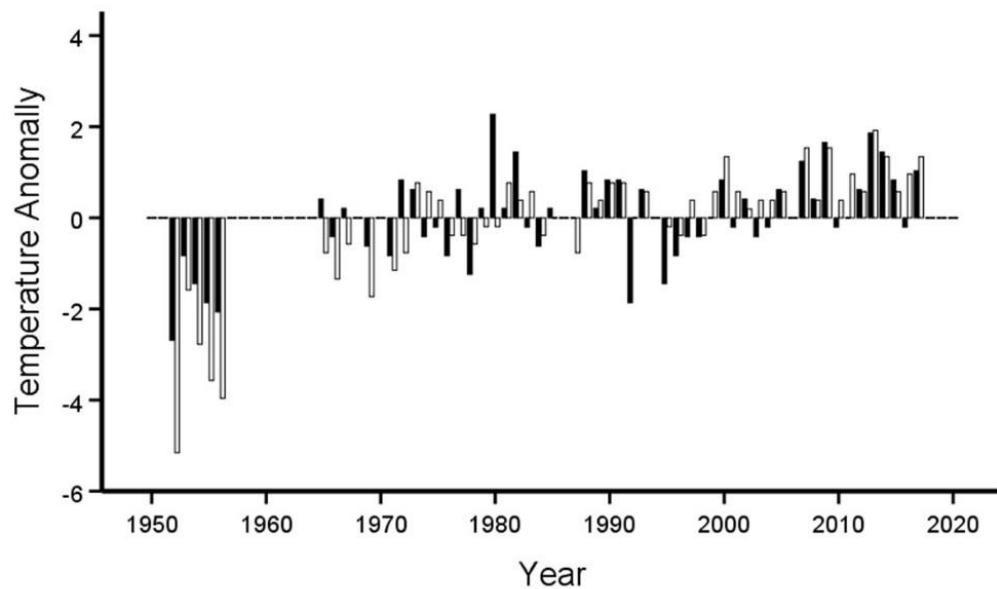


Figure 1.4: Temperature anomaly showing increasing mean annual maximum (■) and minimum (□) temperatures at Warooka between 1952 and 2017. Temperature anomalies are expressed as the deviation between observed and mean values in units of standard deviation. Colder than average years have negative values and hotter than average years have positive values.

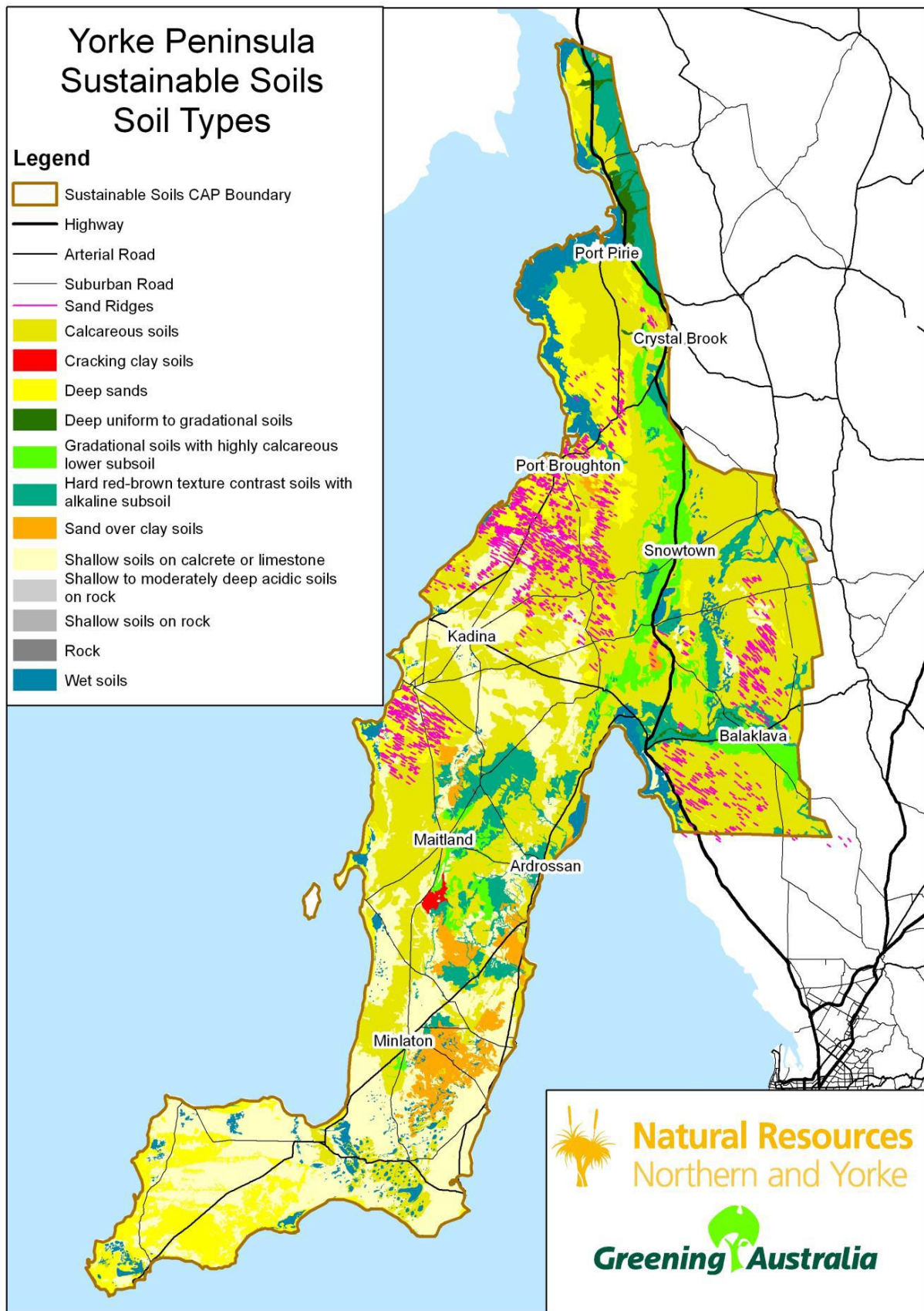


Figure 1.5: Map showing the distribution of soil types on Yorke Peninsula. From McGregor (2016)

Rock outcrops on Yorke Peninsula are concentrated along the coast. Early Proterozoic (1845–1710 million years old) granite gneisses and amphibolites are conspicuous at the base of coastal cliffs. There is a long gap in the geological sequence visible on southern Yorke Peninsula. Glaciers eroded the surface of older rocks 295 million years ago, and deposited rocks that had been transported from elsewhere by glaciers, called erratics. Erratics are often conspicuously different from local rock outcrops. Another long gap in the geological sequence follows the Permian glaciation. Most visible rocks on Yorke Peninsula were deposited during the Cainozoic (i.e. over the past 2.5 million years). This was a period when sea level changes by more than 100m during a series of cyclic expansion (glaciation) and reduction of the polar icecaps. During the glaciations water was locked in the icecaps the sea-level the coast of Southern Australia approximated the edge of the continental shelf, much further south than the current coast. During these periods Yorke Peninsula was continuous with Kangaroo Island to the south, Eyre Peninsula to the west and the Fleurieu Peninsula to the east. The Spencer and St Vincent Gulfs were broad alluvial valleys. During the interglacial periods the sea level was sometimes higher than it is presently, and the area between Hardwicke and Sturt Bays would have been a sea strait. The Cainozoic cycles had a periodicity of ~ 100 thousand years. We are currently in an interglacial phase.

The coast of Yorke Peninsula is largely defined by a series of faults. These faults lie at the edge of a horst (an uplifted area, such as Yorke Peninsula) and grabens (down-thrust areas, such as Spencer and St Vincent Gulfs on either side of the peninsula). The faults were caused as Australia and Antarctica broke away from each other about 65 million years ago, and Australia drifted north towards its present location. The southern half of Yorke Peninsula is an undulating plain rising to a maximum of 200 metres in the centre of the Peninsula. Landforms include semi-saline wetland systems in low-lying areas near Warooka and Yorketown, rugged high-energy coastlines in the south-west, low-energy cliffs and dunes on the east and upper west coast and undulating to low hilly plains throughout inland areas. No part of the peninsula is more than 20km from the coast.

The soils of southern Yorke Peninsula can broadly be classified as deep carbonate sands in the south-west, shallow soils on calcrete or limestone, naturally saline wetlands, duplex sand over clay, and calcareous soils (McGregor 2016; Figure 1.5). The soils of Yorke Peninsula are naturally of low fertility (Prescott and Piper 1932, French et al. 1969). The use of fertilizers for crops and nutrient supplements for stock has increased agricultural productivity on Yorke Peninsula, as it has across much of southern Australia. There are particularly productive areas for agricultural crops around Warooka and, further north, between Maitland and Ardrossan (McGregor 2016). Soil salinization has resulted from vegetation clearance in large areas of the agricultural regions of Australia (Natural Heritage Trust 2001), South Australia (Jolly et al. 2000), and parts of Yorke Peninsula (Northern and Yorke Natural Resources Management Board 2008, cf. McGregor 2016).

1.3.3 Biogeography

The plants and animals on Yorke Peninsula include species that occur either (1) on the northern part of the Peninsula and are widely-distributed across the hot, dry interior, or (2) on the south-western ‘toe’ of the Peninsula and are most closely related to species in Victoria or south-western Australia (Neagle 2008). As such, Yorke Peninsula lies between the Eyrean (dry, hot inland) and Bassian (cooler, wetter southern) biogeographic sub-regions (Keast 1981). Southern Yorke Peninsula is an ‘island’ of suitable habitat for many Bassian species, such mallee whipbirds (McGuire et al. 2011, Burbidge et al. 2017), heath goanna (*Varanus rosenbergii*; Smith et al. 2007) and many plants (Wood

1930, Chandler and Crisp 1997). The cyclic sea level changes during the Cainozoic (see 1.1.1.2 Geomorphology, Geology and Soils) allowed these Bassian species to repeatedly expand and contract their geographic ranges over time, and in some cases lead to the divergence of relict populations and the evolution of endemic species (Wood 1930, Schmitt 1978, Schmitt and White 1979, Keogh et al. 2005, Byrne 2008, Myers et al. 2012, Mesibov 2017). The most recent sea-level rise dissected the once continuous Bassian biota into relict patches on the southern tips of Eyre, Yorke and Fleurieu Peninsulas and Kangaroo Island (Wood 1930; Figure 1.6, Crocker and Wood 1947, Keast 1981).

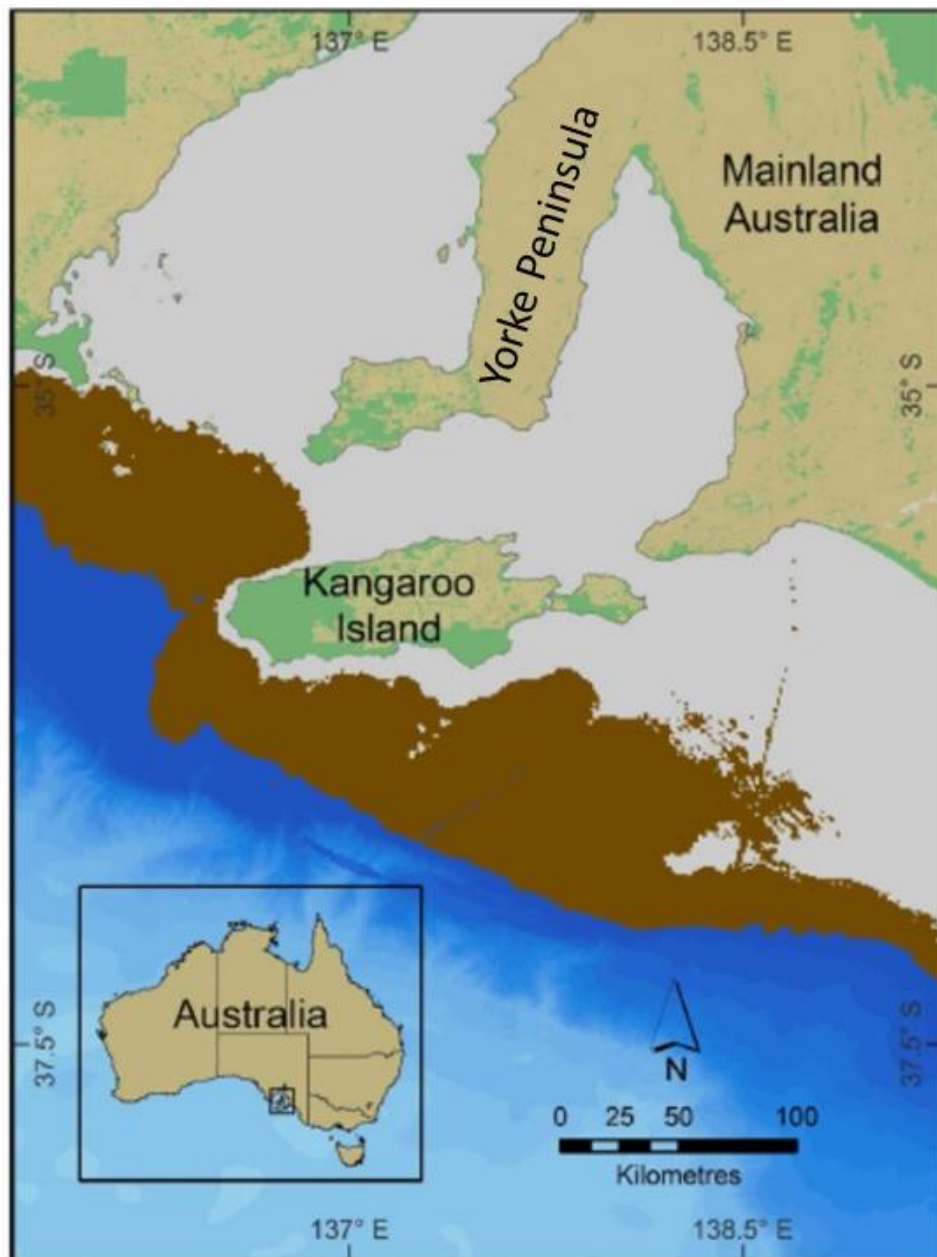


Figure 1.6: Map showing the changing coastline of South Australia during Cainozoic glacial cycles. Tan shading represents the present-day exposed land mass, with remnant vegetation shown in green. Grey shading represents exposed land 14 thousand years ago (60 m below present sea level). Brown shading represents exposed land during the height of the last glacial maximum between and 17 and 22 thousand years ago (125 m below present sea level). Modified from Peacock et al. (2018)

1.3.4 Native Vegetation

The natural vegetation on Yorke Peninsula reflects the impoverished soils and high natural incidence of fires (Keith et al. 2014). It has been described in detail previously (French et al. 1969, Specht 1972, Neagle 2008, Mahoney 2013). Much of the native vegetation in the far south west of the region remains intact, as the deep calcareous sands are unsuitable for intensive agriculture. Other areas have been extensively cleared for agriculture, with only roadside strips and isolated patches of native vegetation remaining (Neagle 2008).

The deep sands on the south west of Yorke Peninsula supported Kingscote mallee–coastal white mallee (*Eucalyptus rugose* – *E. diversifolia*) associations with patches of vegetation dominated by white mallee (*E. dumosa*–*E. phenax*) (Croft 2008). Remnants of this vegetation indicate a shrubby understorey that includes dryland tea-tree (*Melaleuca lanceolata*), mallee wreath wattle (*Acacia triquetra*), rock wattle (*A. rupicola*) and wedge-leaved pomaderris (*Pomaderris obcordata*). By contrast, the shallow soil overlying calcrete further north and east supported dryland tea-tree – drooping sheoak (*M. lanceolata*–*Allocasuarina verticillata*) low open woodland (Croft 2008). Prior to being grazed the understorey was probably open and grassy with sparse shrubs or understorey trees (Croft 2008). Low shrublands occur on exposed rocky cliffs are dominated by pale turpentine Bush (*Beyeria lechenaultii*), coast velvet-bush (*Lasiopetalum discolor*), cushion bush (*Leucophyta brownii*) and cushion fan-flower (*Scaevola crassifolia*). Shrublands on coastal dunes are dominated by coast daisy-bush (*Olearia axillaris*), coastal beard-heath (*Leucopogon parviflorus*), wattles (*Acacia spp*) and common boobialla (*Myoporum insulare*). Ephemeral wetlands are dominated by samphire shrublands or swamp paperbark (*Melaleuca halmaturorum*) forests, occasionally with cutting grass (*Gahnia filum*) sedgelands on less saline ground (Berkinshaw et al. 2017; Figure 1.7).

Of the 659 native vascular plant species recorded on southern Yorke Peninsula, 21 (3%) are critically endangered, 43 are (6%) endangered, 62 are (9%) vulnerable to extinction and 141 species (21%) considered Rare (Gillam and Urban 2008). Species of note include several threatened orchids (e.g. *Caladenia brumalis*, *C. macroclavia*, *C. conferta*, *C. intuta*, *Prasophyllum goldsackii*), two nationally threatened wattle species (*Acacia rheticarpa*, *A. enterocarpa*), silver daisy-bush (*Olearia pannosa*), annual candles (*Stackhousia annua*) and several swamp-associated species including bead samphire (*Tecticornia flabelliformis*) and silver candles (*Pleuropappus phyllocalymmeus*) (Berkinshaw et al. 2017).

1.3.5 Native Fauna

The south-western tip of the Yorke Peninsula is a refuge area for many mammal, bird and reptile species which are declining or have disappeared from elsewhere in southern South Australia. Fourteen mammal, bird and reptile species of national or state conservation significance occur on southern Yorke Peninsula (Berkinshaw et al. 2017). Four species are listed as nationally threatened including the Australian sea-lion (*Neophoca cinerea*), plains wanderer (*Pedionomus torquatus*), the malleefowl, and the mallee whipbird. The beaches are also home to an important population of hooded plovers (*Thinornis cucullatus*). Although no bird extinctions have been recorded on Yorke Peninsula, many mallee bird species are declining in abundance (Birdlife Australia 2015, Boulton and Lau 2015). Of the 258 vertebrate species recorded on the southern Yorke Peninsula seven (3%) are critically endangered, 24 (9%) are endangered, 28 (11%) are vulnerable to extinction and 68 (26%) are

considered rare (Gillam and Urban 2008). Two species of native millipede are known only from Yorke Peninsula (Mesibov 2017). A detailed account of the fauna has been provided by Corbett (1973) and Neagle (2008).

Like the rest of Australia, the fauna of Yorke Peninsula has been depleted by Quaternary (2.5 million years before present - present) and historical mammal extinctions (Pledge 1992, Johnson 2006a, McDowell et al. 2012). Fossils up to ~ 5 million year old from caves in the Curramulka area include at least 27 species of mammals (Pledge 1992). Notable among the species represented are a giant koala, ring-tailed possum, wombat, short faced kangaroo and several diprotodontids. Marsupial lions (*Thylacoleo*) and thylacine (*Thylacinus cynocephalus*) were also present. Modern records and subfossils from Balgawa, Innes National Park and Troubridge Hill (Figure 1.6) show that 30 species of non-volant, terrestrial native mammals inhabited southern Yorke Peninsula at the time of European settlement (Aitken 1973, Brandle 2008, McDowell et al. 2012; Table 1.1), 24 (80%) of which no longer occur on the peninsula. The causes of these local mammal extinctions include habitat clearance, hunting and the introduction of foxes, cats and rabbits (Aitken 1973, Johnson 2006a). The local extinction of most of the mammal fauna represents a major loss of natural ecosystem processes.

1.3.6 Key ecological processes

Known key ecological processes in on southern Yorke Peninsula are summarised in Figure 1.8. This figure is modified from Keith et al. (2014) summary of ecological processes in heathlands, so that it relates specifically to mallee communities in agricultural landscapes. The central panel of the figure reflects trophic flow of nutrients upwards from soil to vegetation, and thence to herbivorous and predatory animals. Key natural drivers are indicated to the left, and anthropogenic drivers are indicated to the right. Arrows indicate ecological processes that determines the structure and function of the community. Downward-pointing arrows indicate top-down processes, whereas upward-pointing arrows indicate bottom-up processes.

Our knowledge of natural ecosystem processes in Australia generally has largely developed since the extinction of a large proportion of native mammals (Johnson 2006a). Therefore, the model outlined in Figure 1.8 reflects the best available information, but it does not accurately reflect the importance of native mammals as drivers of ecological processes. From ecological restoration projects it is becoming increasingly clear that native mammals can play pivotal roles in Australian ecosystems (e.g. Fleming et al. 2014, Morris and Letnic 2017, Mills et al. 2018). The rewilding of Yorke Peninsula is likely to greatly improve our understanding of the role of mammals in southern Australian ecosystems, with commensurate improvements in our ability to manage these landscapes for biodiversity, societal and economic outcomes in the future (Sharp 2018).

1.3.6.1 Natural drivers of ecological processes

The first natural driver of ecosystem dynamics on southern Yorke Peninsula is the nutrient status of the soils (Figure 1.8). The soils on the peninsula are nutrients poor due to prolonged leaching over geological time and the high lime content in the soils, which reduces the availability to plants of the nutrients that are present (Prescott and Piper 1932, Specht 1972, Groves 1981). Such nutrient poor soils limit the kinds and abundance of plants that can live on them (French et al. 1969, Specht 1972, Lambers et al. 2010, Keith et al. 2014).

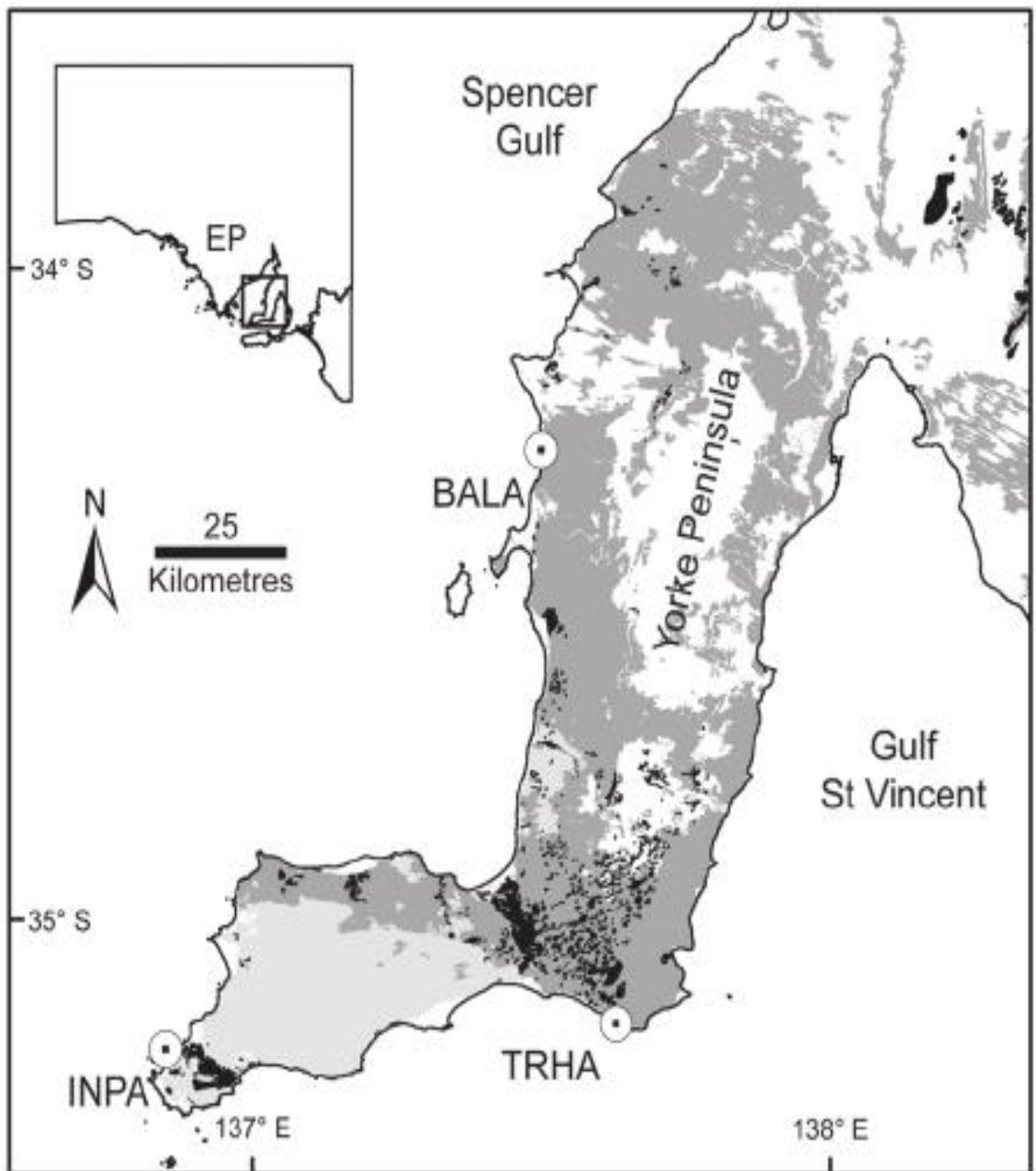


Figure 1.7: Pre-European vegetation and geomorphology of Yorke Peninsula, South Australia. Ephemeral salt lakes (black); dryland tea-tree – drooping she-oak low open woodland on calcrete (dark grey); mallee on sand (pale grey) and other vegetation types (white). The location of the Balgowan (BALA), Innes National Park (INPA) and Troubridge Hill (TRHA) fossil sites are indicated (⊙). Modified from McDowell et al. (2012).

Table 1.1: The body mass, ecosystem function, conservation status and occurrence in two habitats of the thirty species of non-volant, terrestrial mammal species present on Yorke Peninsula at the time of European settlement, based on Aitken (1973), Brandle (2008), McDowell et al. (2012) and Travouillon and Phillips (2018). Species under consideration for reintroduction are indicated in bold (see Taggart 2014).

Species	Family	Mass (g) ¹	Ecosystem function ²	Conservation status	Mallee on sand	Woodland on calcrete
<i>Tachyglossus aculeatus</i>	Tachyglossidae	4,000	Digging insectivore	Extant		✓
<i>Dasyurus geoffroi</i>	Dasyuridae	1,500	Terrestrial predator	Locally extinct		✓
<i>Phascogale calura</i>	Dasyuridae	70	Semi-arboreal predator	Locally extinct	✓	✓
<i>Sminthopsis griseoventer</i>	Dasyuridae	25	Terrestrial predator	Locally extinct	✓	
<i>Sminthopsis crassicaudata</i>	Dasyuridae	20	Terrestrial predator	Locally extinct	✓	✓
<i>Sminthopsis dolichura</i>	Dasyuridae	20	Terrestrial predator	Locally extinct	✓	✓
<i>Macrotis lagotis</i>	Peramelidae	2,000	Terrestrial omnivore	Locally extinct		
<i>Isodon obesulus</i>	Peramelidae	1,000	Terrestrial omnivore	Locally extinct	✓	✓
<i>Perameles notina</i>	Peramelidae	285	Terrestrial omnivore	Globally extinct	✓	✓
<i>Trichosaurus vulpecula</i>	Phalangeridae	2,500	Arboreal omnivore	Extant		✓
<i>Pseudocheirus peregrinus</i>	Pseudocheiridae	900	Arboreal browser	Locally extinct		✓
<i>Cercatetus concinnus</i>	Burramyidae	18	Scansorial omnivore	Extant	✓	
<i>Macropus fuliginosus</i>	Macropidae	28,000	Grazing herbivore	Extant	✓	✓
<i>Macropus robustus</i>	Macropidae	25,000	Grazing herbivore	Extant		✓
<i>Macropus eugenii</i>	Macropidae	5,500	Grazing herbivore	Reintroduced	✓	✓
<i>Lagorchestes leporides</i>	Macropidae	3,200	?	Globally extinct		✓
<i>Lagastrophus fasciatus</i>	Macropidae	2,000	Terrestrial browser	Locally extinct		✓
<i>Bettongia penicillata</i>	Potoroidae	1,600	Digging mycivore	Locally extinct		✓
<i>Bettongia lesueur</i>	Potoroidae	1,600	Digging mycivore	Locally extinct		✓
<i>Potorous platyops</i>	Potoroidae	800	?	Globally extinct	✓	
<i>Lasiorninus latifrons</i>	Vombatidae	27,000	Grazing herbivore	Extant		✓
<i>Leporillus apicalis</i>	Muridae	150	Terrestrial frugivore	Extinct	✓	✓
<i>Rattus fuscipes</i>	Muridae	125	Terrestrial omnivore	Locally extinct	✓	✓
<i>Rattus tunneyi</i>	Muridae	130	Terrestrial omnivore	Locally extinct	✓	
<i>Notomys mitchelli</i>	Muridae	52	Terrestrial omnivore	Locally extinct	✓	✓
<i>Pseudomys shortridgei</i>	Muridae	70	Terrestrial omnivore	Locally extinct	✓	✓
<i>Pseudomys gouldii</i>	Muridae	50	?	Globally extinct	✓	✓
<i>Pseudomys australis</i>	Muridae	65	Terrestrial omnivore	Locally extinct	✓	✓
<i>Pseudomys occidentalis</i>	Muridae	45	Terrestrial omnivore	Locally extinct	✓	✓
<i>Canis familiaris</i>	Canidae	20,000	Terrestrial carnivore	Locally extinct		✓

¹Mass data from Menkhorst & Knight (2011). ²Ecosystem function from Menkhorst and Knight (2011) and Breed and Ford (2007).

A second key driver of ecosystem dynamics on Yorke Peninsula is rainfall (Figure 1.8), which determines the soil moisture available for plant growth, flower production, and recruitment (Morrison and Myerscough 1989, Lamont et al. 1991, Bradstock and Bedward 1992, Enright and Lamont 1992). The influence of moisture in mallee is evident across regional climatic gradients (Griffin et al. 1983, Cowling and Lamont 1985, Brown 1989, Keith et al. 2014) and across local topographic gradients (Hnatiuk and Hopkins 1981, Myerscough et al. 1996, Hyde 2001, Keith et al. 2014).

Fire is the third key natural driver of change in ecosystems on Yorke Peninsula (Figure 1.8). Fire is an important source of natural disturbance (Whelan 1995, Bond et al. 2005b). Natural fires are generally caused by lightning strikes, and temporarily remove above ground vegetation, and in so doing make available nutrients from plant canopies, increase soil moisture by reducing evapotranspiration and increase light levels and space at soil level, thereby creating gaps for plant recruitment (Whelan 1995). The occurrence of a fire instigates the development of a pyrric succession of ecological communities (Specht 1972, Noble et al. 1980). The frequency and spatial occurrence of fires results in a mosaic of communities at different successional stages, which is important to maintain biodiversity (Doherty et al. 2017b). Indeed, several threatened species rely on the availability of habitat patches of a particular burn history to survive (Bennett et al. 2010a).

The limited information available on the fire history on Yorke Peninsula has been summarised by (Taylor et al. 2014). The pre-European fire regime (the frequency, intensity and season of fire events) is not well understood. Following European settlement fire frequency likely increases as fire was used to clear vegetation. Subsequently emphasis has been placed on fire exclusion and control to protect infrastructure. Available evidence suggests that as little as 14% of the 77,781 ha of remnant vegetation on the SYP has experienced a fire since 1956 (Figure 1.9). Social demands for fire suppression are likely to keep fire frequency low (Taylor et al. 2014), but the intensity of fires is expected to increase under the influence of anthropogenic climate change (Cary 2002, Turco et al. 2018).

The frequency and intensity of fires in mallee communities regulate temporal variation in resource availability and ecosystem processes (Keith et al. 2002, Enright et al. 2012). Fire suppression has changed the temporal and spatial distribution of fire-driven successional communities (Berry et al. 2015, Bowman and Legge 2016). By altering vegetation structure (Specht 1972, Noble et al. 1980, Clarke et al. 2010, Gibson et al. 2016), fires influence the habitat suitability for a variety of fungi (Vernes et al. 2001), invertebrate (Andersen and Yen 1985, Smith and Morton 1990, Schlesinger et al. 1997, Swengel 2001, Gunawardene and Majer 2005, Langlands et al. 2006, Avitabile et al. 2015) and vertebrate fauna (Fox 1982, Friend 1993, Nimmo et al. 2012, Nimmo et al. 2014, Hale et al. 2016).

Bottom-up processes appear to predominate in mallee ecosystems. However, this generalisation may reflect the historical extinction of many native mammals in southern Australia (see section 1.3.5 native fauna), as top-down ecosystem processes elsewhere are well documented and understood from studies of mammals (Dickman 2014a, Glen and Dickman 2014b, Hollings et al. 2014, Fancourt et al. 2015). The loss of ecological processes associated with the missing mammal fauna in Australia has increased the threat of bushfires in Australia (Hayward et al. 2016). Rewilding has the potential to reduce the impact of fires (Johnson et al. 2018).

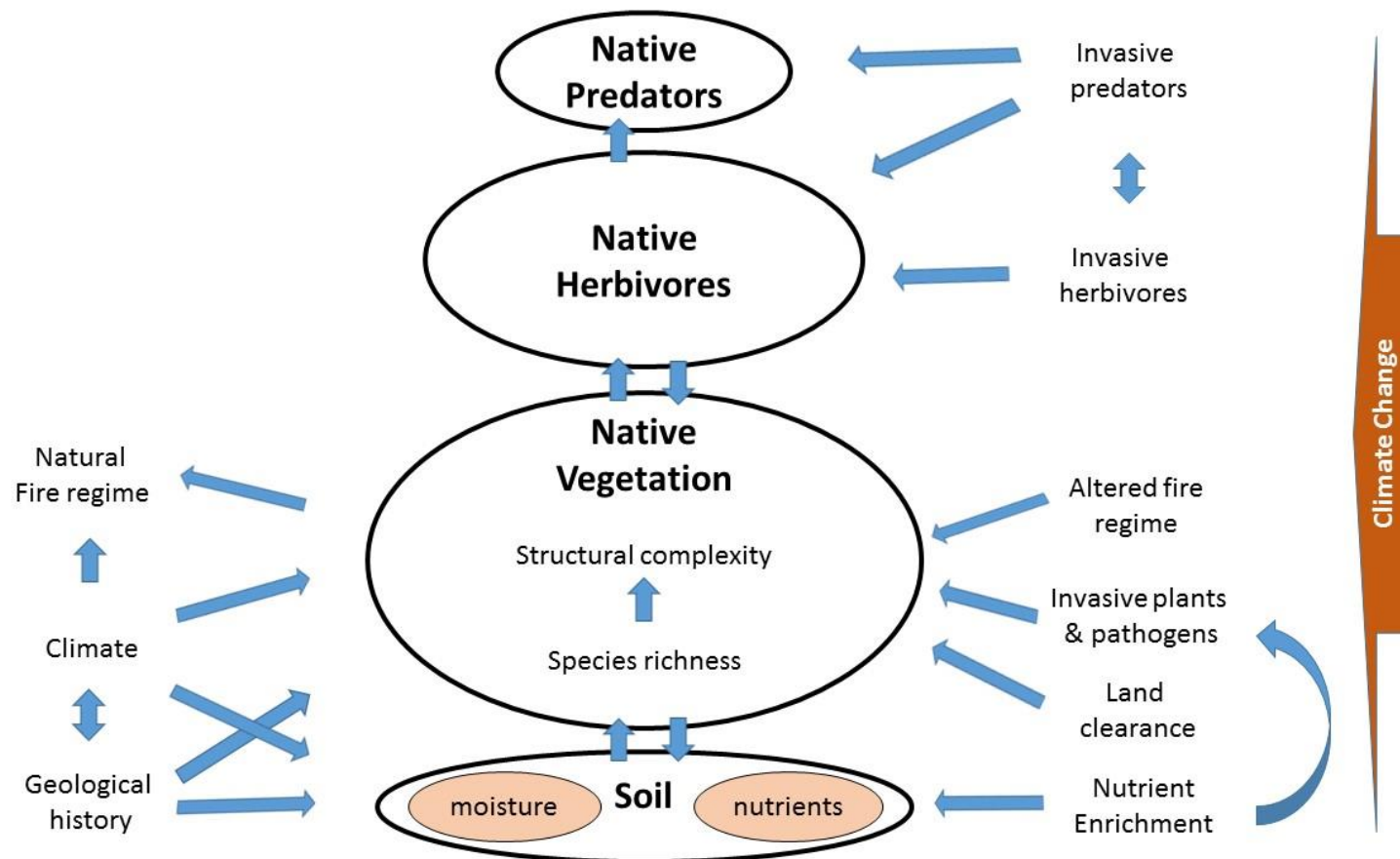


Figure 1.8: Conceptual model of ecosystem dynamics on Southern Yorke Peninsula (adapted from Keith et al., 2014). Ovals indicate key components of the physical and biological components of the natural environment. These are arranged in the form of an energy pyramid, where the nature of upper components rely on energy & nutrient transfer from lower components. Major natural factors that influence ecological processes are listed on the left, whereas human induced factors are listed on the right. Arrows indicate processes and interactions (not exhaustive) between ecosystem components and the factors that influences them.

1.3.6.2 Anthropogenic drivers of ecological processes

Land clearing is a major cause of biodiversity decline globally (Tilman et al. 2001). Southern Yorke Peninsula retains 18% native vegetation overall, with most located on the south-western 'toe' (Berkinshaw et al. 2017; Figure 1.9). Land clearance has resulted in fragmentation of native vegetation into patches that vary in size and distance from each other. Cleared land limits dispersal between remnant vegetation patches. Although small, isolated patches of vegetation can also be important for conservation (Wintle et al. 2018) they are subject to greater degradation around their edges and are able to support fewer native species than larger, less isolated patches (Diamond 1975). The speed with which native vegetation was cleared on Yorke Peninsula during the mid-20th century is likely to have resulted in an extinction debt. That is a mismatch between the number of species each remaining patch can support, and the number of species present when patch were formed followed by a delayed loss of species until an equilibrium species diversity is reached (Tilman et al. 2001). This extinction debt, together with deterioration of ecological processes in remaining habitat patches may explain the continued loss of biodiversity across the agricultural regions of southern Australia (Ford and Howe 1980, Birdlife Australia 2015, Boulton and Lau 2015).

The characteristics that allow native plants to live on nutrient poor soils make them susceptible to nutrient enrichment (Shane et al. 2004, Lambers et al. 2010). In agricultural landscapes, nutrient enrichment occurs mainly through wind transport of fertilisers from nearby crops and pastures (Keith et al. 2014). Runoff from roads can be locally significant for roadside remnants. Aerial applications of crop fertilisers, herbicides, fire retardants and disease-treatment chemicals can result in incidental nutrient enrichment of nearby native vegetation remnants, resulting in changes to species composition due to toxicity effects on native species and weed invasion (Bell et al. 2005). Nutrient enrichment can also change activity patterns of some ant species which may affect seed dispersal and plant recruitment (Seymour and Collett 2009). Displacement of native plants by non-native weeds has trophic implications for ecosystems, especially through assemblages of host-dependent invertebrates (Wilkie et al. 2007).

Introduced plant pathogens pose a major threat to Australia's heathland biodiversity (Keith et al. 2014). The most important of these is cinnamon 'fungus' (*Phytophthora cinnamomi*; in fact, an introduced soil-borne, water mould) that kills plants after infecting their roots. Different species of native plants vary greatly in their susceptibility to this disease (Cahill et al. 2008). The community-level impacts include declines in species diversity, changes in competitive relationships and shifts in species composition, changes in vegetation structure and animal habitats, loss of food resources for nectarivores, and loss of plant hosts for dependent invertebrates and microorganisms (Garkaklis et al. 2004a, Keith et al. 2014). The disease has serious implications for biodiversity because many locally endemic plants with restricted ranges are susceptible (Shearer et al. 2007). Major infestations of *P. cinnamomi* occur in mesic parts of southern Australia (Cahill et al. 2008). Although conditions are suitable for cinnamon 'fungus' on southern Yorke Peninsula (Cahill et al. 2008), there are currently no known infestations (Dr Andy Sharp, personal communication, 2018).

Fire suppression and controlled burning to protect people, infrastructure and crops alters the fire regime at a landscape scale. Adverse fire regimes may lead to local extinctions and declines in biodiversity (Keith 1996, Boulton and Lau 2015). Different species require different fire regimes to maintain viable populations. For example, the nationally vulnerable plant *Swainsona pyrophila* is short lived and requires fire to stimulate germination of a residual soil seed bank (Tonkinson and Robertson 2010). In contrast,

mallee fowl and mallee whipbirds require long unburned habitat patches (Woinarski et al. 1988, Benshemesh 1990, Cale and Mladovan 2008, Parsons and Gosper 2011). Thus, fire management for biodiversity requires specific knowledge to ensure the right balance of fire suppression and planned burning (Gill and McCarthy 1998, Clarke 2008, Flannigan et al. 2009, Bennett et al. 2010a, Driscoll et al. 2010, Haslem et al. 2011, Haslem et al. 2012). Inappropriate fire regimes are a significant threat to remnant biodiversity on Yorke Peninsula (Department for Environment and Heritage 2010, Taylor et al. 2014).

Exotic predators and herbivores have had a significant impact on Australian ecosystems (Rolls 1969, Low 2002). The impact and management of introduced, predatory foxes (*Vulpes vulpes*) and cats (*Felis catus*) is outlined in Chapter 2. Introduced herbivorous rabbits (*Oryctolagus cuniculus*) have had a significant impact on the environment, society and economy in Australia (Rolls 1969, Lange and Graham 1983, Bird et al. 2012, Cooke 2012, Mutze 2016, Mutze et al. 2016, Munday 2017). The impact and management of introduced mice (*Mus musculus*) is outlined in Chapter 4. Introduced invertebrate herbivores (e.g. the Mediterranean snails *Certhia virgata*, *Theba pisana* and *Cochlicella acuta*) also have a significant impact on agriculture (Baker 1986), but their impacts on ecosystem processes are largely unknown in Australia. They feed on crops during winter and spring and contaminate harvested grain when they aestivate on cereals leading to economic losses to growers (Baker 1986, Baker and Hawke 1990, Baker 2008). There is also the possibility that additional tourism attracted by the rewilding project will alter ecological processes on Yorke Peninsula if it is not appropriately managed (Dragovich and Bajpai 2012, Gogarty et al. 2018).

Underpinning all other anthropogenic influences on ecosystems on a landscape scale, is the influence of global climate change (Hughes 2003, Australian Academy of Sciences 2015, Environment Protection Authority 2018, International Intergovernmental Panel on Climate Change 2018). Climate change is expected to impact natural, economic and social aspects of southern Yorke Peninsula (Suppiah et al. 2006, Environment Protection Authority 2018). The predicted impacts of global temperature increases on sea level rise along the coast of Yorke Peninsula have been addressed by Australian Water Environments (2015). Yorke Peninsula is predicted to become warmer and drier in the future, and the available data support these predictions (Figures 1.3 & 1.4). These changes will affect soil moisture and agricultural growing seasons (Luo et al. 2005). Changes in atmospheric chemistry over the past four decades have resulted in reduced nitrogen availability in terrestrial ecosystems globally (Craine et al. 2018). This limits productivity of natural and unfertilised agricultural landscapes, carbon-uptake in terrestrial systems and increases bottom-up nutritional stress in ecosystems. It is not clear how this affects ecosystems which are already nutrient-limited. The frequency of extreme fire weather days has increased, resulting in more frequent and more extensive fires (Clarke et al. 2013).

Globally, anthropogenic climate change has had three major effects on biological systems: it has caused changes in geographic range (Parmesan and Yohe 2003, Thomas et al. 2004, Hoegh-Guldberg et al. 2008), flowering and breeding phenology (Stevenson and Bryant 2000, Parmesan and Yohe 2003, Pedersen et al. 2017), and body size (Gardner et al. 2011). Models of potential geographic ranges under the influence of climate change suggest substantial contraction in the geographic ranges for many species (Fitzpatrick et al. 2008, Yates and Ladd 2010, Delean et al. 2013). Different rates of change in phenology threaten to decouple mutual relationships between plants and their pollinators (Mommott et al. 2007). These changes can have far-reaching effects in ecosystems (Shackell and et al. 2010). Although our knowledge of ecosystem processes in Australia is limited,

observations from elsewhere suggest a range of interactions may be under threat (Hughes 2003, Cabrelli et al. 2014). The complex changes due to climate change should be taken into account when monitoring and interpreting changes due to management actions as part of the southern Yorke Peninsula rewilding project (Fordham et al. 2012, Jarvie and Svenning 2018).

1.4 Society

People have lived on southern Yorke Peninsula for approximately 50,000 years (Tobler et al. 2017). The aboriginal people have been on the peninsula throughout this period, but have been largely supplanted by non-indigenous people since the 1840s (Carmichael and Mudie 1973, Krichauff 2010).

The following information was obtained from the Australian Bureau of Statistics (Australian Bureau of Statistics 2018). In 2017 there were 11,286 people living in the Yorke Peninsula Local Government Area, 90.0% of whom were born in Australia, and 2.7% of whom identified as of aboriginal or Torres Strait Islander. The population density is 1 person per 1.9km². Of the total population, 1113 (9.9%) are children 1- 14 years old, 2522 (22.3%) are of reproductive age (15-44 years), and 7651 (67.8%) are older than 45 years. Women on Yorke Peninsula produce an average of 2.4 children. The median age of people on Yorke Peninsula is 55 years. Most identified as Christian (60.6%) and 29.3% had no religious affiliation. English is the language spoken in the home of 98.4% of the population. In terms of education, 0.2% of the population have no formal schooling, 18.9% attended school for less than nine years, 29.5% completed year twelve, and 7.3% attended university. Another 39.7% of the population have some other post school qualification (e.g. trade certificate). There are 4647 households on Yorke Peninsula, of which 1,591 (34%) contain a single occupant and 2,974 (64%) contain a family. The average household consisted of 2.1 people.

1.4.1 Indigenous people

The aboriginal people of Yorke Peninsula (the 'Narungga), were never the subject of detailed anthropological investigation. Consequently, published information of their traditional life is scant (Taplin 1879, Curr 1886, Sutton 1890, Howitt 1904, Tindale 1936, Berndt 1939, Campbell and Walsh 1947, Hill and Hill 1975). The Narungga nation was made up of four clans, the 'Kurnara in the north of the peninsula, 'Winderera in the east, 'Wari in the west and 'Dilpa in the south. Some 'Narungga names have been adopted and are still used by non-indigenous people. For example, the 'Narungga kangaroo hunting place 'Pandalawi, is now widely known as Ponda'lowie. Similarly the township of Wa'rooka's name is derived from the 'Narungga word 'Wiru'ka (Tindale 1936; Figure 1.10).

The 'Narungga were thought to number 500 in 1847 and to have been reduced to half that number by 1856 (Curr 1886). During that decade the 'Narungga forcefully resisted incursions into their country by non-indigenous settlers (Pope 1989, Krichauff 2010). By 1880 there were less than 100 'Narungga people (Curr 1886). Following its establishment in 1867, Point Pearce Mission Station became a source of food and health care for 'Narungga (Krichauff 2008). Although most 'Narungga chose not to, some who had converted to Christianity lived at the mission. Mission residents were joined by aboriginal people from Poonindie and Port Macleay, when those missions closed in the 1880s and 1890s. The history of Point Pearce Mission has been described elsewhere (Archibald 1915, Heinrich 1972, Hill and Hill 1975, Graham and Graham 1987, Wanganeen 1987, Krichauff 2008, 2010).

Three hundred and five people on Yorke Peninsula currently identify as aboriginal or Torres Strait Islander (Australian Bureau of Statistics 2018). These include descendants of the 'Narungga. Over the past decade considerable efforts have been made to document and recover 'Narungga history and culture (Narungga Aboriginal Progress Association 2006, Eira and Narungga Aboriginal Progress Association 2010, Krichauff 2010, Narungga Aboriginal Progress Association 2010).

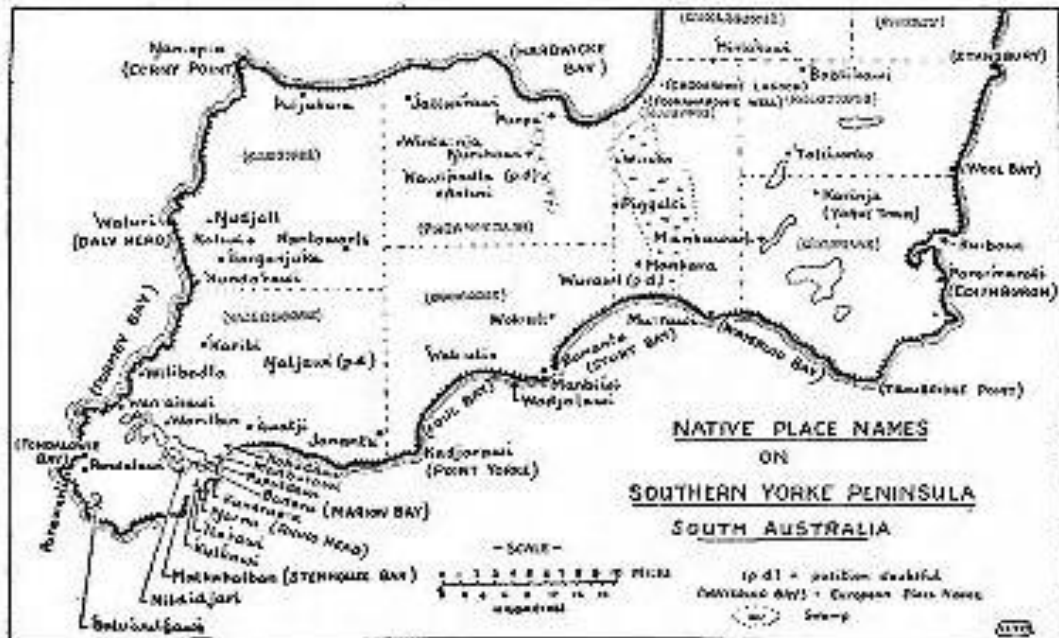


Figure 1.10: Map showing 'Narungga place names on southern Yorke Peninsula, from Tindale (1936).

1.4.2 Non-indigenous people

The non-indigenous influence on Yorke Peninsula commenced in 1802, with the expeditions of Flinders and Baudin (Peron and Freycinet 1807, Matthew 1814, Baudin 1974). Between 1803 and 1836 sealers and whalers frequently visited the shores of St Vincent and Spencer Gulfs from Kangaroo Island (Jameson 1842, Cumpston 1974). A few European adventurers, surveyors and speculators visited 'Narungga country between 1936 and the commencement of pastoral development there in 1946 (Krichauff 2008). By 1879 all of Yorke Peninsula had been surveyed and sold for agricultural settlement (Krichauff 2008). The subsequent settlement of Yorke Peninsula has been ably described elsewhere (Carmichael and Mudie 1973, Cook 1975). Livestock grazing (predominantly sheep) began in 1851 near Stansbury and extended over virtually the entire 'foot' of the peninsula and up the western side of the 'leg' by 1865. Extensive land clearance for agricultural crops occurred after 1927, with advances in agriculture relating to an improved understanding of trace elements and the development of fertilisers (Berkinshaw et al. 2017).

Of the 11,286 people living on Yorke Peninsula, 54.5% are of working age (15-64 years). The work participation rate is 45.2% and 5.7% are unemployed. Twenty seven percent of employees work in agriculture and fishing. The median wage is \$AU 32,749 per annum (Australian Bureau of Statistics 2018).

1.5 Economy

Like many regional areas, Yorke Peninsula is in slow demographic and economic decline. Notwithstanding this, southern Yorke Peninsula has important agricultural and tourism industries. Forty five percent of the 1,309 businesses on Yorke Peninsula are involved in agriculture. Tourists are attracted to coastal towns and Innes National Park, and other nearby conservation parks. Rewilding is expected to appeal to the tourism industry because of the breadth of experiences it will provide through interactions with wildlife. Eco-tourism opportunities linked to the rewilding of southern Yorke Peninsula will drive an increase in the share of the South Australian tourism market and provide the base from which a national and international tourism industry can be developed. The project feeds in to current State Government policy to assess the feasibility of an eco-tourism resort within Innes National Park, by providing an additional incentive to visit the resort.

The southern Yorke Peninsula rewilding project is expected to provide considerable economic benefits in the local area and also to the State of South Australia (Bailey et al. 2018). Conservatively, the southern Yorke Peninsula rewilding project is predicted to offer a 10% increase in return for allocation of public money compared to current practices. The added economic returns will come from additional program staff, investment in additional infrastructure, decreased costs and increased yield in agriculture, and increased visits by tourists. In the first year, the expected impact on total Gross South Australian Product will be \$0.5 million per year and is expected to increase to \$2.6 million in the twentieth year. The project is expected to provide the equivalent of four new full-time jobs in the first year and rise to thirteen in the twentieth year.

1.6 Progress to date

In the absence of the ecological functions of 24 locally extinct native mammal species, the area's native vegetation communities are undergoing a slow deterioration in condition. Forty one percent of native plant species are in decline in the Northern and Yorke Natural Resource Management Region (Gillam and Urban 2008). The rewilding of southern Yorke Peninsula aims to re-establish missing ecological processes by reintroducing keystone mammal species. The project builds upon a previously reintroduced population of South Australian mainland tammar wallabies (*Notamacropus eugenii*) and baiting established to control foxes to support biodiversity and agriculture. These isolated activities are a firm basis upon which to develop an integrated rewilding program (Tanentzap and Smith 2018).

1.6.1 Reintroduction of Tammar Wallabies

Tammar wallabies became extinct on the South Australian mainland in the 1930s due to clearance of its preferred habitat for agriculture and predation by red foxes (Jones 1924, Department for Environment and Heritage 2004). Before going extinct in their native range, a feral population of tammar wallabies from mainland South Australia was established in New Zealand (Poole et al. 1991, Taylor and Cooper 1999). Tammar wallabies from New Zealand were reintroduced to Innes National Park between 2004 and 2008 (Sharp et al. 2010, Kemp 2011). Despite initial opposition from some sections of the local community (Peace 2009), the reintroduction is now generally regarded positively (Department of Environment and Natural Resources 2011). Following the release of 125 wallabies,

they have bred for several generations in the wild and dispersed beyond the initial release site (Sharp et al. 2010, Department of Environment and Natural Resources 2011, Kemp 2011). The wallabies have established a self-sustaining population and offer an attraction for tourists visiting Innes National Park. The wallabies at Innes represent a population that had become extinct in the wild and has now been returned to its former range (Sharp et al. 2010, Department of Environment and Natural Resources 2011, Kemp 2011). As a result of new understanding of their genetic relationships (Eldridge et al. 2017b) and the reintroduction to Yorke Peninsula, SA mainland tammar wallabies are no longer regarded as endangered (Burbidge and Woinarski 2016b). This was significant achievement for biodiversity conservation.

1.6.2 ‘Baiting for Biodiversity’

Reducing the threat of fox predation was an important precondition to reintroducing tammar wallabies to Yorke Peninsula (Department for Environment and Heritage 2004). An intensive fox control program began on Innes National Park in 2003 and continues to date. Passive tracking stations (Engeman and Allen 2000) demonstrated a significant reduction in fox abundance on the park between 2003 and 2007 (Sharp et al. 2010). In 2006, a community-based fox control program was initiated on lands surrounding Innes National Park, to reduce the immigration rate of foxes into the park and thereby lower the intensity of predator control required on the park. By 2008-2009, 24 landholders were participating in the program, covering approximately 60,000 ha (Sharp et al. 2010). The Northern and York Natural Resources Management Board has been supporting a community-driven fox control program (called 'Baiting for Biodiversity') across 170,000 Ha on southern Yorke Peninsula since 2012. These baiting programs have been followed by the re-appearance of bush stone curlew (*Burhinus grallarius*) after 40 years of no sightings, and echidnas after 20 years. They have also resulted in stable or increasing populations of malleefowl, hooded plovers and Rosenberg's goannas (Andy Sharp, pers comm 2018). Baiting for Biodiversity program resulted in a 30% increase in lamb survival (Max Barr - personal communication, 2018).

1.6.3 The next steps

In 2015/16, a consortium of nine environmental NGOs, local councils and community groups developed the Rewilding Yorke Peninsula concept (Sharp 2018). The project is led by the Northern and Yorke Natural Resources Management Board, WWF Australia, Zoos SA and the FAUNA Research Alliance.

The Rewilding of Southern Yorke Peninsula will include: (1) a program of integrated management of invasive animals, (2) a threatened species recovery program, (3) a sustainable agriculture program, and (4) an economic renewal program and is envisaged as a 25 year project (Sharp 2018). The integrated pest management program will build on the established fox baiting to include control of cats and mice, all important threats to biodiversity and agriculture (see Chapters 2 - 4). Staged reintroductions of keystone native mammals to the landscape will seek to re-establish ecological function (Table 1.2) to forestall the decline of the district's ecological communities and to facilitate improvements in the resilience and adaptability into the system, in the face of shifting climatic conditions.

It is anticipated that the reintroduction of native *soil engineers* will increase nutrient turnover within soils, improve water infiltration and soil moisture, facilitate the dispersal of mycorrhizal fungi across the landscape, and create micro-habitat conditions for the germination and establishment of native plant seedlings (see Chapter 5). The reintroduction of *native rodents* will increase levels of seed dispersal and germination of native plants (Chapter 6). Returning *native predators* to the system is anticipated to reduce the abundance of vertebrate pest species, including house mice (red-tailed phascogale, barn owl, western quoll), rabbits (western quoll) and potentially feral cats (Tasmanian Devil; Chapter 7).

Table 1.2: Proposed reintroduction, augmentations and translocations included in the southern Yorke Peninsula rewilding project outlined by Sharp (2018).

Project phase	Species	Ecological function
1: 2019 – 23	brush-tailed bettong	soil engineer, disperser of fungi
	southern brown bandicoot	soil engineer
	red-tailed phascogale	small native predator
	augment barn owl population	small native predator
2: 2024 – 28	native rodents	seed dispersal
3: 2029 – 33	western quoll	medium-sized native predator
4: 2034 – 38	Tasmanian devil	medium-sized native predator

The rewilding of southern Yorke Peninsula is a proof-of-concept project that seeks to document the many potential benefits of rewilding (Sharp 2018). Therefore, a significant focus has been placed on monitoring and evaluation to enable the detection and assessment of any positive (or negative) effects that result from the project (Sharp 2018). The remainder of this report reviews the scientific evidence underpinning the Rewilding Yorke Peninsula project to identify important knowledge gaps, risks, and what factors should be the focus of monitoring for adaptive management through the life of the project.

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2 Reducing invasive predators

Invasive species were a cause of more than one-half of all recent species extinctions globally (Clavero and Garcia-Berthou 2005). They also have significant negative impacts on humans through economic losses, costs of management, reduced ecosystem services, and negative effects on health and well-being (Pimentel et al. 2005, Olson 2006, Pejchar and Mooney 2009).

Predation by introduced mammals has led to population declines of many species around the world. The influence of introduced predators on oceanic islands is well documented (Blackburn et al. 2004), and they can also have strong effects on mainland populations (Salo et al. 2007). Introduced predators have significantly greater negative effects on native prey species than native predators do (Salo et al. 2007). Australia's unenviable record as the site of one-third of the world's mammalian extinctions in historical times is due largely to the impact of introduced European red foxes (*Vulpes vulpes*) and domestic cats (*Felis catus*) (Burbidge and McKenzie 1989b, Johnson 2006a, Abbott et al. 2014, Fisher et al. 2014, Woinarski et al. 2015). Many mammals between 35 g and 5500 g in weight (the 'critical weight range') have been extirpated from large areas of Australia by these invasive predators (Burbidge and McKenzie 1989b, Johnson and Isaac 2009, Woinarski et al. 2014b). Indeed, threatened species management in Australia is frequently reliant on the control or eradication of introduced mammalian predators (Orell and Morris 1994, Burbidge and Morris 2002, Moseby et al. 2011, Morris et al. 2015, Sharp et al. 2015, Brandle et al. 2018, Legge et al. 2018b, Radford et al. 2018, Ringma et al. 2018, Wayne 2018). Introduced predators have had greater impacts in Australia than on any other continent (Salo et al. 2007).

Predation by the introduced European red foxes and feral cats are recognised as key threatening processes under the Australian Environment Protection and Biodiversity Conservation Act 1999. Key threatening processes threaten the survival, abundance or evolutionary development of a native species and/or ecological communities (DEWHA 2008, Department for the Environment 2015).

Fifteen of the thirty species of non-volant, terrestrial mammals that occurred on southern Yorke Peninsula at the time of European settlement are Highly or Extremely susceptible to predation by foxes and/or feral cats (Radford et al. 2018; Table 2.1). Yorke Peninsula is part of the Eyre Yorke Block, that has been identified as a priority area for strategic placement of a fox and cat refuge that would contribute to global protection of eight of these species (Ringma et al. 2019; Table 2.1).

The available evidence suggests that reducing the density (or complete removal) of foxes and cats from southern Yorke Peninsula would offer considerable benefits to biodiversity and agriculture in the area and contribute globally to conservation of Australian endemic mammals. The Southern Yorke Peninsula Rewilding project aims to expand the existing 'baiting for biodiversity' program for fox control (Chapter 1) to include feral cat control, and reduce dispersal of foxes and cats onto southern Yorke Peninsula from the north with an exclusion fence (see Chapter 2)(Sharp 2018). Figure 2.1 summarises the interactions that are expected to occur in response to reducing fox and cat population density of Yorke Peninsula.

Table 2.1: Local status on southern Yorke Peninsula, Global Conservation status, susceptibility to fox and cat predation, and whether a predator free refuge on Yorke Peninsula would contribute toward global protection of species susceptible to introduced predators for the thirty species of non-volant, terrestrial mammal species present on Yorke Peninsula at the time of European settlement.

Scientific name ¹	Common name	SYP Status	Global Conservation status ²	Susceptibility to fox and cat predation ³	Eyre Yorke Block Contribution ⁴
<i>Tachyglossus aculeatus</i>	echidna	extant	Least Concern	Low	
<i>Dasyurus geoffroii</i>	western Quoll	extinct	Near Threatened	High	✓
<i>Phascogale calura</i>	red-tailed phascogale	extinct	Near Threatened	High	
<i>Sminthopsis griseoventer</i>	Grey-bellied dunnart	extinct	Least Concern	Low	
<i>Sminthopsis crassicaudata</i>	Fat-tailed dunnart	extinct	Least Concern	Low	
<i>Sminthopsis dolichura</i>	Little long-tailed dunnart	extinct	Least Concern	Low	
<i>Macrotis lagotis</i>	bilby	extinct	Vulnerable	High	✓
<i>Isodon obesulus</i>	southern brown bandicoot	extinct	Near Threatened	High	✓
<i>Perameles notina</i>	South Australian striped bandicoot	extinct	Extinct	Extreme	
<i>Trichosaurus vulpecula</i>	brush-tailed possum	extant	Least Concern	Low	
<i>Pseudocheirus peregrinus</i>	common ring-tailed possum	extinct	Least Concern	Low	
<i>Cercartetus concinnus</i>	western pygmy possum	extant	Least Concern	Low	
<i>Macropus fuliginosus</i>	western grey kangaroo	extant	Least Concern	Not Susceptible	
<i>Macropus robustus</i>	euro	extant	Least Concern	Not Susceptible	
<i>Macropus eugenii</i>	tammar wallaby	extant	Least Concern	High	
<i>Lagorchestes leporides</i>	eastern hare wallaby	extinct	Extinct	Extreme	
<i>Lagastrophus fasciatus</i>	banded hare wallaby	extinct	Vulnerable	Extreme	✓
<i>Bettongia penicillata</i>	brush-tailed bettong	extinct	Critically Endangered	High	✓
<i>Bettongia lesueur</i>	burrowing bettong	extinct	Near Threatened	Extreme	
<i>Potorous platyops</i>	broad-faced potoroo	extinct	Extinct	Extreme	
<i>Lasiorhinus latifrons</i>	southern hairy-nosed wombat	extant	Near Threatened	Not Susceptible	
<i>Leporillus apicalis</i>	lesser stick-nest rat	extinct	Extinct	Extreme	
<i>Rattus fuscipes</i>	bush rat	extinct	Least Concern	Low	
<i>Rattus tunneyi</i>	pale field rat	extinct	Least Concern	High	✓
<i>Notomys mitchelli</i>	Mitchell's hopping mouse	extinct	Least Concern	Low	
<i>Pseudomys shortridgei</i>	heath mouse	extinct	Near Threatened	High	✓
<i>Pseudomys gouldii</i>	Gould's mouse	extinct	Extinct	Extreme	
<i>Pseudomys australis</i>	plains mouse	extinct	Vulnerable	High	✓
<i>Pseudomys occidentalis</i>	western mouse	extinct	Least Concern	Low	
<i>Canis familiaris</i>	dingo	extinct	Least Concern	Not susceptible	

¹ Taxonomy follows Menkhurst and Knight (2011) and Travouillon and Phillips (2018) ² Radford et al. (2018); ³ Radford et al. (2018) ; ⁴ Ringma et al. (2019).

2.1 Red fox

Red foxes (*Vulpes vulpes*) occupy the largest natural range of any terrestrial wild mammal, occurring naturally throughout North America, Eurasia and the Middle-East (Larivière and Pasitschniak-Arts 1996, Schipper et al. 2008, Sacks et al. 2018). They were introduced to Australia from United Kingdom for recreational hunting (Rolls 1969, Long 2003). Several attempts to reintroduce foxes between 1845 and 1865 failed, but they became established in the Werribee–Geelong district of southern Victoria in the 1870s (Long 2003). From there they spread rapidly throughout the southern three quarters of the continent (Jarman 1986). Foxes were first recorded on Yorke Peninsula in 1897 and followed closely after the arrival of rabbits in 1884 (Copley et al. 1984).

2.1.1 Reproduction and population ecology

Foxes breed seasonally (McIntosh 1963, Lloyd 1980). In males the testes enlarge during April-August and females in oestrus occur from June-August (McIntosh 1963). Most pregnancies are completed by the end of September and lactation by early December (McIntosh 1963, Ryan 1976). The single oestrus lasts 1-6 days and females are receptive for up to 4 days (Lloyd 1980). Gestations lasts 51-52 days and (McIntosh 1963). Females give birth to between one and twelve cubs (more usually 4 – 6) in a breeding burrow or den (Lloyd 1980, Carter et al. 2012b). If a den is disturbed litters may be moved to a new den (MacDonald 1980). Cubs venture out of the den from four weeks of age and are weaned in six to ten weeks when they weigh ~ 2kg (Lloyd 1980). Foxes mature at 10 months of age, and few exceed seven years of age (Jarman 1986). Population turnover is rapid with 20-80% of foxes dying annually (Lloyd 1980, Jarman 1986).

Maturing cubs disperse in autumn and winter, coincident with the autumn increase in testis size of adults and juveniles (Storm and Montgomery 1975). Male cubs leave the natal territory unless the dominant male dies, but not all females disperse. Males move sooner and further than females in their first year (Ables 1969, 1975). Males disperse 4.6 - 41.8 km, whereas females disperse 1.9 - 21.4 km (Ables 1969, 1975, Lloyd 1975, Jarman 1986). Dispersing males move 15 km.night⁻¹ whereas dispersing females move 9 km.night⁻¹ (Storm and Montgomery 1975, Storm et al. 1976). Male red foxes are larger than females (mean \pm standard deviation = 6.3 \pm 0.77 kg for males and 5.5 \pm 0.72 kg for females)(McIntosh 1963).

In Australia population densities of foxes range from 0.2 to 7.2 individuals.km² in rangelands, remnant vegetation and temperate grazing areas, up to 12 individuals.km² in suburban Melbourne (Saunders et al. 1995). The social organization and home range area covary with population density and resource distribution. Solitary individuals occupy ranges of up to 50 km² in areas where population density is low, and family groups of six to seven members occupy shared ranges of three to seven square kilometres where population density is higher (Bubela 1995, Mahon 1999, Saunders et al. 2002, Moseby et al. 2009b, Carter et al. 2012a, Newsome et al. 2017). Foxes occur as pairs in low density populations or in groups of one male and two or more closely related females in high density populations (Lloyd 1980, Jarman 1986). In groups there is a strong dominance hierarchy among females. Only the dominant one or two vixens produce cubs, but subordinate vixens may breed if a dominant vixen dies (Englund 1970, MacDonald 1980). Non-breeding subordinate vixens do not suckle cubs, but groom, play with and bring food to the group's cubs (MacDonald 1979, MacDonald 1980). Between four and 88% of female

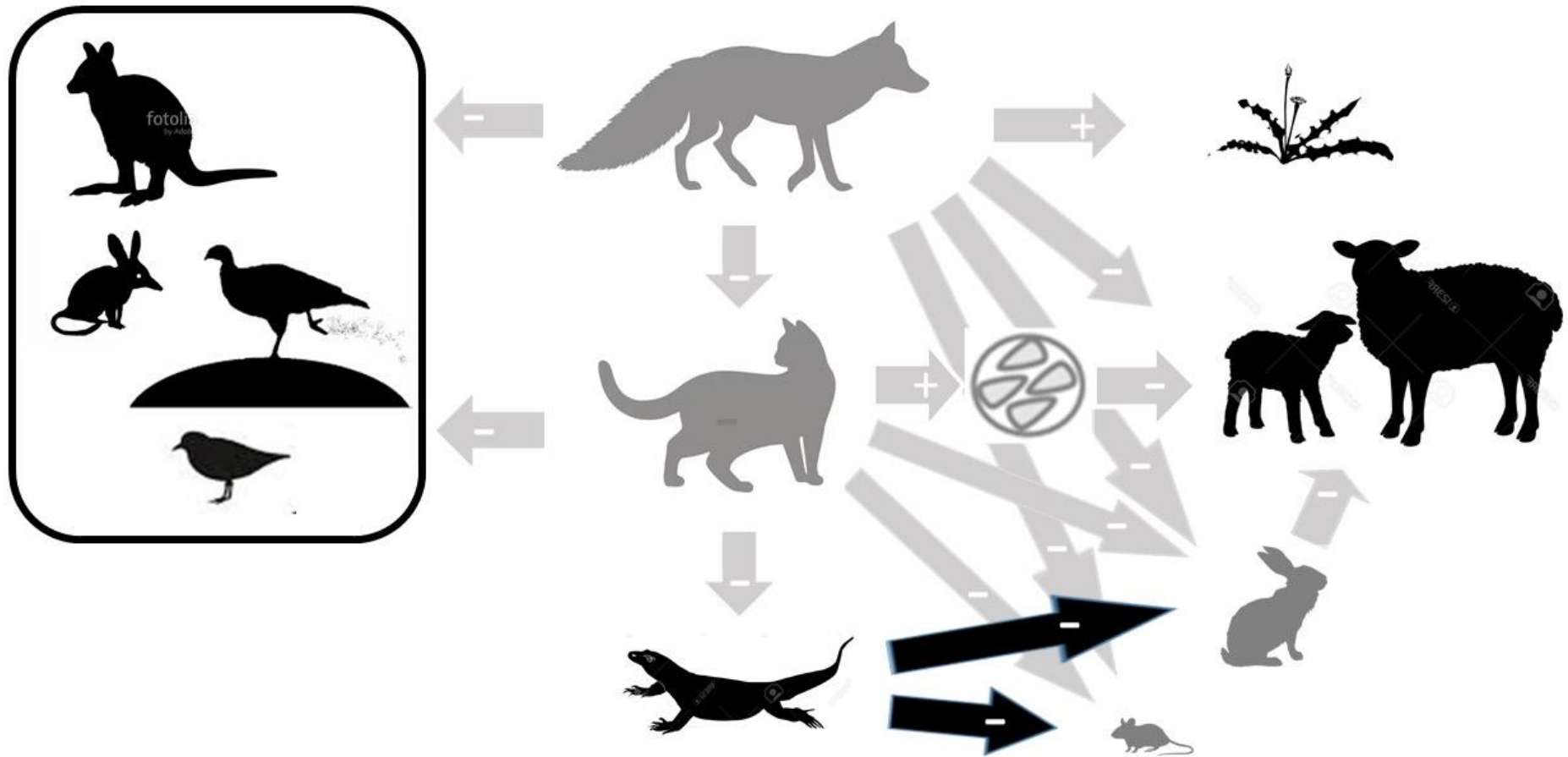


Figure 2.1: Hypothesized impact of reducing the density of foxes and feral cats on southern Yorke Peninsula. Arrows indicate ecological effects between species. Organisms that will decrease in abundance are shown in grey. Organisms that will increase in abundance are shown in black.

foxes may be non-breeders, depending on the population density (Lloyd 1975, Ryan 1976, England 1980, Kolb and Hewson 1980), suggesting negative density-dependent reproduction. The male may bring food to females for a week or two after birth of the cubs but spends little time in the company of other group members (Jarman 1986).

Foxes hunt solitarily and are highly mobile, often travelling more than ten kilometres per day (Larivière and Pasitschniak-Arts 1996, Meek and Saunders 2000, Carter et al. 2012a, Towerton et al. 2016). Foxes live in a wide range of habitats from arid deserts to wet sclerophyll forest (Jarman 1986, Menkhorst and Knight 2011). They are generalist predators, consuming a broad range of vertebrates, invertebrates, carrion, leaves, fruit and berries (Norman 1971, Bayly 1978, Lloyd 1980, Hart 1994, Banks et al. 2000, Greentree et al. 2000, Lapidge and Henshall 2001, Read and Bowen 2001, Paltridge 2002, Saunders et al. 2004, Heywood et al. 2008, Cupples et al. 2011, Glen et al. 2011, Marlow et al. 2014, Fleming et al. 2016, Johnston 2016, Nielsen and Bull 2016, Forbes-Harper et al. 2017). Foxes are usually opportunistic predators, but they are also able to selectively prey upon particular species, or types of food (Lloyd 1980, Dickman 1988, Green 2002).

2.1.2 Impact on native species and ecosystems

The strong negative effects of foxes on native mammals were apparent to early observers (Hoy 1923, Jones 1925, Troughton 1941). In South Australia Jones (1925, p. 358) described foxes as ‘. . . the most baneful disturbing influence brought about by the human folly of introducing animals into a new country.’

Throughout Australia the arrival of foxes at a locality usually coincided with the decline and disappearance of small and medium-sized mammals and ground-dwelling birds (Finlayson 1936, White 1952, Finlayson 1961, Jenkins 1974, King and Smith 1985, Short 1998, Risbey et al. 1999, Ford et al. 2001, Abbott 2011, Abbott et al. 2014). Studies of Australian offshore islands and their wildlife in the presence and absence of foxes strongly implicate foxes in the extinction or decline of many species (Norman 1971, Burbidge and McKenzie 1989b, Burbidge et al. 1997, Abbott 2000, Gates and Paton 2005, Johnston 2016, Burbidge and Abbott 2017, Legge et al. 2018b). Species that were once widespread and common are now extinct or rare on the Australian mainland, yet still survive on offshore islands not colonized by foxes (Burbidge and Manly 2002). Reintroduction programs frequently fail because of fox predation (Short et al. 1992, Priddel et al. 2007, Short 2009, Finlayson et al. 2010, Morris et al. 2015, Moseby et al. 2015a). Management programs in which foxes have been removed from specific areas allowed the recovery of potential prey species, including reptiles, birds and mammals (Morris et al. 1995, Risbey et al. 1999, Burbidge and Morris 2002, Moseby et al. 2011, Shorthouse et al. 2012, Morris et al. 2015, Wayne et al. 2017a, Brandle et al. 2018, Legge et al. 2018b, Wayne 2018). These observations strongly suggest that foxes have strong negative effects on native fauna but cannot rule out alternative explanations (Abbott 2006, Saunders et al. 2010). Experimental removal of foxes induced no response in some potential prey, including lizards (Mahon 1999, Risbey et al. 2000), birds (Davey et al. 2006) and mammals (Banks 1999). However, most field experiments have confirmed that foxes suppress a wide variety of prey, from small lizards to kangaroos (Kinnear et al. 1988, Banks et al. 1998, Kinnear et al. 1998, Mahon 1999, Banks et al. 2000, Kinnear et al. 2002, Spencer 2002, Olsson et al. 2005, Spencer et al. 2006, Dexter and Murray 2009, Kinnear et al. 2010).

In addition to directly consuming them as prey, foxes can also influence other species through competition and transmission of disease. Extensive overlap in resource use occurs between foxes and spotted-tailed quolls *Dasyurus maculatus* (Glen and Dickman 2011). Increases in populations of

western quolls following removal of foxes (Morris et al. 2003) have resulted from some combination of intraguild predation and competition (Saunders et al. 2010). There also appears to be competitive interactions between foxes and invasive cats (see section 2.3.1), and possibly goannas (see section 2.3.2).

Foxes carry a wide range of viral, bacterial and protozoan parasites, but their helminths and ectoparasites have most potential to affect native fauna (Saunders et al. 2010). The endoparasitic roundworm *Toxocara canis* and tapeworm *Spirometra erinaceid* are common (> 30% prevalence) in fox populations in Australia (Newsome and Coman 1989) and have also been recorded in many species of native mammals, snakes and frogs (Reddacliff and Spielman 1990). Among ectoparasites, the mite *Sarcoptes scabiei* has been passed from foxes to native mammals (Vogelnest et al. 2008, Fraser et al. 2019). This mite causes sarcoptic mange: severe itching, excoriation and skin inflammation, leading to loss of hair. In wombats, mange is debilitating and often fatal (Ruykys et al. 2009, Ruykys et al. 2013, Old et al. 2018, Fraser et al. 2019).

In addition to more obvious direct effects of foxes through predation, competition and disease transmission, foxes can also have less obvious indirect effects on species and ecosystem processes (Saunders et al. 2010). For example, depletion of herbivore populations by foxes ‘releases’ vegetation from grazing pressure, and so facilitates populations of preferred forage species (Newsome et al. 1997, Morris and Letnic 2017). Foxes are important dispersers of seeds of several exotic and invasive plants (Twigg et al. 2009), which results in a cascade of pervasive indirect effects on ecological communities (Low 2002, Murray et al. 2007; section 1.3.6.2; Figure 1.8). In contrast, recruitment of native plants is expected to decline when pollinators and soil-engineers decline due to fox predation (Dickman 2006b; chapter 5).

2.1.3 Impact on agriculture

Foxes are a significant agricultural pest because they prey on livestock, particularly sheep (Lugton 1993, Molsher et al. 2000). Sheep and lamb constitute between 20 and 40% of the diet of foxes by volume (Coman 1973, Croft and Hone 1978, Saunders et al. 2004). However, the role of the fox as a predator of otherwise viable lambs has been the subject of controversy (Saunders et al. 1995), principally due to difficulties in separating the interacting effects of fox predation and death of lambs from other causes (e.g. starvation due to poor maternal care; Dennis 1969, Rowley 1970). Foxes undoubtedly kill many lambs that would not have survived otherwise, but predation can also be a primary cause of lamb losses (Smith 1964, Turner 1965, McDonald 1966). For example, almost half of 3000 lamb carcasses were mutilated by predators, but a maximum of ten percent died primarily due to predator attack (McFarlane 1964). The proportion of lambs killed by foxes that would otherwise have survived varies between 3 and 30% (McFarlane 1964, Turner 1965, Moore et al. 1966, Dennis 1969, Lugton 1993, Greentree et al. 2000). Properties that took part in the ‘baiting for biodiversity’ program of fox control of Yorke Peninsula saw a 30% increase in lamb survival (Sharp 2018), showing that lambs lost to fox predation on Yorke Peninsula occurs at the maximum rate previously recorded.

2.1.4 Control methods

Foxes presently inhabit an estimated 5.8 million km², 76% of Australia (DEWHA 2008). There are few techniques that could be applied at a continental scale to eradicate foxes. Historically, attempts

to eradicate foxes have involved shooting, trapping, den fumigation, den destruction, exclusion fencing and lethal baiting, with varying degrees of success, over relatively small areas (Saunders et al. 2010). Fertility control through immune-contraception has been investigated but has been abandoned for technical and logistical reasons (McLeod et al. 2007, Saunders and McLeod 2007). Potential chemical fertility control agents are being explored, but debate remains about their efficacy, target specificity, welfare and cost-effectiveness for broadscale use (Saunders and McLeod 2007). Attempts to find a biological control agent to control foxes have been unsuccessful (Strive et al. 2007). Using gene editing techniques to control invasive animals is currently an active area of research, but we are in the early days of this developing technology (Moro et al. 2018). Currently, lethal baiting is the most effective and widely used method of fox control (Saunders et al. 1995, Saunders et al. 2010), and is the basis of several landscape scale fox control programs in Australia (Armstrong 2004, Marlow et al. 2015b, Brandle et al. 2018, Wayne 2018). The 'Baiting for Biodiversity' fox control program on Yorke Peninsula is based on this technique (Sharp 2018).

2.2 Cats

Archaeological and genetic evidence suggest that all domestic cats were derived from the Near Eastern wildcat (*Felis silvestris lybica*) (Driscoll et al. 2007). Domestication occurred around 10,000 years ago as agriculture developed in the Fertile Crescent (Baca et al. 2018). Domestic cats spread into Europe as commensal animals in two waves, 4000 years ago with the Neolithic farmers and then 2000 years ago with Greek and Phoenician traders (Baca et al. 2018). As a consequence of human introductions, domestic cats now have a near-global distribution, occurring on all continents except Antarctica and on hundreds of islands (Long 2003).

Cats first arrived in Australia at Sydney in 1788 as companion animals transported from England and subsequently spread with Europeans as they colonised the country (Abbott 2002, Abbott 2008b). Cats became established throughout Australia within 70 years (Abbott 2008b, Koch et al. 2015, Spencer et al. 2015). This rapid expansion was facilitated by the release and spread of rabbits, which provided abundant prey across much of the continent (Abbott 2008b). Cats were also spread in attempts to control rabbits, mice, or native species (Abbott 2008b). Although there are no specific records of cats in South Australia prior to 1844 (Abbott 2002, Abbott 2008b), cats probably arrived at Adelaide in 1836 and reached Yorke Peninsula with the first agricultural settlers in the 1840s (Copley et al. 1984; Chapter 1).

In Australia, cats are both valued domestic pets and introduced feral predators. Recognising this variation in social attitude toward cats, they can be classified usefully into three groups (Moodie 1995). Housecats are almost entirely dependent on humans. Stray cats were originally housecats that now refrain from human contact. Feral cats are free-living, independent of humans and reproduce in self-perpetuating populations.

2.2.1 Reproduction and population ecology

Average body mass for feral or stray cats in Australia vary from 3.68 - 4.73 kg for males and 2.80 - 3.68 kg for females (Jones and Horton 1982, Risbey et al. 1999, Molsher 2001, Read and Bowen 2001, Short and Turner 2005, Denny and Dickman 2010).

In Australia feral cats breed seasonally during spring, summer and autumn (Jones and Coman 1982b, Molsher 2001, Read and Bowen 2001, Denny 2005). The breeding period is shorter in warm, semi-arid regions (three months; Read and Bowen 2001) than in cooler, more temperate regions (eight months; Jones 1977). Female cats reach sexual maturity at 10-12 months and have an average of two litters per year (Jones and Coman 1982b). Litters usually contain four kittens (range 2-8; (Hall and Pierce 1934, Jones and Coman 1982b, Read and Bowen 2001, Murray et al. 2007), but vary from 1.2 kittens when prey is scarce to 5.4 when prey is abundant (Short and Turner 2005). House cats can live for 31 years (Comfort 1956) and males up to 16 years of age can sire offspring, whilst females can breed up to 12 years of age (Morris 1996). Few feral cats live beyond seven years (Warner 1985). Despite this relatively short life expectancy, feral cat populations can double in 8.5 months (Short and Turner 2005).

Globally cats occur in densities of <1 to more than 2800 cats/km² (Liberg et al. 2000). In Australia cat densities in native vegetation vary between 0.03 – 4.7 cat.km² (Jones and Coman 1982a, Read and Bowen 2001, Short and Turner 2005, Denny and Dickman 2010), whereas higher densities occur in highly modified rural and urban habitats (0.7 - 800 cats.km²; Denny et al. 2002, Denny 2005, Denny and Dickman 2010). The highest densities of cats occur in resource-rich habitats such as rubbish dumps and built-up areas (Mirmovitch 1995, Hutchings 2003, Denny 2005, Denny and Dickman 2010). We could find no population estimates for cats on mainland Yorke Peninsula, but they occur at densities of 20-30 cats.km² on Althorpe Island (Copley 1991). Cats occur in almost all habitats across Australia, from subantarctic Islands to tropical savannah (Legge et al. 2017).

The social organisation and home range size of cats vary in response to landscape productivity, food abundance and population density (Liberg et al. 2000, Bengsen et al. 2016). Cats are generally solitary (e.g. Jones 1977, Jones and Coman 1982b) but live in social groups where food availability allows high population densities, such as rubbish dumps (MacDonald 1983, Turner and Meister 1994). Male home ranges are usually larger than those of females, and males increase their home ranges during the breeding season as they seek receptive females (Liberg et al. 2000, Bengsen et al. 2016). In areas of mixed agricultural and native vegetation in south-eastern Australia male home ranges vary from 4.2 to 6.2 km² and female home ranges from 1.7 to 2.4 km² (Jones and Coman 1982a, Molsher et al. 2005, Bengsen et al. 2016).

Feral cats are solitary, carnivorous hunters that generally prey on animals weighing less than 200g but can kill prey up to 5 kg (Paltridge et al. 1997, Denny and Dickman 2010, Moseby et al. 2015b). Feral cats preferentially hunt in relatively open areas that have been recently burnt and heavily grazed (McGregor et al. 2014a, McGregor et al. 2016a), where their hunting success is higher (Leahy et al. 2015, McGregor et al. 2015). Cats are widely thought of as opportunistic hunters that consume prey in proportion to their relative local abundance (Read and Bowen 2001, Paltridge 2002). However most studies of cat diets have not compared prey availability with prey consumption to test this hypothesis (Mutze 2017). Cats can preferentially hunt particular prey under some circumstances (e.g. Moseby et al. 2015b, Spencer et al. 2017, Read et al. 2018), but it is unclear whether this is generally the case. As with foxes (see section 2.1.1) introduced rabbits are important items in the diet

of cats, especially when rabbits are abundant (Read and Bowen 2001). When rabbit numbers decline cats can eat other prey (Read and Bowen 2001, Doherty et al. 2015), but even after rabbits declined by 90% following the spread of rabbit haemorrhagic disease they remained the primary prey of cats (Molsher et al. 1999).

2.2.2 Impact on native species and ecosystems

Feral cats negatively affect native species through predation and possibly disease transmission (Denny and Dickman 2010, Doherty et al. 2017a). There is limited evidence for competition over food between feral cats and quolls (Glen et al. 2011) and between cats and raptorial birds (Pavey et al. 2008). In some circumstances, feral cats can have a positive impact on native species by limiting populations of introduced mice and rabbits (Dickman 2009).

Predation by feral cats is one of the most severe threats to mammal species in Australia (Woinarski et al. 2015, Doherty et al. 2017a). Cats threaten 74 mammal taxa, 40 birds, 21 reptiles and four amphibian species (Department for the Environment 2015). There are three lines of evidence for this assertion: cat diet, the correlation of declines and the abundance of cats, and experimental manipulation of cat abundance. Feral cats consume over 400 vertebrate species (including native and introduced mammals, birds, reptiles and amphibians) and insects from 18 orders (Doherty et al. 2015, Woinarski et al. 2018a) and kill 272 million birds and 466 million reptiles annually in Australia (Woinarski et al. 2017, Woinarski et al. 2018a, Woinarski et al. 2018b). Reptiles are consumed more frequently in warmer months in arid and semi-arid areas (Paltridge et al. 1997, Molsher et al. 1999, Paltridge 2002, Kutt 2011, Woinarski et al. 2018a). Invertebrates can be a seasonally important food source (Paltridge et al. 1997, Kutt 2011, Yip et al. 2014). Carrion can also be a significant component of the diet of feral cats (Molsher et al. 1999, Woinarski et al. 2018a).

There is a consistent association between the arrival of cats and local extinction of native mammals weighing 35 g – 5 kg (Burbidge and McKenzie 1989b, Abbott et al. 2014), including in northern Australia where foxes are absent (Fisher et al. 2014). Remnant mammal populations in southern Australia have also declined after localised increases in cat abundance (Marlow et al. 2015a). Cat predation has been implicated in the failure of several small mammal reintroductions in Australia (Christensen and Burrows 1995, Short 2009, Moseby et al. 2011, Hardman et al. 2016, Short 2016). However, it is often difficult to distinguish the effects of co-occurring cats and foxes in southern Australia. Two field manipulations have confirmed that cats can have negative impacts on small mammal and reptile populations in the absence of foxes (Risbey et al. 2000, Frank et al. 2014, Stokeld et al. 2018). Removal of foxes and cats resulted in a doubling in the abundance of small mammals, whereas there was an 80 % reduction in their abundance where feral cats occurred in the absence of foxes (Risbey et al. 2000). In the absence of dingoes and foxes, native rodents declined to extinction within three months when exposed to low cat densities, but persisted at control sites from which cats were excluded (Frank et al. 2014).

Thirty-six pathogens or diseases have been recorded from Australian feral cats (Henderson 2009). The protozoan parasite *Toxoplasma gondii* is the most significant and well-studied, due to its potentially harmful effects on humans, wildlife and livestock (Henderson 2009). In Australia, cats are the sole definitive host of *Toxoplasma gondii*, which is spread through oocysts in cat faeces. Seroprevalence of stray and feral cats in Australia ranges from 0 to 96% (mean = 45 %, n = 13; Fancourt and Jackson 2014). Infection with *Toxoplasma gondii* can cause death in some marsupials

(Bettiol et al. 2000), but the prevalence and clinical effects of *Toxoplasma gondii* infection in native species are poorly understood (Parameswaran et al. 2009a, Parameswaran et al. 2009b, Hillman et al. 2016). Declining populations of eastern quolls had higher exposure (77-100 %) to *T. gondii* than stable populations (9-29 %), which probably reflects predation pressure from cats rather than any impact of the parasite (Fancourt et al. 2014).

2.2.3 Impact on agriculture

The impacts of feral cats on agriculture are not well understood, possibly because cats do not prey on sheep or cattle and a perception that cats are an economic asset to farmers because they control mice and rabbits (Doherty et al. 2017a). However, feral cats reduce agricultural production as vectors of disease (Masters 2015). The greatest economic impact of cats on agriculture is caused by the spread of two protozoan parasites which cause toxoplasmosis and sarcosporidiosis (O'Callaghan et al. 2005). Both parasites impact heavily on the sheep industry around the world (O'Donoghue 1978). In Australia, cats are the only primary host in which these parasites reproduce sexually (Munday 1975, O'Donoghue and Ford 1986).

2.2.3.1 Effect of toxoplasmosis on sheep production

Toxoplasma gondii is an intracellular parasite that infects warm-blooded vertebrates. Domestic and wild cats of various species (family Felidae) are the definitive hosts, in which the sexual part of the *T. gondii* life cycle occurs (Dubey 1995, 1996). Other mammals and birds act as intermediate hosts, in which the parasite undergoes the asexual part of its life cycle. Intermediate hosts become infected by ingesting *T. gondii* oocysts that are shed into the environment in cat faeces (Plant et al. 1974, Buxton et al. 2007). Once ingested, the parasite activates in the gut, enters the bloodstream and spreads to cells in the striated muscle and brain (Buxton 1990). After activation of the host's immune response, the parasite differentiates into the slow-growing intra-cellular bradyzoite, which cause microscopic cysts to develop (Buxton 1990). The life-cycle is completed if a cat consumes infected prey or carrion (Buxton 1990).

Toxoplasma gondii came to Australia with domestic cats and they are the only wild definitive hosts in Australia. Between 14 and 95 % of domestic cats are sero-positive to *T. gondii*, and the figure is higher among feral cats than owned cats (Dickman 1996b, Nutter et al. 2004, Elmore et al. 2010). Cats can shed oocysts for up to 14 days following consumption of infected tissue (Buxton 1990). Oocysts take 1-5 days to sporulate and become infectious (Elmore et al. 2010), and can then remain viable in the environment for up to two years (Innes et al. 2009). Infected cats typically shed oocysts only once during their lifetime, but latent infections can be triggered by illness or other stressors and cause additional shedding of oocytes (Buxton 1998).

Most birds and mammals (including humans) can be intermediate hosts of *T. gondii* (Innes 2010). Intermediate hosts usually become infected by consuming oocytes shed in cat faeces, but offspring can be infected *in utero* from infected mothers in rodents and marsupials, allowing the parasite to persist where cats are in low densities or for some time after cats have been extirpated from an area (Buxton 1990, Dubey 2009, Innes et al. 2009, Parameswaran et al. 2009a). Intermediate hosts cannot transmit toxoplasmosis between themselves, so leaving them in an area does not

increase the risk of infection to other intermediate host species, whether they are humans, sheep or native animals. The impacts of *T. gondii* infection in native animals are not well understood, but it can cause clinical disease and mortality in bandicoots (Smith and Munday 1965, Obendorf and Munday 1990, Masters 2015, Hillman et al. 2016). The prevalence of *T. gondii* among intermediate hosts is generally greater in cool temperate regions of southern Australia (Smith and Munday 1965, Dickman 1996a) and is positively associated with the population density of feral cats (Hollings et al. 2013).

Sheep are exposed to *T. gondii* oocysts in pasture, feed or bedding. The presence of cats is associated with *T. gondii* infection in domestic stock (Mainar et al. 1996, Vesco et al. 2007, Pinheiro et al. 2009, Cenci-Goga et al. 2013). If infected rodents are consumed by cats in farm buildings or paddocks, they can indirectly increase exposure of sheep to infection by *T. gondii* (Cenci-Goga et al. 2013). Toxoplasmosis in domestic stock and humans causes foetal resorption, abortion, stillbirth or neonatal mortality (Dubey 2009). Seventy percent of experimentally-infected pregnant ewes aborted (Dubey and Beattie 1988). In naturally-infected flocks of sheep more than 90% of unsuccessful pregnancies were infected with *T. gondii*, although 42 – 46% of viable lambs were also *T. gondii* positive (Duncanson et al. 2001, Williams et al. 2005). On the South Australian mainland 25% of sheep carry toxoplasmosis, but infection is close to 50% on Kangaroo Island (O'Donoghue et al. 1987, Fowler 2017). The high prevalence of toxoplasmosis on Kangaroo Island may be due to the high abundance of feral cats there (O'Donoghue et al. 1987). A live vaccine (Toxovax™) is licensed for use overseas but is not currently available for use in Australia. The cost of toxoplasmosis to the South Australian sheep industry is estimated to be \$AU 70 million annually (Fowler 2017).

2.2.3.2 Effect of sarcosporidiosis on sheep production

Sarcosporidiosis is caused by several species of *Sarcocystis*. All are obligate intracellular parasites with a two-host life cycle (Ford 1986). Each species of *Sarcocystis* has its own definitive host (a carnivore) and intermediate host (an herbivore) (Buxton 1998). Sporocysts shed into the environment by the definitive host are ingested when the intermediate host consumes pasture contaminated with faeces of the definitive host. The sporocyst hatches in the intermediate host's intestine and burrows through the intestinal wall, entering the blood stream and eventually forming a cyst in the muscle tissue (Buxton 1998). The definitive host becomes infected by consuming infected flesh of an intermediate host. The cysts rupture in the definitive host's intestine where the parasite undergoes its sexual lifecycle, ultimately resulting in sporocysts being shed in the faeces (Buxton 1998). Once shed the sporocysts can survive in the environment for up to a year (McKenna and Charleston 1992, McKenna and Charleston 1994).

Sheep are the intermediate hosts for four species of *Sarcocystis*, all of which have been identified in Australia (Savini et al. 1993). *S. tenella* and *S. arieticanis* are transmitted by canids, form microscopic cysts in the tissues and can cause clinical symptoms (including abortion and mortality) in sheep (Leek and Fayer 1978, Buxton 1998). *S. gigantea* and *S. medusiformis* are transmitted by cats and do not cause clinical disease in sheep but cause macroscopic cysts in muscle tissue (Leek and Fayer 1978, Buxton 1998). Due to the host specificity of *Sarcocystis* species, native mammals and humans cannot be infected by consuming tissue cysts in sheep meat and cats can only become infected by consuming sheep meat that contains cysts of *S. gigantea* and *S. medusiformis*.

Sarcocystis gigantea and *S. medusiformis* cause large white cysts to develop in the meat of infected sheep, leading abattoirs to reject part or all of the infected carcasses (O'Donoghue and Ford

1986). Carcasses with light infestations can be trimmed, but heavy infestations result in rejection of entire carcasses (O'Donoghue and Ford 1986, Martínez-Navalón et al. 2012). The specificity of serological tests does not allow identification of the *Sarcocystis* infection to species level (Tenter 1995). Microcysts can occur in infected lambs by four months of age and macroscopic cysts only become apparent seventeen months after consumption of *Sarcocystis* sporocysts (Whiting 1972, Munday 1975, Ford 1986, Pythian et al. 2018). Macrocyts can persist and grow in living sheep for years after the initial infection (Tenter 1995). The prevalence of *Sarcocystis* infection in sheep can vary widely (O'Donoghue and Ford 1986). In south-eastern Australia cysts occurred in 0.06 - 2.5% of carcasses overall, but in some breeds of sheep all carcasses had cysts (Whiting 1972, Ford 1974, Munday 1975). More than 93 % of sheep carcasses from South Australian abattoirs showed evidence of *Sarcocystis* infection, but fewer than 7% of carcasses exhibited visible macrocysts (O'Donoghue and Ford 1986). More recent estimates from Australian abattoirs report a 0.01% rate of whole carcass rejection and a 0.3% rate of carcass trimming to remove infected tissue (GHD Pty Ltd 2015). Costs associated with sarcosporidiosis in Australia have been estimated at \$0.89M annually (GHD Pty Ltd 2015).

2.2.4 Control methods

Cats inhabit the entire 7,692 million km² of Australia (Denny and Dickman 2010). There are currently no techniques that could be applied at a broad scale to eradicate cats. Gene editing techniques to control invasive animals is currently an active area of research, but we are in the early days of this developing technology (Moro et al. 2018). There are no biological control agents suitable for use in Australia (Van Rensburg et al. 1987, Department for the Environment 2015). Detector dogs can locate cats where densities are low or cats remain after application of other control methods (Fisher et al. 2015, McGregor et al. 2016b). Historically, it has only been possible to eradicate cats over relatively small areas on islands and within fenced predator exclosures using a combination of shooting, trapping, and non-specific lethal baiting (Veitch 1985, Dickman 1996b, Nogales et al. 2004, Hayward et al. 2014, Doherty et al. 2017a).

Currently, lethal baiting is the most promising method of cat control, due to recent development of a 'chipolata' style meat bait designed specifically for cats (Bonner 1999, Marks et al. 2006, Doherty et al. 2017a). Eradicator® baits contains 4.5 mg of sodium monofluoroacetate (1080) poison per 15 g bait and was registered in December 2014 for use in Western Australia. Curiosity® baits contain 78 mg of para-aminopropiophenone (PAPP) poison encapsulated in a 'hard shelled delivery vehicle'. Curiosity® is not yet registered for use but has been developed for use where native fauna is less tolerant of 1080. A third bait, Hisstory® contains encapsulated 1080 and is being tested for use in areas where native species susceptible to PAPP are active at the time of baiting (Algar et al. 2015). PAPP is considered to cause death more humanely than 1080 (de Tores et al. 2011). Curiosity® baiting in the Flinders Ranges resulted in death of 2/20 (10%) of radio-collared cats in the baited area, and a further 2 cats survived after consuming baits (Johnston et al. 2012). In contrast, Eradicator® baiting in the same area resulted in the death of 85% of radio-collared cats in the baited area (Gerlach 2018). Curiosity® baits have caused measurable reductions in cat density in field trials (Johnston et al. 2011, Johnston et al. 2012, Johnston et al. 2014). Eradicator® baits have also substantially reduced cat populations in Western Australia (Algar et al. 2002, Burrows et al. 2003, Algar and Burrows 2004). Cat baits are generally distributed from an aircraft at 50 baits km² (Algar et al. 2013), although they have also been dispersed by hand (Doherty and Algar 2015). They have also been suspended above ground

to limit interference from non-target species on Christmas Island (Algar and Brazell 2008). Despite early indications to the contrary (Hetherington et al. 2007), encapsulation of the toxin in a hard shelled delivery vehicle has still resulted in poisoning of non-target species (de Tores et al. 2011). The effectiveness of baiting for feral cats is highest when prey availability is lowest (Christensen et al. 2013) and may be reduced if high rainfall before baiting elevates prey populations (e.g. Johnston et al. 2012). Baiting should take place in autumn and early winter when rabbit abundance and reptile activity are lowest (Algar et al. 2013).

Non-food-based methods of cat control may help address the issue of low bait uptake when prey is abundant. Novel devices for cat control include 'grooming traps' that exploit cats' innate grooming responses when sprayed by liquid and are designed specifically to target cats and not effect non-target species. Trials of grooming traps on captive feral cats using PAPP spray resulted in 50% mortality (Read et al. 2014). However, grooming traps are not yet commercially available and still require optimisation. Another novel method to reduce the impacts of cat predation involves targeting individual cats that specialise in hunting vulnerable species by making prey items lethally toxic or unpalatable (Read et al. 2015). Subcutaneous insertion of toxic small capsules of toxin or fitting toxic collars to individual animals during monitoring or reintroduction programs could poison or deter individual cats that capture these 'toxic trojans'. This approach to minimising cat impacts on reintroductions could be trialled during planned reintroductions of soil engineers and native predators to Yorke Peninsula (see chapters 5 and 7).

2.3 Expected ecosystem responses to invasive predator control

Large predators typically exert top-down effects on ecosystems through their direct predatory and competitive interactions with herbivores and smaller predators (Ritchie and Johnson 2009, Ritchie et al. 2012). This means that changes in the abundance of one predator in an ecosystem may affect the abundance, distribution and/or behaviour of others. Thus management of one predator can have strong implications for the management of other predators and cascading effects on prey and plant communities (Ritchie et al. 2012). The precise nature of the cascading effects are difficult to predict with our present knowledge (Courchamp et al. 2003b) and are likely to be site specific.

2.3.1 Cat responses to fox control

Foxes and cats have strongly overlapping diets in Australia, indicating potential for competition (Bayly 1978, Catling 1988, Brunner et al. 1991, Molsher et al. 1999, Molsher et al. 2000, Read and Bowen 2001, Molsher et al. 2005, Glen et al. 2011, Molsher et al. 2017, Woinarski et al. 2018a). Foxes usually dominate cats in direct encounters (Molsher et al. 1999) and prey on cats (Coman 1973, Brunner et al. 1991, Risbey et al. 1999, Paltridge 2002). Removal of foxes is often associated with increases in cat abundance (Short et al. 1994, Risbey et al. 2000, Davey et al. 2006, Robley et al. 2010, Marlow et al. 2015a, Wayne et al. 2017a, Hunter et al. 2018). One study found no relationship between fox and cat abundance, but reported differences in cat diet, behaviour and habitat use between sites with fox control and sites without fox control (Molsher et al. 2017). Reduction in the density of foxes on southern Yorke Peninsula may have the unwanted effect of releasing feral cats from top-down control. Should this occur, cats will increase in abundance with commensurate impacts on biodiversity and agriculture (see sections 2.2.2 and 2.2.3). This highlights the need for integrated

control of both foxes and cats to achieve the desired outcome of the southern Yorke Peninsula Rewilding Project.

2.3.2 Rabbit responses to invasive predator control

Foxes and cats are both reliant upon (Newsome et al. 1997, Courchamp et al. 1999, Edwards et al. 2002, Holden and Mutze 2003, Robley et al. 2004) and can suppress rabbit populations (Pech et al. 1992, Banks et al. 1998, Banks et al. 1999, Banks 2000, Robley et al. 2004, Guerrero-Casado et al. 2013, Fernandez-de-Simon et al. 2015). Foxes and cats can regulate rabbit populations at low to medium rabbit densities (Catling 1988, Newsome et al. 1997). However, environmentally induced flushes in primary productivity (e.g. following rainfall) may allow rabbits to increase in density to an extent that predators are no longer able to regulate them (Newsome 1990, Pech et al. 1992, Davey et al. 2006). Any significant reduction in the density of cats and/or foxes may result in an increase in rabbits due to decreased predation pressure, especially following periods of high rainfall. Any increase in rabbit abundance is likely to negatively impact native vegetation and agriculture through increased grazing pressure (Cooke 2012, Mutze et al. 2016). In the absence of predation by cats and foxes, the rewilding project aims to regulate rabbit density by reintroducing native predators (Chapter 7). Increased abundance of goannas following control of invasive predators is also likely to regulate rabbit abundance.

2.3.3 Goanna responses to invasive predator control

Reptiles are rarely considered in the context of predator management programs (Bryant 2012, Sutherland and Bryant 2014). Foxes and cats show considerable dietary overlap with medium-sized and large goannas (Sutherland et al. 2011a) and goannas have been interpreted as the nearest ecological equivalent to foxes in Australia (Pianka 1986, Pianka 1989). Small to medium-sized goannas are also preyed upon by foxes (Catling 1988, Paltridge 2002, Olsson et al. 2005, Woinarski et al. 2018a). Therefore, controlling foxes may lead to an increase in the population density of the medium-sized Rosenberg's goanna (*Varanus rosenbergii*) on Yorke Peninsula, due to decreased predation and/or competitive release.

This possibility is supported by a doubling of lace monitor (*Varanus varius*) abundance in areas where foxes had been controlled (Hu et al. 2019), and a five-fold increase in sand goannas (*V. gouldii*) following fox removal at two different sites (Olsson et al. 2005, Read and Scoleri 2015), though it is not clear whether reduced predation or competitive release explain these responses. Rosenberg's and sand goannas are closely related species (Pianka et al. 2004), both medium-sized and similar in many aspects of their biology, including body size and diet (King and Green 1999, Sutherland 2011). Therefore, it is possible that Rosenberg's goannas would show a similar response to that of sand goannas to decreased fox density on southern Yorke Peninsula.

Dietary overlap is higher between medium-sized goannas and cats than it is to foxes (Sutherland et al. 2011a). Cats are also known predators of goannas in general (Catling 1988, Dickman 1996a, Molsher et al. 1999, Read and Bowen 2001, Paltridge 2002, Doherty et al. 2015, Woinarski et al. 2018a), and of Rosenberg's goannas in particular (Rismiller and McKelvey 2003). Eradication of

feral cats from Faure Island in Shark Bay, Western Australia, caused a large increase in sightings of sand goannas (Rowles 2008).

Goannas can limit or regulate prey populations and have been used for pest control. In India, goannas are protected because they eat crabs that plague rice paddy dykes (Greer 1989). On some Pacific islands mangrove monitors (*Varanus indicus*) regulate populations of polynesian rats (*Rattus exulans*) and some beetles (Uchida 1967, Cota 2008). More locally, Rosenberg's goannas were reportedly introduced to Reevesby Island in Spencer Gulf to reduce the number of snakes (Mirtschin 1982, Robinson et al. 1985).

Should the population density of Rosenberg's goannas increase on Yorke Peninsula following control of cats and foxes, it is likely to result in top-down changes in the abundance of the species preyed upon by goannas. Small lizard captures increased in a fox, cat, rabbit and goanna-free enclosure compared to control sites in New South Wales (Olsson et al., 2005), though it is not clear which individual or combination of predator species was responsible for this change. In contrast, small lizard captures declined in a similar enclosure at the Arid Recovery Reserve, but where goannas had not been excluded (Moseby et al. 2009a).

The increased abundance of goannas inside the Arid Recovery Reserve may, at least partially, explain the decline in small reptile captures there (Read and Scoleri 2015). Goannas can also influence mammal and insect populations. On the Marshall and Palau islands Mangrove monitors (*Varanus indicus*) were considered a nuisance and cane toads were introduced to control their numbers (Dryden 1965, Uchida 1967, Cota 2008). Following the decline in mangrove monitors, Polynesian rats increased, and the numbers of beetles that harm coconuts also rose (Uchida 1967, Cota 2008). Predation by the arboreal lace goanna (*V. varius*) prevented population recovery of ringtail possums (*Pseudocheirus peregrinus*) following a bushfire (Russell et al. 2003).

Rosenberg's goannas are ectotherms, relying on environmental heat and solar radiation to maintain their preferred body temperature (King 1980, Green et al. 1991, King and Green 1999). They emerge from burrows after dawn to bask and commence activity when their body temperature approaches 35°C and return to their burrow in the evening, when they can no longer maintain their preferred body temperature. Their diurnal activity is strongly influenced by ambient temperature and solar radiation across the year. Consequently, goanna diel activity is longest in summer (King 1980, King and Green 1999, Sutherland and Predavec 2010). In contrast endothermic predators, such as cats and foxes, are not restricted by ambient temperature and solar radiation. They can hunt for prey over a wider range of times. Cats and foxes are generally crepuscular or nocturnal (Sutherland and Predavec 2010, Brook et al. 2012), although they can be active at any time of the day. Ectothermic predators have resting metabolic rates 10-20 times less than endotherms (Pough 1980). The rates at which ectotherms take prey, and their populations can respond to changes in prey abundance are lower than endotherms. However the relatively low energy demands of ectotherms mean they may occur at higher densities than mammalian or avian predators occupying similar niches (Sutherland and Bryant 2014). We are not aware of any empirical data allowing comparison of the relative impact of ecologically equivalent reptilian and mammalian predators, but reptilian predators such as Rosenberg's goanna, should pose a smaller potential impact on their prey than mammalian predators due to their lower energetic requirements and greater restriction by ambient temperatures.

Rosenberg's goannas may consume fox and cat baits laid during baiting programs on Yorke Peninsula (Short et al. 1997, de Tores et al. 2011, Jessop et al. 2013). However goannas show a high tolerance for sodium monofluoroacetate ('1080'; Mcilroy et al. 1985), suggesting that consumption of 1080 baits is unlikely to harm them. This is supported by a five-fold increase in the density of sand

goannas in areas that had been subjected to 1080 baiting over ten years (Olsson et al. 2005). In contrast to 1080, goannas are susceptible to para-aminopropiophenone (PAPP; de Tores et al. 2011). The ectothermic physiology of Rosenberg's goannas could be used to mitigate any potential for Rosenberg's goannas to take baits laid to control foxes and/or cats. Laying baits in cold weather, when goanna activity is low, should minimise the risk of goannas taking PAPP-based baits (Jessop et al. 2013).

In the absence of other predators, island populations of Rosenberg's goanna are larger than their mainland counterparts (Case and Schwaner 1993). The reasons for these differences are unclear, but may reflect competitive release in the absence of predators with whom they co-occur on the mainland (Sutherland et al. 2011a). Equally it could be due to other differences between islands and the mainland (e.g. food availability due to seabird colonies) that allow goannas to live longer and reach larger sizes on islands (e.g. Johnston 2011). For whatever reason, we may see changes in goanna body size in response to control of non-native predators and re-introduction of native predators on Yorke Peninsula.

2.4 Knowledge gaps

The existing 'baiting for biodiversity' program to control foxes on southern Yorke Peninsula has resulted in measurable increases in lamb production on properties that lay baits (Sharp 2018). However, the success of such community-based baiting programs relies on most landholders being involved. Foxes readily use unbaited areas as refuges and dispersal routes to colonise areas where baiting has reduced fox density (Greentree et al. 2000, Gentle et al. 2007). Although baiting for cats has only recently become practicable, cats are likely to respond in similar ways to foxes in this regard. Monitoring of fox and cat density across the rewilding project areas should be undertaken to (1) demonstrate the effectiveness of the baiting program, and (2) identify areas of high fox/cat density or dispersal that could be targeted for more intensive management to enhance effectiveness of the overall program.

The effectiveness of fox and cat baiting for biodiversity conservation and agriculture in general has been demonstrated (see sections 2.1.2 to 2.3.4 above), but the responses of individual native species and changes in ecosystem processes can be unpredictable and site specific (Short 2009, Wayne et al. 2017a, Wayne 2018). For this reason, it is important to monitor responses of target species to allow adaptive management in the event of unexpected deleterious responses to fox and cat control. In southern Yorke Peninsula monitoring of responses to fox and cat control should focus on mice (see chapter 4), rabbits (see section 2.3.2), Rosenberg's goannas (see section 2.3.3), key threatened birds such as mallee fowl, stone curlews, hooded plovers and mallee whipbirds, along with all reintroduced mammals.

Western grey kangaroos are regarded as not susceptible to fox and cat predation (Radford et al. 2018). This may be true for cats, but there is evidence that foxes may influence grey kangaroo behaviour and population density. Foxes attack western grey kangaroos (Meek and Wishart 2017) and western greys use olfactory cues to avoid areas where foxes have defecated (Mella et al. 2014). Closely related eastern grey kangaroos (*Macropus giganteus*), show slower population growth in areas where foxes are present due to higher mortality of young recently emerged from the pouch (Banks et al. 2000). They also alter their foraging behaviour to avoid foxes (Banks 2001). These observations suggest that removal of foxes could result in an increase in kangaroo abundance on

Yorke Peninsula. This would be indicated by an increase in the frequency of female kangaroos with young-at-foot in areas where fox baiting has occurred compared to unbaited areas. Should this be observed, control of kangaroo populations may be necessary. Kangaroo populations are generally regulated by access to water, and the effects of rainfall on vegetation growth (Pople et al. 2000).

Reducing the density of feral cats on southern Yorke Peninsula should result in a decrease in the prevalence of toxoplasmosis and sarcosporidiosis in agricultural stock and native animals (see section 2.2.3), resulting in improved animal health and reduced losses to agriculture. It is unclear what density of cats is required to achieve this, or how long it might take to see a decline in the prevalence of toxoplasmosis and sarcosporidiosis. Data collected by Biosecurity SA may be suitable for monitoring this aspect of the southern Yorke Peninsula rewilding project (Dr Elise Spark, Senior Veterinarian, Biosecurity SA – personal communication with G. Johnston, 2019).

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3 Barrier Fence

A key part of the rewilding of southern Yorke Peninsula will be construction of two barrier fences (Figure 3.1). The first of these will enclose the area subject to co-ordinated fox control under the existing 'Baiting for Biodiversity' program (Northern and Yorke Natural Resources 2018). The fence will reduce movement of foxes from outside the area to enhance the effectiveness of baiting for biodiversity conservation and agricultural outcomes. The 'Baiting for Biodiversity' program will continue inside the fenced areas, and new technologies for targeted cat control will also be employed. Future co-ordinated fox control outside of the first fence may expand the area of reduced fox density, which would be augmented by a second fence further up the peninsula, with further fox control north of that. The need to allow unimpeded vehicular traffic along public roads will require breaks in the fence. Consequently, the planned barrier fences will act as 'leaky' or semi-permeable barriers that will severely limit introduced predator re-invasion, rather than excluding them entirely.



Figure 3.1: Map of southern Yorke Peninsula showing the approximate position of two proposed barrier fences to reduce re-invasion of foxes into areas of low fox population density created by the baiting for biodiversity program.

There is a long history of using barrier fences to limit dispersal of animals in Australia. Fences were initially used to protect agriculture and have been used for conservation of endangered wildlife more recently. The first barrier fences in Australia were built by private land holders with the hope of limiting the spread of introduced rabbits in the 1860s, using whatever materials were at hand

(Munday 2017). After wire mesh became available in 1885 (Munday 2017), mesh fences were built in an attempt to limit dispersal by rabbits (Broomhall 1991). Between 1900 and the 1960s a 5,600 km-long wire mesh barrier fence was built across South Australia, New South Wales and Queensland to support dingo control in sheep grazing country to the south by limiting movement of dingoes back into areas from which they had been (largely) removed (McKnight 1969, Yelland 2012). The dingo fence is an example of a semi-permeable barrier fence that is effective at reducing dingo re-invasion into sheep country to the advantage of agricultural productivity.

The first barrier fence for conservation was erected to protect endangered native mammals from introduced cats and foxes at Warrawong Sanctuary near Adelaide in 1975 (De Alessi 2003). Since then, over fifty areas have been protected for conservation by barrier fences in Australia (Somers and Hayward 2012, Australian Wildlife Conservancy 2018). Most of these fences have been built to create predator 'exclosures' in inland areas. Three other peninsulas have been fenced for conservation: Peron Peninsula and Heirison Prong in Western Australia (Short et al. 1994, Morris et al. 2004), and Venus Bay in South Australia (South Australian Department for Environment and Heritage 2006). The water surrounding a peninsula provides a natural barrier which can be augmented by a barrier fence across an isthmus at much reduced cost than fully fencing an equivalent area. In addition to the planned barrier fence(s) across southern Yorke Peninsula, there are plans erect barrier fences at for conservation across two other peninsulas in Australia: Wilson's Promontory in Victoria (Robley et al. 2003) and Dudley Peninsula on Kangaroo Island, South Australia (Kangaroo Island Natural Resources 2018) (Table 3.1).

Fenced 'exclosures' have demonstrated that removal or reduction of feral predators (cats and foxes) greatly enhance the likelihood of successful reintroduction of native mammals for conservation (Moseby et al. 2011, Somers and Hayward 2012, Frank et al. 2014, Hayward et al. 2014, Legge et al. 2018a, Ringma et al. 2018), and results in increased densities of remnant native fauna, and restoration of ecosystem functions (Read et al. 2008, Moseby et al. 2009a, Munro et al. 2009, James et al. 2010a, Hayward et al. 2014). The combined experience from projects utilising barrier fences is converging on two key points that are critical to the effectiveness of barrier fences: (1) fence design and (2) maintenance are both critical to the ability of cats and foxes to breach a barrier fence.

3.1 Fence design

A barrier fence must be designed with the physical characteristics and abilities of the target species in mind, including their size, ability to jump, dig, climb and bite through fence materials (Dickman 2012). In Australia, most barrier fences for conservation have aimed to restrict movement of foxes, cats and rabbits.

Feral cats are unable to bite through standard gauge fence wire, but foxes can tear through some types of woven wire fences (Coman and McCutchan 1994, Forbes-Harper et al. 2017). Foxes are generally considered to be poor climbers, but have been observed scaling netting and chain mesh fences over two metres high and are capable of leaping over 1.3 m high fences and climbing trees to a height of 1.4 m (Coman and McCutchan 1994, Moseby and Read 2006, Robley et al. 2007, Mella et al. 2018). Cats are good climbers and are capable of leaping fences up to 1.4 m high (Coman and McCutchan 1994, Robley et al. 2007). Cats are reluctant diggers, but can excavate short burrows in soft substrates, whereas foxes are proficient diggers, digging dens (burrows) of their own construction (Lloyd 1980).

Early barrier fence designs curtailed movements of cats and foxes but did not stop them entirely (Poole and McKillop 2002, Moseby and Read 2006, Robley et al. 2007). Systematic trials of



Figure 3.2: The barrier fence at the Arid Recovery Reserve near Roxby Downs in South Australia. Photo: Charlotte Mills from Mills et al. (2018)

several different fence designs showed that wire netting fences 180 cm high with a foot apron and a 60 cm-wide curved overhang ("floppy top") were effective at containing captive predators (Moseby and Read 2006, Robley et al. 2007). Use of metal (rather than wooden) fence posts greatly reduced animals' ability to climb a fence. Hard wire mesh was necessary to prevent foxes chewing through. A "floppy top" was necessary to stop animals from climbing over the fence from underneath. Electric wires could add slightly to the effectiveness of an appropriate physical fence.

The most effective barrier fence to date was based on these findings and was first installed at the Arid Recovery Project near Roxby Downs, South Australia (Moseby and Read 2006; Figure 3.2). While Moseby & Read (2006) recommended a high (1.8m) and a low (1.2m) version of this fence design based on short-term trials in small areas, they had only used the taller version as a barrier fence in the field at the time their article was published. Since then the lower fence has been installed in the field and has been found wanting (John Read and Katherine Moseby, Ecological Horizons Pty Ltd, and Katherine Tuft, Manager, Arid Recovery Project, personal communication to G. Johnston, 2018). The low fences at the Arid Recovery Reserve are now being replaced by the high fences as a matter of priority (Katherine Tuft, Manager, personal communication to G. Johnston, 2018). The key problems with the low fence have been the ability of foxes and cats to jump onto the floppy top and use it to climb over the fence, particularly following damage to the floppy top by kangaroos and goats (John Read, Katherine Moseby & Katherine Tuft, personal communication to G. Johnston, 2018). The 1.8m high "floppy top" barrier fence is now the standard fence design being used to manage cat and fox densities for conservation, e.g. Newhaven Reserve in the Northern Territory (Australian Wildlife

Conservancy 2018); planned fences at Wilson's Promontory in Victoria (Robley et al. 2003) and Dudley Peninsula South Australia (Kangaroo Island Natural Resources 2018).



Figure 3.3: Barrier fence installed in 1995 on the Peron Peninsula, Western Australia to exclude introduced predators. Foxes and cats are present on the far side of the fence. The road provides public access and limits effective exclusion of these predators despite the presence of a stock grid and an ultrasonic noise emitter. Photo: Western Australia Department of Environment and Conservation in De Tores and Marlow (2012).

3.2 The need to monitor and maintain barrier fences

For a barrier fence to remain effective it must be monitored regularly and any damage swiftly repaired to maintain its integrity (McKnight 1969, Broomhall 1991, Coman and McCutchan 1994, Short et al. 1994, Morris et al. 2004, Moseby and Read 2006, South Australian Department for Environment and Heritage 2006, Pickard 2007, Connolly et al. 2009, Moseby et al. 2009a, Moseby et al. 2011, De Tores and Marlow 2012, Yelland 2012, Hayward et al. 2014). The type and level of maintenance required will vary according to the fence design, construction, and features of the local environment, including community attitudes toward the fence. Fences are easier to build and maintain in open environments than in heavily forested or topographically diverse ones (Mills 2010). Maintenance requirements can be minimised if the fence is meticulously constructed with high quality materials, and with due consideration to environmental conditions. Methods for monitoring barrier fences include remote access trail cameras, direct inspection of fence and animal spoor (tracks, faeces and traces of fur). Community support will be critical to avoid vandalism to the Yorke Peninsula barrier fence because the human population is high, relative to most other areas where barrier fences have been used for conservation.

3.3 Managing the ends of fences on the coast

Fenced peninsulas rely on the surrounding sea, as well as a fence to act as a barrier against introduced predators. Limiting predator incursions at the point where the fence meets the coast remains a challenge (Short et al. 1994, Morris et al. 2004, South Australian Department for Environment and Heritage 2006). The major problems are (1) maintaining a fence exposed to corrosion from salt water, (2) build-up of seaweed and tidal flows damage fence and footings, and (3) preventing animals from moving around the ends of the fence at low tides.

3.4 Managing public access roads through the fence.

The southern Yorke Peninsula barrier fence will intersect with several public access roads. This is a feature it has in common with other peninsulas that have already been fenced for conservation in Australia (Figure 3.3; Short et al. 1994, Morris et al. 2004, De Tores and Marlow 2012). Public access roads limit the effectiveness of barrier fences to some extent, making them effectively semi-permeable. However, these 'leaky' barrier fences have limited dispersal of cats and foxes sufficiently to allow successful suppression of their populations inside fences areas, reintroduction of some locally extinct mammals and population increases in resident native animals (Morris et al. 2004). Similarly, a semi-permeable barrier fence has sufficiently limited dispersal of dingoes to ensure viability of sheep farming in south-eastern Australia (McKnight 1969, Yelland 2012). Techniques to limit movement of foxes and cats through necessary breaks in barrier fences are being actively explored for other planned barrier fences, including the use of cattle grids, fence 'wings' to direct predators away from breaks in the fence and baits, audio aversion stimuli and electrified road mats (Robley et al. 2003, Morris et al. 2004, Kangaroo Island Natural Resources 2018). Ongoing communication with staff managing those projects should be maintained to ensure maximum effectiveness of the proposed southern Yorke Peninsula barrier fence. The ability of natural geographic barriers to limit dispersal (Olsson and Widen 2008, Noren et al. 2009) suggest that even a 'semi-permeable' barrier fence will greatly enhance the management of invasive predators within the fenced area.

3.5 Disadvantages of barrier fences

Barrier fences have clear conservation benefits (see above), but they also have several potential disadvantages, the relevance of which varies from one situation to another (Long and Robley 2004, Bode and Wintle 2009, Somers and Hayward 2012). Firstly, fenced populations of animals can suffer from inbreeding depression, especially if the founding stock have limited genetic variation. This may limit their evolutionary potential to adapt to environmental changes in the future (e.g. climate change). However, when one is dealing with endangered species, their population size is already low, and building the number of individuals increases the possibility of new mutations to add genetic variation. This is particularly true in large areas, such as southern Yorke Peninsula, which have the potential to harbour large populations of several species of endangered mammals. Should different genetic stocks be available, managed mixing of them will maximise genetic diversity.

Secondly, reintroduction of endangered animals into predator free areas can result in the loss of behaviours that protect them from predators (Miller et al. 1990, McLean et al. 1994, Blumstein and Daniel 2005, Banks and Dickman 2007, Jolly et al. 2018). Training to improve the anti-predator behaviours in animals has been attempted to address this issue, but the results have been varied (Miller et al. 1990, Maloney and McLean 1995, Griffin et al. 1999, Azevedo and Young 2006, Moseby et al. 2012, West et al. 2018). Most demonstrations of the effectiveness of anti-predator training have been restricted to captive situations (cf. West et al. 2018) and we could find few studies that sought to test whether such training leads to improved survival in the field (cf. White et al. 2005, Moseby et al. 2018a). Planned predator control on Yorke Peninsula is intended to extirpate introduced predators completely, and planned reintroductions include native predators. Therefore, any animals reintroduced to Yorke Peninsula are likely to develop/retain anti-predator behaviours. Existing and reintroduced populations have persisted or shown rapid increases in response to fox control without complete removal of foxes from the landscape on Yorke Peninsula (SA mainland tammar wallabies) and other places (Danell and Hörnfeldt 1987, Short et al. 1994, Armstrong 2004, Mawson 2004, Hayward et al. 2005a, Jarnemo et al. 2005, Olsson et al. 2005, Dexter and Murray 2009, Wheeler and Priddel 2009, Kinnear et al. 2010, Sharp et al. 2010, Robley et al. 2014, Moseby et al. 2015a, Sharp et al. 2015, Brandle et al. 2018).

Thirdly, fences designed to limit movement of a target species also have impacts on non-target species (e.g. McKillop and Wilson 1987, McKillop and Sibly 1988, Lund and DeSilva 1994). These unintended impacts can include limiting movements of non-target species and animals becoming entangled in fences (Burger and Branch 1994, Lund and De Silva 1994, Van der Ree 1999, Long and Robley 2004, Hayward and Kerley 2009, Ferronato et al. 2014). By limiting movement, barrier fences can prevent breeding between animals on either side of the fence (Williamson and Williamson 2009, Linnell 2016), block routes used by seasonally migrating animals (Hailey and DeArment 1969, Johnson 2006b, Russell and Cohn 2012, Bradby et al. 2014, Madani et al. 2016). Fence designs that allow movement of selected non-target species can overcome this problem in some circumstances (Beckmann 1990, Lehnert and Bissonette 1997, Finch et al. 2006, Crisp and Moseby 2010, Coates 2013, Butler et al. 2019), but may still limit movement of animals (Olsson and Widen 2008).

In enclosed areas populations may increase to the point that they exceed the natural carrying capacity of the area (Crisp and Moseby 2010, Hayward et al. 2014, Linley et al. 2017, Moseby et al. 2018b). In this circumstance the population may need to be actively managed to prevent starvation of the animals and to prevent damage to vegetation and other features of the animal's habitat (Gowans et al. 2010, Herbert et al. 2010, Morgan and Pegler 2010, Linley et al. 2017, Moseby et al. 2018b). This may well occur with kangaroos on southern Yorke Peninsula.

Table 3.1: Summary of barrier fences constructed across peninsulas in Australia.

Location	Date	Area (Ha)	Biome	Key species protected	Fence design
Venus Bay, SA ¹	1996	1,460	Coastal heath, grassland, woodland	<i>Burhinus grallarius</i> <i>Macrotis lagotis</i> <i>Bettongia penicillata</i> <i>Leporillus conditor</i>	2.5 km long. 2.1 m high mesh fence with wooden uprights, a floppy top and a mesh apron.
Herrisson Prong, WA ²	1991	1,200	Coastal heath, open low shrubland	<i>Perameles bougainville</i> <i>Bettongia lesueur</i> <i>Leporillus conditor</i> <i>Pseudomys fieldi</i>	3 km long. 1.35 m high with 45° solid over-hang outside. Constructed of 30 mm wire mesh over 15 alternating electrified and unpowered horizontal wires. A 30 cm apron of 30 mm mesh on the ground outside.
Peron Peninsula, WA ³	1995	105,000	Open low shrubland, dense closed shrubland, some hummock grassland	<i>Leipoa ocellata</i> <i>Isoodon obesulus</i> <i>Macrotis lagotis</i> <i>Bettongia penicillata</i> <i>Lagostrophus fasciatus</i> <i>Lagorchestes hirsutus</i>	3 km long. 2m high with a 30cm wide horizontal overhang of rabbit mesh with two electric wires underneath the overhang. A buried mesh apron outside,
Dudley Peninsula, Kangaroo Island, SA ⁴	planned	6,500	Coastal heath, grassland, woodland, agricultural land	Cat extermination for all native species	6 km long. 1.8 m high with 0.6 m curved 'floppy top' on both sides. Supported by 2.1 m star pickets spaced 4 m apart. Constructed of 50 mm wire mesh. A 30 cm apron will be secured into the ground on both sides. Two solar powered 7 kV electric wires will be suspended from the fence at 1.2 and 1.5 m.
Southern Yorke Peninsula, SA	planned		Coastal heath, grassland, woodland, agricultural land	Barrier fence to limit immigration of foxes & cats for agricultural and conservations purposes.	?
stage 1		140,500			19 km long fence.
Stage 2		83,900			31km long fence.
Total		224,400			

Sources: ¹ South Australian Department for Environment and Heritage (2006) ; ² Short et al. (1994) ; ³ Morris et al. (2004), De Torres & Marlow (2012); ⁴ (Kangaroo Island Natural Resources 2018).

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4 Removing invasive house mice

The house mouse (*Mus musculus*) is a cosmopolitan, invasive omnivore that has spread by colonisation and introduction from central Asia (Long 2003). Their global distribution equals that of humans, with whom they can live as commensals (Bronson 1979, Brooke and Hilton 2002). Mice were probably first brought to the east coast of Australia from the United Kingdom after 1770 on ships (Gabriel et al. 2011). They now occur throughout Australia (Menkhorst and Knight 2011). The oldest specimen of house mouse in an Australian Museum was lodged in 1841 (Mahoney and Richardson 1988). Mice have important impacts in urban areas, on agriculture and biodiversity (Capizzi et al. 2014). On Yorke Peninsula their impacts on biodiversity and agriculture are of primary concern.

House mice are generally thought to have modest ecological impacts compared to other rodents (Simberloff 2009). However, their impacts have been overlooked or confounded with the impacts of other co-occurring rodents, particularly rats (*Rattus* spp.) (Angel et al. 2009). Predation by introduced mice has caused population reductions and extinctions in plants (Smith and Steenkamp 1990, Chown and Smith 1993, Avenan and Smith 2004, Angel et al. 2009), invertebrates (Marris 2000, Jones et al. 2003, St Clair 2011, McClelland et al. 2018), lizards (Norbury et al. 2014) and birds (Smith et al. 2002, Wanless et al. 2007). Much of this information comes from sub-Antarctic islands, and little is known about the biodiversity impacts of house mice on biodiversity in other environments. In contrast, the impacts of house mice on agriculture have been well documented.

4.1 The impact of mice on agriculture

Globally, house mice are regarded as one of three important rodent pests due to their extensive distribution and impacts on agricultural crops (Capizzi et al. 2014). They are serious agricultural pests in Australia, particularly in southern grain growing regions (Singleton et al. 2005, Singleton et al. 2007b). House mice thrive in highly modified landscapes due to a lack of other small mammal competitors and its ability to exploit available resources (Brown et al. 2010). The impacts of mice occur at all stages of cereal crop development from sowing to harvest and post-harvest losses can also occur through mice consumption and contamination of stored grain and damage to infrastructure (Makundi et al. 1999, Brown and Singleton 2000).

Most agricultural costs due to mice have been reported in association with mouse plagues. A plague in Victoria during 1984 caused \$10.3 million losses in winter crop and horticultural production (Kaboodvandpour and Leung 2010). Another plague in Victoria and South Australia during 1993-1994 cost the farming industry \$64.5 million (Caughley et al. 1994b). A plague on the Eyre Peninsula in 2010 caused losses in grain yields valued at \$20-40 million (Mutze 2014). Mice at non-plague densities can also cause substantial damage, although this has not been widely documented. Mice at a density of 200-300 mice.ha⁻¹ per caused \$1.6 million damage to soybean crops in one region of Victoria (Singleton et al. 1991). A crop simulation model predicted that significant crop damage (>5%) could occur when mice reach densities of 100 per hectare (Brown et al. 2007). Conducting pre-

emptive mouse control every year is not economically viable, in part due to the unpredictability of population densities and future plagues (Davis et al. 2004).

4.2 Population dynamics of mice in Australia

Wild house mouse population densities vary between < 6 mice.ha⁻¹ and 2716 mice.ha⁻¹ (Singleton et al. 2007b, Brown et al. 2010, Wilson et al. 2018). The highest densities occur erratically during 'plagues' in Australia (Singleton et al. 2005). Mouse populations are regarded as low at densities less than 50 mice.ha⁻¹ (trap success $< 8\%$), outbreak at densities of 50-200 mice.ha⁻¹ (trap success of 8-32%), minor plague at densities of 200-800 mice.ha⁻¹ (trap success of 32-125%) and major plague at densities greater than 800 mice.ha⁻¹ (trap success $> 125\%$) (Singleton et al. 2005). Plagues don't occur outside of Australia, except for occasional outbreaks in the north-west plateau region of China (Redhead 1988). Mouse plagues occur on average every 6.3 years in South Australia, but are rare in the wheatbelt of Western Australia (Singleton et al. 2005). During plagues mouse populations irrupt synchronously over thousands of square kilometres (Caughley et al. 1994a). Sixteen mouse plagues occurred in South Australia between 1904 and 2004, and 44% of these extended into Victoria (Mutze 1989, Singleton et al. 2005).

The occurrence of plagues reflects the extraordinarily high reproductive potential of house mice (Long 2003). They become sexually mature at 50-60 days of age, gestation lasts 13-31 days and young are weaned and leave the nest at 20-23 days. Thus, mice have a generation time of 120 days, and have the potential to produce 11 litters per year (6-7 is more normal) over a lifetime of 1-3 years in the field, or up to 6 years in captivity. Litters can consist of up to 12 young. However, under most field conditions house mice do not reach their potential population growth.

Food availability appears to be the major determinant of population density and growth in house mice (Newsome 1969b, a, Ylonen et al. 2003, Singleton et al. 2005, Singleton et al. 2007b, McClelland et al. 2018). However, food availability may be dependent on other factors such as ambient temperature and precipitation, which have an indirect effect on mouse population densities (Newsome 1969b, a, Singleton et al. 2005, Singleton et al. 2007b, McClelland et al. 2018). Predation and interspecific competition with other rodent species may also limit mouse population size (Sinclair et al. 1990, Innes et al. 1995, Ruscoe et al. 2011, Norbury et al. 2013, Wilson et al. 2018).

In South Australia mice breed seasonally in different habitats according to the amount of available food and shelter (Newsome 1969b, a). Reproduction in house mice is determined by a balance between physiological costs of growth, maintenance and reproduction on the one hand and food availability on the other (Bronson 1979, Manning and Bronson 1990, Speakman 2008). Mice tend to be less common in intact vegetation than they are in disturbed habitats (Wayne et al. 2017a). In grain growing areas, mice use farm buildings and roadside remnant and riparian vegetation as refuges during phases of low population density, and then colonise croplands when conditions become favourable (Newsome 1969b, a). In tilled barley fields mice are undetectable in late winter and early spring when fields are fallow. They move into the fields and breed there as crops grow during late spring and over summer. They decrease precipitously when crops are harvested (Newsome 1969b, a). Seasonality of changes in mouse density vary with different crops and different farm management practices. Management practices that do not disturb burrows and/or provide shelter and food resources for mice (e.g. low tillage farming) allow maintenance of higher mouse

density throughout the year, with less seasonal fluctuation (Caughley et al. 1998, Singleton et al. 2005).

Population outbreaks leading to mouse plagues occur over periods of 12-18 months, followed by a steep population decline to low densities for several years (Boonstra and Redhead 1994, Singleton et al. 2005). Mouse abundance and the likelihood of a population outbreak leading to a plague depend on both extrinsic environmental and intrinsic demographic factors (Mutze et al. 1990, Pech et al. 1999, Krebs et al. 2004). Extrinsic factors include rainfall, food supply and quality, disease and predation (Mutze 2009). Rainfall and food availability are key extrinsic environmental factors that allow increased reproduction over an extended breeding season (Pech et al. 1999, Singleton et al. 2001, McClelland et al. 2018). Rainfall alone explains 70% of the variance in the onset of mouse outbreaks across Victoria and South Australia (Krebs et al. 2004), although the mechanisms through which rainfall influences mouse densities are not fully understood. Timing of rainfall following a drought can be a significant predictor of plague occurrence (Mutze et al. 1990). Crop yield alone is not a good predictor of mouse abundance in fields during autumn, when plague densities reach their peak (Krebs et al. 2004). Factors other than food availability can also influence the likelihood of mouse plagues. Specific rainfall conditions that permit mice to create burrows and nesting sites may stimulate population growth (Pech et al. 1999). In areas where water is limited, food availability alone may not be sufficient to trigger plagues (Ylonen et al. 2003). Disease and predation can also regulate mouse populations (Sinclair et al. 1990, Pech et al. 1999) (Mutze 2009).

These extrinsic factors interact with intrinsic demographic factors to result in mouse plagues. For example, peak autumn mouse density in fields (when plagues usually peak) is best predicted by rainfall combined mouse abundance during the preceding spring (Krebs et al. 2004). When mice are able to persist in fields over the winter prior to an outbreak they are more likely to increase population growth and plague (Mutze 1991). At low population densities mice are territorial, which limits their abundance and allows most females to produce large litters (Singleton et al. 2005, Sutherland and Singleton 2006). During resource pulses, when mouse density increases, mice become nomadic. This releases socially imposed regulation of the population, despite a lower proportion of females producing smaller litters (Singleton et al. 2005, Sutherland and Singleton 2006). The flexible breeding season and social organisation of house mice allow them to rapidly exploit favourable conditions induced by resource pulses (Singleton et al. 2001, Singleton et al. 2005, Sutherland and Singleton 2006, Kaboodvandpour and Leung 2010).

The impacts of house mice are likely to increase with climate change, especially in landscapes where rodents may be constrained by interaction between severe climate and food availability. Population density of mice on sub-Antarctic islands have quadrupled over the past 30 years with increasing temperatures and reduced precipitation (McClelland et al. 2018). Increased temperatures resulted in milder winters which allowed mice to breed earlier and extend their breeding season. The resulting higher densities of mice directly reduced invertebrate densities by up to two-orders of magnitude, with commensurate consequences for ecosystem processes (McClelland et al. 2018). In more temperate areas such as south-eastern Australia mouse plagues have become increasingly frequent over the past 100 years (Singleton et al. 2005, Singleton et al. 2007b). The 10-year rolling mean of the frequency of plagues per year has changed from 0.2 in 1920, to 0.3 in 1960 and 0.6 in 2000. The frequency of mouse plagues has increased significantly in some states since 1980 (Singleton and Brown 1999). These recent increases may be the result of changes in the diversity of crops, cropping intensity and approaches to farming (Singleton et al. 2005). However, the slower, long-term increase in the frequency of plagues may be, at least partly, related to changing climate.

The only recorded plagues in Western Australian wheatbelt have occurred since the 1960s, and the first mouse plague in Tasmania occurred in 2003 (Singleton et al. 2005).

The costs of invasive house mice to agriculture and biodiversity have encouraged considerable effort to develop better techniques to control them. Fertility control via an immuno-contraceptive vaccine has been the focus of research toward management of mouse plagues in Australia for some time (Singleton et al. 2002, Williams 2002, Redwood et al. 2007), but is not yet practicable. A current focus is on using gene editing techniques to control invasive animals, but we are in the early days of this developing technology (Moro et al. 2018). Currently we rely upon strategic deployment of rodenticides to reduce outbreak populations of mice developing into a plague (Jacob and Buckle 2018). Most rodent baits are based on anti-coagulants, which are very effective at killing the target rodents. However, they can also affect non-target species, particularly predators that consume poisoned mice or by leaching into the soil causing environmental pollution (Cox and Smith 1990, Albert et al. 2010, Brink et al. 2018). Rodenticide application can also be prohibitively expensive (Stenseth et al. 2003, Davis et al. 2004, Singleton et al. 2005, Makundi and Massawe 2011). Furthermore, any non-target reduction of mouse predators reduces one of the ecological processes that constrains mouse populations (Singleton et al. 2007a, Brink et al. 2018).

Management to reduce foxes and cats on Yorke Peninsula (Chapter 2) is likely to reduce predation pressure on mouse populations, especially in the absence of native mammalian predators (Chapter 1). This may result in an increase in mouse abundance. Reintroduction of native mammalian predators (Chapter 7) and management to increase predation by raptorial birds are proposed to keep populations of house mice in check and reduce their impact of biodiversity and agriculture (Sharp 2018; Figure 4.1).

4.3 Avian predators for sustainable rodent control

Enhancing the density of predators offers a potential means of controlling agricultural pests without the use of pesticides (Labuschagne et al. 2016, Lindell et al. 2018). Bird predation on a coffee pest saved coffee farmers in Jamaica between US\$44 and US\$310.ha⁻¹ (Kellermann et al. 2008, Johnson et al. 2010). Insectivorous great tits (*Parus major*) lured to apple orchards with nest boxes reduced caterpillar damage by 50% compared to control orchards in Europe (Mols and Visser 2007). Falcons (*Falco novaeseelandiae*) introduced to New Zealand vineyards saved farmers \$234–\$326.ha⁻¹ in grape losses to pest birds (Kross et al. 2012a) and helped redress the declining falcon population (Kross et al. 2012b). Of 48 separate studies, 34 (71%) showed that enhancement of avian predator density reduced crop pests (Lindell et al. 2018). With specific regard to controlling rodents, avian predator enhancement reduced pest rodent abundance by 82% and reduced crop damage by 17% (Figure 4.2)(Labuschagne et al. 2016). Attracting barn owls to mixed fields and orchards using nest boxes provided a profit of \$236.ha⁻¹.year⁻¹ compared to control areas where no nest boxes were provided (Kan et al. 2013). Using barn owls to control pest rodents is more cost-effective per rodent removed than using poisoning and trapping methods (Browning et al. 2016).

Barn owls (*Tyto alba* complex) have been a focus of interest for controlling rodent damage to crops in many countries (Kross and Baldwin 2016, Labuschagne et al. 2016, Huysman et al. 2018). Barn owls occur on every continent except Antarctica and show considerable variation morphology (Del Hoyo and Collar 2014). This variation has resulted in confusion over the classification of barn

owls, and description of more than 30 subspecies of barn owl globally (Higgins 1999, Del Hoyo and Collar 2014). A recent comprehensive molecular phylogenetic analysis has resolved this confusion, and showed that the barn owl complex, in fact, consists of three allopatric species: the American barn owl, *T. furcata*, the western barn owl, *T. alba*; and the eastern barn owl, *T. javanica* (Uva et al. 2018). An endemic subspecies of the eastern barn owl occurs in Australia (*Tyto javanica delicatula*) (Menkhorst et al. 2017, Uva et al. 2018). This improved understanding of the relationships among barn owls across the world has no real implications for rewilding Yorke Peninsula because all three species of barn owl play a similar ecological role in different parts of the world (Roulin 2003, Taylor 2004, Uva et al. 2018).

All members of the *Tyto alba* complex are nocturnal and crepuscular predators of small mammals, especially rodents (Morton et al. 1977, Morton and Martin 1979, Valente 1981, Perrin 1982, De Bruijn 1994, Kross and Baldwin 2016, Muñoz-Pedreros et al. 2016) which also consume birds, lizards, amphibians and large invertebrates (Morton et al. 1977, Morton and Martin 1979, Valente 1981, De Bruijn 1994). Barn owls are capable of swallowing entire prey up to 280g in mass (Morton and Martin 1979). The normal daily food intake of barn owls varies between 42 and 110g of live prey (Morton and Martin 1979, Mikkola 1983, Steyn 1983, Yom-Tov and Wool 1997, Muñoz-Pedreros et al. 2016). In southern Australia non-breeding barn owls commonly consume four 15-20g mice (total 60-80g) per night (Morton 1975, Raid 2012). Breeding pairs of barn owls deliver an estimated 152 prey items to each chick between hatching and fledging (Browning et al. 2016). Two adult barn owls and their offspring will consume 98kg of prey per breeding attempt (Kross and Baldwin 2016).

Barn owls generally consume different prey species in the same frequency as they occur in the landscape (Glue 1967, Jaksic et al. 1982, Alvarez-Castaneda et al. 2004, Tores et al. 2005, Granjon and Traore 2007, Magrini and Facure 2008, Charter et al. 2009, McDowell and Medlin 2009, Meek et al. 2009, Bernard et al. 2010) but they sometimes selectively capture larger species (Perrin 1982, Yom-Tov and Wool 1997, Tores et al. 2005, Askew et al. 2007, Bernard et al. 2010). House mice are the predominant prey of barn owls in southern Australia, even when they have access to other species of small mammals (Morton 1975, Morton and Martin 1979, Dickman et al. 1991).

Barn owls search for prey from perches or while flying over open areas (Baker-Gabb 1984, Lenton 1984, Arlettaz et al. 2010). Although they are found in a variety of habitats including desert pavement (gibber), agricultural fields, urban areas, tropical savannah and boreal pine forests (Steyn 1983, Higgins 1999, Salvati et al. 2002, Taylor 2004, Bond et al. 2005a), they preferentially hunt in areas of open vegetation where they can best detect and capture prey (Baker-Gabb 1984, De Bruijn 1994, Salvati et al. 2002, Arlettaz et al. 2010, Hindmarch et al. 2017).

Barn owls nest in a wide range of natural and artificial cavities, including tree hollows, caves, rooves and wall cavities (Higgins 1999, Taylor 2004, Meyrom et al. 2009). They can breed when they are seven months old (Marti 1997, Higgins 1999). Most breed for only one year, but some may breed for up to seven years (Marti 1997). A wild Australian barn owl lived for 7 years and 8 months (ABBBS 2019). Hatching success varies from 69% to 72% (Lenton 1984, Mahmood-UI-Hassan et al. 2007) and fledgling success varies from 50 to 68% (Roulin 2002b, Hindmarch et al. 2014). Survival of fledged barn owls varies from 17-35% in their first year (De Bruijn 1994, Altwegg et al. 2003), and climate influences juvenile and adult survival. In the northern hemisphere barn owls commonly die during harsh winters because snow cover makes it difficult to locate prey (De Bruijn 1994).

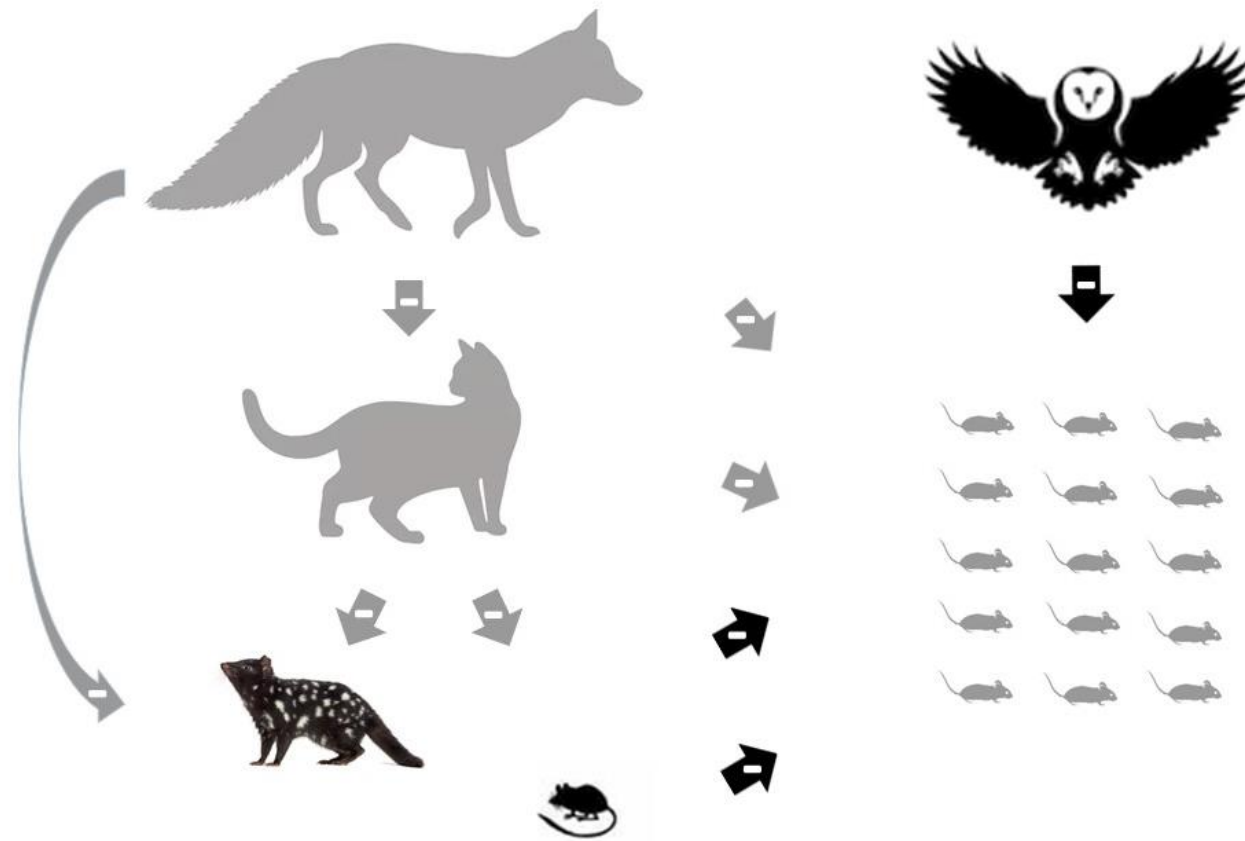


Figure 4.1: Hypothesized impact on mice of reducing fox and cat populations, reintroducing western quolls and red-tailed phascogales and increasing the abundance of barn owls on southern Yorke Peninsula. Arrows show predatory relationships between each species. Organisms that will decrease in abundance are shown in grey. Organisms that will increase in abundance are shown in black.

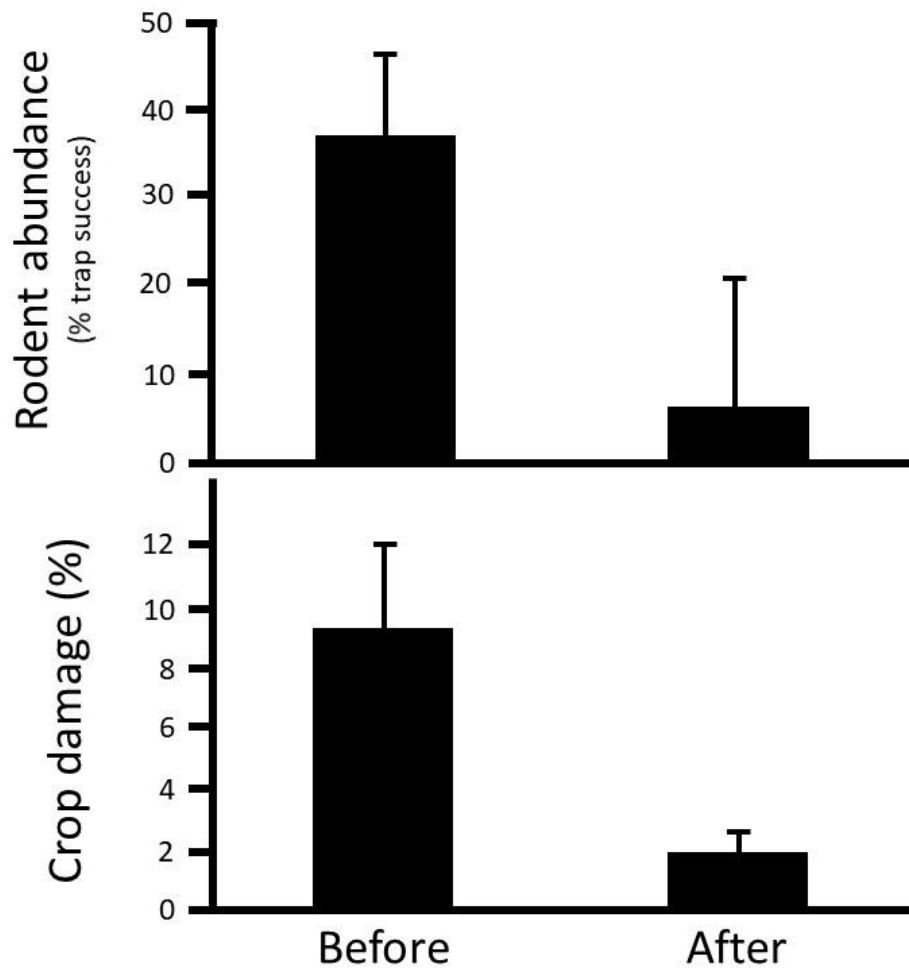


Figure 4.2: The effect of enhancing populations of avian predators on rodent abundance (upper) and crop damage due to rodents (lower). The difference between the left-hand value (before predators increased) and the right-hand value (after predators increased) indicates the influence of enhancing bird populations. Modified from Labuschagne et al. (2016).

Barn owls breed seasonally in many parts of the world (Marti 1997, Martínez and López 1999, Walk et al. 1999, Taylor 2004, Meyrom et al. 2009, Charter et al. 2012, Chausson et al. 2014). In Australia they breed seasonally when rodents are not plaguing but can breed continuously in response to rodent plagues (Olsen and Marples 1993, McLaughlin 1994, Higgins 1999). Clutches contain between one and twelve eggs, with means of between three and seven in different populations (Olsen and Marples 1993, Marti 1997, Higgins 1999, Martínez and López 1999, Roulin 2002b, Mahmood-Ul-Hassan et al. 2007, Hindmarch et al. 2014). Eggs are incubated by the female for 31-35 days and chicks remain in the nest for 60 days before fledging (Higgins 1999, Béziers and Roulin 2016). Males provide food for incubating females and their young (Higgins 1999, Taylor 2004). Fledged young can remain around the nest and be fed by parents for between 30 and 60 days (Higgins 1999, Béziers and Roulin 2016). Thus, a breeding cycle takes about 4-5 months.

In seasonally breeding populations most pairs produce one clutch per year (Lenton 1984, Marti 1997, Martínez and López 1999), but some pairs double-brood (Higgins 1999, Béziers and Roulin 2016). Females double-brood more often than males, by deserting the first brood (leaving the

male to provision the chicks) and commencing a new brood with a different male (Béziers and Roulin 2016). For the most part barn owls are socially monogamous (Smith et al. 1974, Marti 1997, Taylor 2004). Continuously breeding pairs have raised five clutches over two years (Walk et al. 1999).

Breeding in barn owls is strongly influenced by food availability (Salvati et al. 2002, Altwegg et al. 2003, Bond et al. 2005a, Chausson et al. 2014, Hindmarch et al. 2014, Charter et al. 2017). They show many characteristics of species which exhibit brood reduction, an adaptation to unpredictable food availability (Mock 1994, Mock and Forbes 1995). Clutch size is larger when food is abundant (McLaughlin 1994, Higgins 1999). Eggs are laid 2-2.5 days apart and incubation begins when the first egg is laid (Higgins 1999). Consequently, eggs hatch in a staggered manner, producing an age-size hierarchy among nestlings (Roulin et al. 1999, Roulin 2001b). When food is limited, nestlings compete for food and parents selectively feed the oldest chicks (Roulin et al. 1999, Roulin 2001b, Roulin 2001a, Roulin et al. 2009, Hindmarch et al. 2014). Death of the younger nestlings allows adults to adjust their brood size down when food is limited, whereas younger nestlings survive when rodent prey are abundant (Honer 1963, Wilson et al. 1986, Roulin 2002a, Chausson et al. 2014). Thus, breeding success in barn owls tracks food availability (Mock and Forbes 1995, Taylor 2004, Pavlůvčík et al. 2016).

Food availability also determines the abundance of barn owls. Barn owls increase in population density when prey are abundant and decrease in density when available prey declines (Higgins 1999, Pavlůvčík et al. 2016). Population crashes of prey are accompanied by owl deaths due to starvation and dispersal of survivors (Morton and Martin 1979, McOrist 1989). Where their rodent prey plague erratically, barn owl populations can grow quickly, as large numbers of owls move into areas with abundant prey (Higgins 1999). One barn owl travelled 1146 km, from Pitt town in New South Wales to Clare in South Australia over 11 months (Higgins 1999). On these occasions barn owls can be incredibly abundant, as exemplified by an observation of 215 road-killed barn owls on 60 km of road during a rodent plague (Higgins 1999). We could find no population estimates during barn owl eruptions in Australia. Resident population densities vary from 0.02 – 2.58 barn owls.km² in various parts of the world (De Bruijn 1994, Salvati et al. 2002, Altwegg et al. 2003, Meyrom et al. 2009). The single population density estimate we could find for barn owls in Australia was 20.km² in urban Williamstown, Victoria (Higgins 1999). Unsurprisingly the home range of barn owls varies widely from 0.006 to 28.5 km², depending on or prey density, quality of foraging habitat and the availability of nest sites (De Bruijn 1994, Salvati et al. 2002, Hafidzi and Na Im 2003, Taylor 2004, Arlettaz et al. 2010, Hindmarch et al. 2017).

In areas where nesting sites are limited, provision of artificial nesting sites (nest boxes) has been used to increase the density of barn owl populations in many parts of the world for over fifty years (Duckett 1976, Colvin 1985, Marti 1994, Parshad 1999, Antkowiak and Hayes 2004, Puan et al. 2012, Kan et al. 2013). Nest boxes attract barn owls to breed and build up the local population density for owl conservation, to reduce disease risk and/or to reduce rodent damage to agricultural crops (Taylor 2004, Labuschagne et al. 2016). Suitably designed and placed nest boxes can be rapidly colonised and have high occupancy rates (Altwegg et al. 2003, Meek et al. 2009, Muñoz-Pedreros et al. 2010, Frey et al. 2011, Charter et al. 2012, Browning et al. 2016, Labuschagne et al. 2016). Provision of nest boxes in a forested region of Chile resulted in an increase in barn owl density from 0.02.km² to 1.81.km² and a decline in rodent capture success from 12.9% to 0.5% (Muñoz-Pedreros et al. 2010). Similarly, provision of nest boxes in a semi-arid region of Israel increased barn owl density from 1.1.km² to between 2.06 and 4.61.km² (Meyrom et al. 2009). Nest boxes have also been installed to enhance barn owl populations for rodent control in the USA (Raid 2012), Spain (Paz et al. 2013), and Malaysia (Wood and Fee 2003), but there are no quantitative measures of their

effectiveness. Barn owls exhibit negative density dependence in survival and reproductive success (Smith et al. 1974, Altwegg et al. 2003, Meyrom et al. 2009), such that the optimum density of nest boxes is probably site specific. Optimising the density of nest boxes (1.61 km apart versus 1.0 km apart) in Israel is likely to increase the occupancy of boxes by barn owls (61% versus 46%), increase production of alfalfa ($17.07 \text{ t}^{-1} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ versus $14.38 \text{ t}^{-1} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$) and increase income (\$US 845. $\text{ha}^{-1} \cdot \text{yr}^{-1}$ versus \$US 236. $\text{ha}^{-1} \cdot \text{yr}^{-1}$) (Kan et al. 2013).

A second method to attract barn owls uses perches to encourage them and other birds of prey to hunt over target areas (Kay et al. 1994, Hawlena et al. 2010, Salo et al. 2010). An advantage of perches is that they are used by both nocturnal and diurnal birds of prey. On Yorke Peninsula nankeen kestrels (*Falco cenchroides*) and black-shouldered kites (*Elanus axillaris*) are important diurnal predators of house mice. Perch-hunting birds of prey have been attracted by provision of artificial perches. In Australia, installation of artificial perches surrounding cropland attracted both diurnal raptors and barn owls and resulted in a reduction in mouse density in treatment sites compared to control sites, although a corresponding reduction in crop damage was not detected (Kay et al. 1994).

Sinclair et al. (1990) reported that predation by mammalian and diurnal avian predators helped regulate mouse populations at low densities in Australia, delaying the formation of plagues, but were unable to regulate high-density mouse populations. A review of predator manipulations showed similar results in a wide variety of ecosystems around the world (Salo et al. 2010). This may be due to a combination of limited nesting sites, and a time lag in the adjustment of owl density to changes in prey abundance during plagues. In theory, increasing predation pressure on mice should further delay the formation of plagues, and potentially regulate population growth, and reduce the magnitude of plagues (Sinclair and Krebs 2002). Achieving predation pressure sufficient to suppress the magnitude of mouse plagues on Yorke Peninsula is more likely through the combined effects of reintroducing small mammalian predators (Figure 4.1) in addition to increasing the density of barn owls on Yorke Peninsula.

Barn owls are one of two species of owl found on Yorke Peninsula (Neagle 2008). Barn owls occur in more open habitats (low woodland, mallee, agricultural fields), whereas boobook owls (*Ninox boobook*) occur in more wooded areas where they require dense foliage for daytime shelter (Neagle 2008, Menkhorst et al. 2017). Boobooks primarily prey on invertebrates, although they will also eat birds, bats, and occasionally terrestrial reptiles and mammals (Higgins 1999).

Barn owls are a totemic ancestral bird in 'Narungga culture called 'winta (Tindale 1936) or 'winda (Berndt 1939).

4.4 Knowledge gaps

Studies of avian predation of rodent pests are few, generally short-term and often lack sufficient experimental rigor to detect and measure changes in rodent abundance and impacts on agricultural production (Labuschagne et al. 2016). Most studies are descriptive, rather than experimental. Many 'experimental' manipulations of predator density are difficult to interpret because they do not provide data on predator and/or prey population density before and after manipulation, do not include paired experimental and control treatments to take into account temporal variation, are not replicated and occur over limited periods of time (Labuschagne et al. 2016). Short term studies (< 10 years) do not allow evaluation of the sustainability of rodent pest suppression (Lindenmayer et al. 2015, Labuschagne et al. 2016). This is particularly true in the context of managing mice in Australia,

where most agricultural damage is caused when mice plague (Singleton et al. 2005, Singleton et al. 2007b). The proposed enhancement of barn owl density on Yorke Peninsula provides an opportunity to address shortcomings in previous studies with replicated control-treatment studies that quantitatively investigate the effects of providing nest boxes to enhance avian predation on house mice and their impact on agricultural productivity. This is necessary to demonstrate the usefulness of manipulating avian predators for rodent pest management (Kross and Baldwin 2016, Labuschagne et al. 2016).



Figure 4.3: Barn owl delivering a mouse to a nest box on Yorke Peninsula. Photo: Kelly Meany

Barn owls could potentially prey on smaller species or juveniles of larger native mammals that are to be reintroduced to Yorke Peninsula. Barn owls prey on native rodents, juvenile bandicoots, and small dasyurids (Eckert 1971, Morton et al. 1977, Valente 1981, Smith and Cole 1989, Heywood and Pavey 2002). Reintroduced phascogales, bandicoots and native rodents on York Peninsula may be subject to some predation from barn owls, should they become particularly abundant in open habitats frequented by barn owls. However, optimal foraging theory suggests barn owls would focus on the high densities of house mice in fields and nearby refugia (Stephens and Krebs 1986, Stephens et al. 2007). Nevertheless, monitoring of diets of barn owls and small mammal abundances surrounding owls' nesting boxes should be undertaken regularly as part of routine data collection.

The simultaneous use of rodenticides and nest boxes has the potential to reduce the growth of barn owl population due to non-target impacts of rodenticides (Brink et al. 2018). Monitoring the presence of rodenticide residues in mice, barn owls and other raptorial birds should be undertaken to determine if there is a conflict between different approaches to rodent control among on Yorke Peninsula. Should such conflict be found, a resolution may require a focussed campaign of community engagement.

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5 Re-establishing ecological processes: soil engineers

Ecosystem engineers are organisms that control the availability of resources to other organisms by making physical changes to the environment (Jones et al. 1994, Jones et al. 1997). Ecosystem engineers may influence the abundance of other species by directly providing physical structures (e.g. termites and fungi create hollows in mature trees) (Jones et al. 1994). They may also change the physical environment in a way that influences the abundance of other species (e.g. by making burrows or nests that are used by other species) (Jones et al. 1994). Ecosystem engineers influence the abundance of other species, but responding species have minimal or no effect on the ecosystem engineers (Dickman 2006a).

Many animals dig burrows for shelter and/or dig pits and discard soil heaps as they forage for food below the soil surface and are regarded as soil engineers (Whitford and Kay 1999, Davidson et al. 2012, Coggan et al. 2018, Davidson et al. 2018, Valentine et al. 2018). Burrowing mammals support other species that use their burrows for shelter, are attracted to above-ground spoil mounds, or prey upon the animals using the burrows (Davidson et al. 2012, Davidson et al. 2018). Foraging pits can improve water infiltration by penetrating hydrophobic soil surfaces (Garkaklis et al. 1998, Valentine et al. 2017), assist nutrient cycling and promote soil microbiome health by mixing organic matter into the soil (Alkon 1999), act as a sink for organic matter by trapping sediment, leaf litter and seeds (James and Eldridge 2007, Newell 2008, James et al. 2009), alter soil fertility (Garkaklis et al. 1998, Eldridge and Mensinga 2007, James et al. 2009, Hagenah and Bennett 2013), and create favourable microsites for seedling germination (Johnson 1996, Alkon 1999, Martin 2003, James et al. 2010b, Travers et al. 2012, Fleming et al. 2014, Valentine et al. 2017, Valentine et al. 2018). Burial of leaf litter speeds decomposition (Beare et al. 1992, Austin et al. 2009) and the digging or raking activities of heteromyid rodents, echidnas and mallee fowl enhance litter decomposition (Eldridge et al. 2012, Smith et al. 2016a, Travers and Eldridge 2016) and facilitate a change in soil nutrients (Platt et al. 2016). By mixing leaf litter into the soil when digging, marsupials can also reduce fire fuel loads (Martin 2003, Hayward et al. 2016).

The importance and effectiveness of each species of soil engineer can vary between landscapes and with their foraging mode and intensity (Yu et al. 2017). For example, the burrows of African mole-rats (Bathyergidae) accumulate more nitrogen compared to undisturbed soil (Hagenah and Bennett 2013), whereas nitrogen is depleted in the mounds of North American pocket gophers (*Thomomys talpoides*) (Yurkewycz et al. 2014). Foraging pits created by European badgers (*Meles meles*) and Australian burrowing bettongs (*Bettongia lesuer*) and greater bilbies (*Macrotis lagotis*) accumulate potassium (James et al. 2009, Kurek et al. 2014), whereas there is no effect on potassium or phosphorus levels in foraging pits created by Australian brush-tailed bettongs (*B. penicillata*) (Garkaklis et al. 2003). The effects of diggings can be greater in harsher, arid environments where increased resources within diggings might be more important than in more benign environments (Crain and Bertness 2006). The effectiveness of diggings may vary with soil type. For example,



Figure 5.1: The two soil engineers chosen for reintroduction to southern Yorke Peninsula: southern brown bandicoot (*Isoodon obesulus*; above) and a brush-tailed bettong (*Bettongia penicillata*; below). Photographs from Taggart (2014).

diggings are less effective in loose dune sand where rain soaks in readily compared with diggings in harder soils where water does not naturally permeate (Alkon 1999). Most studies demonstrating the ecological role of diggings have been in semi-arid or arid areas (Whitford 1998, Whitford and Kay 1999, Eldridge and Mensinga 2007, James and Eldridge 2007, James et al. 2011) or in arid areas with large areas of rock to focus runoff into the diggings (Yair and Shachak 1982). This suggests that reintroduced ecosystem engineers would have a considerable benefit on limestone rich soils on Yorke Peninsula.

The role that digging mammals play in dispersing mycorrhizal fungi is of particular importance for ecosystem restoration in areas of low soil fertility such as southern Yorke Peninsula (see sections 1.3.2 and 1.3.4). Mycorrhizal fungi form symbiotic relationships in the roots of many native plant species and increase the plants' ability to extract nutrients (e.g., nitrogen and phosphorus) and water from soil (Smith and Smith 2011, Tay et al. 2018). About 70% of vascular plants form symbiotic associations with arbuscular mycorrhizae (Brundrett and Tedersoo 2018). Several digging mammals consume underground fruiting bodies of these fungi (truffles) and assist dispersal of fungal spores in their faeces (Claridge and May 1994, Johnson 1996, Garkaklis et al. 2000, Martin 2003, Noble et al. 2007b, Tay et al. 2018). In this way they assist mycorrhizal colonisation of roots (Aguilera et al. 2016, Dundas et al. 2018). The presence (or the removal) of digging animals influences the composition of fungal (Clarke et al. 2015) and vegetation communities (Whitford and Kay 1999, Davidson et al. 2012).

Rabbits are an introduced mammal (Rolls 1969, Munday 2017, Iannella et al. 2018) that burrows and digs while foraging (Rolls 1969, Bird et al. 2012, Munday 2017). Do they effectively replace native soil engineers? The short answer is no. Rabbit diggings are not structurally or functionally equivalent to foraging pits dug by bilbies and bettongs (James and Eldridge 2007, James et al. 2011) and rabbits don't assume the soil engineer role of native marsupials. Foraging pits made by rabbits are shallower and different in shape compared to marsupial diggings, making them less effective litter and seed traps (Sparkes 2001, James and Eldridge 2007). Rabbits are not known to consume fungi (Robley et al. 2001, Ryan et al. 2003), so are not likely to disperse fungal spores. The damage to vegetation they consume outweighs any beneficial effects of their diggings (Ryan et al. 2003, Noble et al. 2007b, Bird et al. 2012). Along with introduced predators (chapter 2), rabbits are one of the reasons why native soil engineers have disappeared over much of Australia. Rabbits have caused widespread degradation of vegetation (Rolls 1969, Lange and Graham 1983, Eldridge and Simpson 2002, Mutze 2016) and mammal communities in Australia (Rolls 1969, Pedler et al. 2016).

Different species of digging animals vary in the ways they affect soils, fungal and vegetation communities, suggesting that some redundancy is required to realise the full advantages of soil engineers in a landscape. For example, both bandicoots and echidnas (*Tachyglossus aculeatus*) turnover large volumes of soil while foraging, but their diggings capture different volumes of leaf litter and host different assemblages of microbes (Travers and Eldridge 2016, Eldridge et al. 2017a). This suggests that the ecological functions of echidnas and bandicoots as soil engineers do not overlap. We could find no comparison of bandicoots and potoroid roles or effectiveness as soil engineers, but bandicoots tend to make narrower foraging pits than potoroids (Newell 2008).

Many Australian mammals dig burrows for shelter and/or dig into the soil while foraging (Fleming et al. 2014), but most have declined in abundance or suffered range reductions since European arrival in Australia (Burbidge and McKenzie 1989a, Short and Smith 1994, Johnson 2006a, Woinarski et al. 2015) and their absence from landscape has contributed to declining vegetation communities (Martin 2003, Fleming et al. 2014). Observations made prior to the widespread loss of

Australian mammals mention “several acres of ground covered with...[bettong]...holes (Kreff 1866), something which is now only observed where digging marsupials have been reintroduced (Martin 2003).

Of the seven mammals known or likely to be soil engineers on southern Yorke Peninsula (Table 1.1), six are locally extinct (McDowell et al. 2012). Only the echidna remains (Aitken 1973, Brandle 2008). The southern brown bandicoot and brush-tailed bettong have been chosen for reintroduction onto Yorke Peninsula (Table 1.2) because they were identified as keystone soil engineers that will help restore landscape functionally (Taggart 2014, Sharp 2018).

5.1 Southern brown bandicoot

Southern brown bandicoots (*Isoodon obesulus*) are small, omnivorous, ground-dwelling marsupials (Figure 5.1) (Braithwaite 1983, Paull 2008). They forage opportunistically on subterranean fungal fruiting bodies, seeds, invertebrates, small vertebrates and nectar (Quin 1985, Claridge and May 1994). Most studies have reported them to be primarily nocturnal (Heinsohn 1966, Watts 1974, Braithwaite 1977), but they can be active during the day (Lobert 1990).

Until recently the southern brown bandicoot included populations in south-eastern and south-western Australia (Menkhorst and Knight 2011). The south-western population is now recognised as a separate species, the quenda (*Isoodon fusciventer*) (Travouillon and Phillips 2018). Of the eight species of bandicoot that used to occur in South Australia, the southern brown is the only one that has not gone extinct in the state (Kemper 1990, Owens and Graham 2009). They still occur in the south-east of the state, the Mount Lofty Ranges, on Kangaroo Island, Eyre Peninsula and Islands of the Nuyts archipelago (Copley et al. 1990, Paull 1995, Owens and Graham 2009).

Southern brown bandicoots were historically common across their range (Paull et al. 2013). They are now regarded as endangered Federally, and vulnerable in South Australia (Taggart 2014), but at least concern with a decreasing population trend internationally (Burbidge and Woinarski 2016a). Threats include predation by introduced foxes and cats, habitat loss and fragmentation and inappropriate fire regimes (Stoddart and Braithwaite 1979, Paull 1995, Short and Calaby 2001, Coates 2008, Bilney et al. 2010, Brown and Main 2010, Long 2010, Paull et al. 2013, Burbidge and Woinarski 2016a).

Southern brown bandicoots occur in heath, shrubland, sedgeland, and woodland vegetation communities that are usually associated with well-drained soils (Heinsohn 1966, Stoddart and Braithwaite 1979, Opie et al. 1990, Paull 1995, Coates 2008, Haby 2013, Paull et al. 2013, Robinson et al. 2018). Within these vegetation types bandicoots typically inhabit areas with a dense shrubby understorey and preferentially use *Xanthorrhoea* species for shelter (Haby 2013, Paull et al. 2013, Robinson et al. 2018). The dense understorey provides cover from predators (Coates 2008) and *Xanthorrhoea* provide nesting material (Haby 2013). They shelter above ground in ‘nests’ ~40-50cm across with a tunnel opening, or in burrows constructed by other animals but are capable of digging their own burrows (Long 2009, Haby 2013, Robinson et al. 2018). Foraging often occurs in more open vegetation (Stoddart and Braithwaite 1979, Haby 2013).

Southern Brown bandicoots can be more abundant in disturbed environments than in uncleared native vegetation, so long as suitable habitat components are present (Packer et al. 2016, MacLagan et al. 2018). Introduced European blackberries (*Rubus fruticosus*) provide suitable shelter

for southern brown bandicoots in highly modified habitats (Packer et al. 2016). Habitat clearance and fragmentation limits dispersal of southern brown bandicoots (Li 2013). Natural disjunctions occurred between areas of suitable climate and vegetation that separated populations (Paull et al. 2013). These disjunctions have been masked by historical habitat loss and fragmentation (Paull et al. 2013).

Early investigations into the habitat relationships of southern brown bandicoots suggested a preference for early successional stage vegetation after disturbance (Stoddart and Braithwaite 1979). However later studies found no such pattern (Lobert 1990, Lobert and Lee 1990, Catling et al. 2001, Shan et al. 2006, Hope 2012), and other studies showed reduced use of recently-burnt sites (Claridge and Barry 2000, Dawson et al. 2007, Brown and Main 2010, Paull et al. 2013). Current information suggests that vegetation structure and soil type have a greater influence on bandicoots than fire history (Brown and Main 2010, Paull et al. 2013).

Southern brown bandicoots have a high reproductive potential (potential $r_{\max} = 4.28$; Cuthbert 2010). They reach sexual maturity in their first breeding season at 7 months of age (Lobert and Lee 1990). Gestation is not precisely known, but is less than 15 days (Lobert and Lee 1990), similar to the 12.5 days in *I. macrourus* (Lyne 1974). Young vacate the pouch when 47 to 55 days old and are weaned at 60 days (Heinsohn 1966, Stoddart and Braithwaite 1979). Litters consist of 1-6 pouch young, with 2-4 being more usual (Heinsohn 1966, Stoddart and Braithwaite 1979, Copley et al. 1990, Lobert and Lee 1990). Some populations breed throughout the year (Stoddart and Braithwaite 1979, Copley et al. 1990, Lobert and Lee 1990), whereas others breed seasonally (Heinsohn 1966, Stoddart and Braithwaite 1979, Lobert and Lee 1990). In seasonal populations they mate in July-August, have pouch young in August-December (Heinsohn 1966, Stoddart and Braithwaite 1979, Lobert and Lee 1990). Females can produce four to five litters per year in continuously breeding populations (Copley et al. 1990), but in seasonally breeding populations they normally produce one, sometimes two, litters per year (Heinsohn 1966, Stoddart and Braithwaite 1979, Lobert and Lee 1990). Breeding corresponds with periods of rain (Copley et al. 1990) and high food abundance (Heinsohn 1966, Lobert and Lee 1990).

The sex ratio of pouch young (1.41) is biased toward the smaller females, but the sex ratio of adults is biased toward males (Copley et al. 1990). This appears to be due to lower survival of females between weaning and their first breeding season (Lobert and Lee 1990). Between 12 and 18% of young born reach maturity (Stoddart and Braithwaite 1979). Both males and females live for up to 3.5 years (Lobert and Lee 1990). Females grow until they are two years old, whereas males grow until they are three years old, and then both sexes reach an asymptote (Lobert and Lee 1990).

Southern brown bandicoots are sexually dimorphic, with males being larger (500–1500 g) than females (400–1000 g) (Menkhorst and Knight 2011). They live largely solitary lives (Heinsohn 1966), and can be aggressive toward conspecifics (Jones 1924, O'Callaghan 1974). Individuals may be territorial at low population densities if resources are defensible and intruder pressure is low, but occupy overlapping ranges if population density is high, as with *I. fusciventer* (Broughton and Dickman 1991). Males have larger average home ranges (2.1–25.0 Ha) than females (1.5–11.0 Ha) with complete overlap between sexes but limited overlap within sexes (Heinsohn 1966, Copley et al. 1990). Population densities vary between 1.3 and 1.4 individuals.Ha⁻¹ (Copley et al. 1990).

Quenda and southern brown bandicoots are very similar in foraging behaviour. All studies of the soil engineer role of brown bandicoots have been done on quenda in Western Australia, but we assume their conclusions are transferrable to southern brown bandicoots. While digging for food, quenda create characteristic conical-shaped foraging pits (~100 mm across and 70 mm deep), with soil

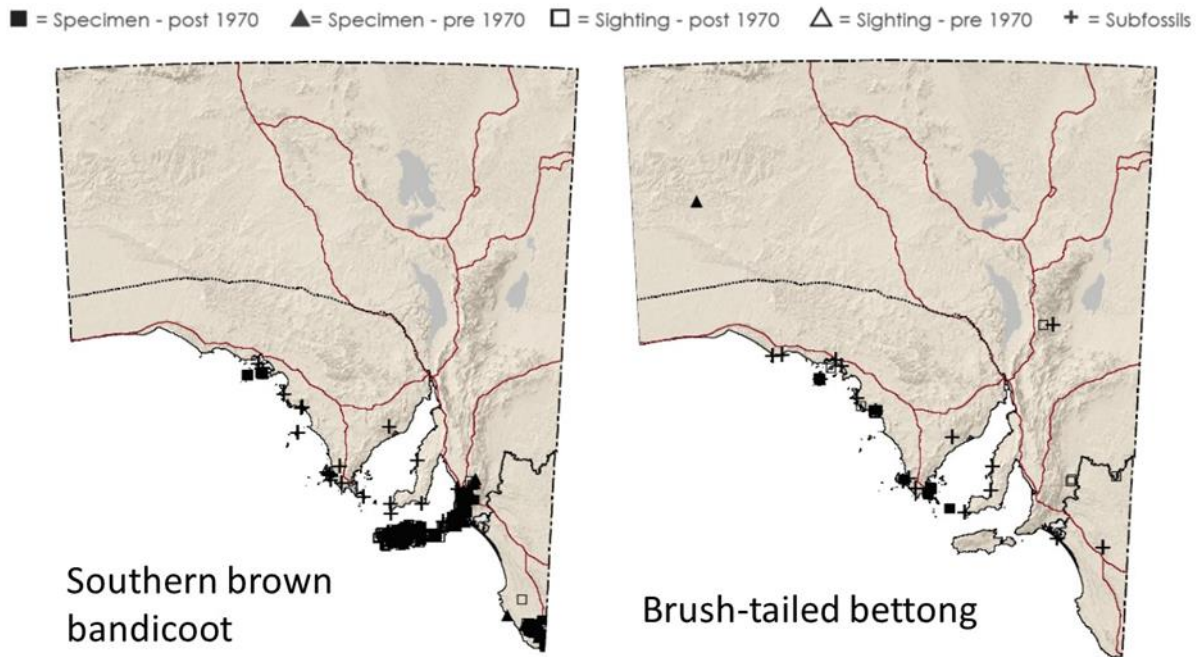


Figure 5.2: Known distribution of southern brown bandicoot and brush-tailed bettong in South Australia. All records of brush-tailed bettong specimens since the 1970s (■) represent translocated populations. From Owens & Graham (2009) .

and any litter present (Valentine et al. 2012). Each individual bandicoot creates ~45 pits per night and displace nearly four tonnes of soil per year (Valentine, Anderson, Hardy, & Fleming, 2013). Foraging pits increased water infiltration, contained higher amounts of leaf litter, nutrients, mycorrhizal fungi and enhanced seedling recruitment and growth compared to undug ground (Valentine et al. 2017, Valentine et al. 2018).

As with other bandicoots, southern browns have a high tolerance for 1080 ($LD_{50} \sim 7.0\text{mg.kg}^{-1}$) (McIlroy 1983), meaning a 1kg bandicoot would have to consume three standard dried meat fox baits (3mg 1080) or two ERADICAT baits (4.5mg 1080) to be exposed to their LD_{50} . Field and laboratory trials have shown that the closely related quenda consume sausage-style baits (and the hard-shelled delivery vehicle containing the toxin) used to control cats (de Tores et al. 2011). Despite the potential risks of bandicoots taking baits, fox baiting programs have resulted in increases in bandicoot population density (Coates 2008, Dexter and Murray 2009, Brown and Main 2010). This suggests that the risks of taking baits to bandicoots are far outweighed by the advantages of reducing predation by foxes.

Short (2009) reports twelve reintroductions of the quenda (75% success), and two of southern brown bandicoot (100% success) (Short 2009). In addition, eleven wild-caught southern brown bandicoots were reintroduced to Booderee National Park in New South Wales in 2016 (Robinson et al. 2018). No mortality was detected, and all females captured six months after release ($n = 4$) carried 3-4 pouch young. Southern brown bandicoots have been returned to the wild successfully following a period of time in captivity (Cooper 2011).

Subfossil southern brown bandicoots occur in the Balgowan, Innes National Park, Troubridge Hill assemblages and on Althorpe Island (Table 1.1; Figures 1.7 & 5.2)(Owens and Graham 2009, McDowell et al. 2012), but there have been no historical records of the species on Yorke Peninsula (Aitken 1973, Kemper 1990, Brandle 2008, Owens and Graham 2009). Distribution models indicate that Yorke Peninsula currently has a suitable climate for southern brown bandicoots (Paull et al. 2013).

5.2 Brush-tailed bettong

Brush-tailed bettongs are small (1-1.5kg) native marsupials with strongly clawed fore-feet used for foraging and nest making (Figure 5.1). Their distinctive prehensile tail has a black brush end and is used to carry strands of seagrass, grass and/or bark as nesting material (Christensen and Leftwich 1980, Claridge et al. 2007). They rest during the day in well-concealed nests built above ground in dense vegetation or in the burrows of other animals (Christensen and Leftwich 1980, Claridge et al. 2007).

Bettongs and their close relatives (subfamily Potoroinae) are the sister group to all other species of kangaroos and wallabies (Meredith et al. 2009, Prideaux and Warburton 2010). Seven species of bettong (genus *Bettongia*) were among the most widespread marsupials in Australia at the time of European settlement (Claridge et al. 2007, Menkhorst and Knight 2011, McDowell et al. 2015a, Haouchar et al. 2016). Three of them are now extinct, and the remaining four persisting only in highly fragmented populations (Woinarski et al. 2014a).

The brush-tailed bettongs are a distinctive subgroup of bettongs, which is generally thought to consist of three extant species (Claridge et al. 2007, Menkhorst and Knight 2011, McDowell et al. 2015a). The endangered northern bettong (*Bettongia tropica*) occurs in north-eastern Queensland. The southern bettong (*Bettongia gaimardi*) is extinct in the coastal south-eastern mainland, but still occurs in Tasmania. The brush-tailed bettong (*Bettongia penicillata*) is extinct in south-eastern Australia inland of the Great Dividing Range and critically endangered in south-western Australia. Holocene fossils appeared to indicate that in pre-European times *B. penicillata* had a continuous distribution between south-eastern and south-western Australia (Owens and Graham 2009, McDowell et al. 2012, Haouchar et al. 2016). Anecdotal evidence also suggested that *B. penicillata* once occurred northward as far as the Great Sandy Desert (Burbidge et al. 1988), but this may involve confusion with the desert bettong (*B. anhydra*)(McDowell et al. 2015a).

This information was the basis for previous reintroductions of brush-tailed bettongs to South Australia from the only remaining populations of *B. pennicillata* in Western Australia (Delroy et al. 1986, Copley 1994, Orell 2004, Short 2009, Finlayson et al. 2010, Yeatman and Groom 2012). However, DNA from subfossil and recent brush-tailed bettongs demonstrated five clearly defined clades in the group (not three, as previously recognised)(Haouchar et al. 2016). The clades all diverged from each other around 2.5 million years ago, suggesting they should be conservatively interpreted as four closely related species. Importantly no South Australian subfossils from which DNA was extracted belong to *B. pennicillata*. Subfossils from western South Australia belong to a previously unrecognised taxon and a subfossil from Kangaroo Island clustered with *B. gaimardi* from Tasmania (Haouchar et al. 2016). This new understanding of brush-tailed bettongs in South Australia is at variance with translocations of brush-tailed bettongs from Western Australia into South Australia (Delroy et al. 1986, Copley 1994, Short 2009, Finlayson et al. 2010, Yeatman and Groom 2012).

Despite this revelation, the translocated *B. penicillata* in South Australia are important for the continued future existence of that species following their recent precipitous declines in Western Australia (Yeatman and Groom 2012, Wayne et al. 2015), and all types of brush-tailed bettongs appear to play very similar ecological roles.

Bettongia penicillata became the first Australian taxon to be removed from threatened species lists in 1996 as a result of active conservation efforts (Yeatman and Groom 2012). However, since 1999 the remaining fragmented populations on Western Australia have undergone a rapid decline of over 90% (Wayne et al. 2015). The reason(s) for the decline is not known, but the leading hypotheses include disease (Wayne et al. 2015) and predation by cats, following reduction in fox abundance (Marlow et al. 2015a). Less than 2000 individuals remain in the wild and they are one of the most threatened marsupials in Australia (Wayne et al. 2015).

The various brush-tailed bettong species occur in a dry, grassy open forest from tropical Queensland (*B. tropica*) to cold-temperate Tasmania (*B. gaimardi*), and dry sclerophyll forest with a dense understory in Western Australia (*B. penicillata*) (Menkhorst and Knight 2011). Populations of *B. penicillata* translocated to South Australia occur on fox-free islands or fenced peninsulas (Yeatman and Groom 2012) with a variety of coastal vegetation communities, including *Eucalyptus diversifolia* mallee heath that occurs on southern Yorke Peninsula (South Australian Department for Environment and Heritage 2006, Neagle 2008).

Brush-tailed bettongs can breed continuously throughout the year (Sampson 1971). Most females usually carry pouch young or suckle a young at heel, but the proportion of females caring for young is lower in the drier months. Females exhibit embryonic diapause, so it is possible for females to carry a blastocyst in the uterus, young in the pouch and have a young at heel (Smith 1989; Smith 1996). Females give birth to one young at a time, from six months of age (Sampson 1971, Yeatman and Groom 2012). Gestation lasts 21 days, young remain in the pouch for 90 days, and females can produce up to three young per year (Yeatman and Groom 2012).

Life expectancy for brush-tailed bettongs in the field is 4-6 years, but they can live up to 14 years in captivity (Yeatman and Groom 2012). They are solitary but mothers may share their nest with a young at heel (Sampson 1971, Christensen and Leftwich 1980). They occupy home ranges, which in size from 6 to 35 Ha, depending on habitat and population density (Sampson 1971, Christensen 1980, Yeatman and Wayne 2015). Males tend to have larger home ranges than females (Sampson 1971, Yeatman and Wayne 2015), and dispersal is strongly male-biased (Pacioni et al. 2011). Males disperse short distances (≤ 9 km) (Christensen 1980) and even populations connected by continuous habitat show short-range genetic differentiation (Pope et al. 2000, Pacioni et al. 2011).

Brush-tailed bettongs consume a variety of roots, tubers, grass, fruits and seeds, bark, insects, arachnids, small vertebrates and fresh carrion, but most of their diet is made up of fungi (Lamont et al. 1985, Claridge and May 1994, Lee 2003, Claridge et al. 2007, Fulton 2017, Zosky et al. 2017). Bettongs are avid meat eaters in captivity (Ride 1970). They consume at least 79 species and 15 genera from 14 families of fungi (Zosky et al. 2017). Fungi are important nutritional component in the diet of bettongs (McIlwee and Johnson 1998), and the amount of fungus in the diet of is tightly linked with body condition and reproductive success (Johnson 1994b, Johnson and McIlwee 1997). Bettongs in a high-density, fenced population eat significantly less fungi than in lower density free-ranging populations (Zosky et al. 2017).

A foraging bettong digs between 38 and 114 holes per night while searching for food, displacing nearly five tonnes of soil per year (Garkaklis et al. 2004b). By excavating and ingesting fungi

bettongs play an important role in dispersing and activating mycorrhizal spores (Johnson 1994a, Dundas et al. 2018), improving nutrient turnover (Garkaklis et al. 2003, 2004b), water penetration of soil (Garkaklis et al. 1998), and plant recruitment (Murphy et al. 2005, James et al. 2010b). Bettongs also cache seeds (Murphy et al. 2005, Chapman 2015). Caching of seeds by brush-tailed bettongs assists dispersal and regeneration of sandalwood (*Santalum spicatum*) which has low rates of regeneration and recruitment in areas where bettongs no longer occur. Where bettongs have been reintroduced there were more sandalwood seedlings and saplings, and they grew further away from parent trees (Murphy et al. 2005).

Their ability to consume meat puts bettongs at risk of consuming baits used to control foxes and cats. Different populations of the brush-tailed bettong group vary in their susceptibility to 1080. *B. gaimardi* are susceptible to 1080 toxicity ($LD_{50} < 1.0 \text{ mg.kg}^{-1}$, similar to most other mammalian herbivores), whereas *B. penicillata* can tolerate doses of 100 mg.kg^{-1} (King et al. 1981). This would suggest that *B. penicillata* would be the best option for translocating brush-tailed bettongs to Yorke Peninsula, given that ongoing predator baiting is likely to be necessary. Populations of *B. penicillata* have grown in areas where foxes have been controlled with 1080 baits (Wayne et al. 2015, Wayne et al. 2017a, Wayne 2018), suggesting any impact of baits is outweighed by the benefits of decreasing predation.

Translocations have played a major role in conservation management of brush-tailed bettongs (Delroy et al. 1986, Copley 1994, Orell 2004, Short 2009, Finlayson et al. 2010). There have been at least 43 translocations of *B. penicillata* to South Australia (n = 14), New South Wales (n = 3) and Western Australia (n=26) since 1977 (Finlayson et al. 2010). Analyses of the limited information on the results of these reintroductions suggests that translocations of captive brush-tailed bettongs have been more successful than those involving wild caught bettongs, translocations to the mainland were less successful than those to islands and translocations to predator free areas were more successful than to areas where predators still occurred (Finlayson et al. 2010). These results must be treated with caution though because captive bettongs have been translocated more often to predator free areas, confounding the influence of these effects on translocation outcomes. Despite a significant proportion of failures, translocations allowed significant recovery of the brush tailed bettong population until 1999.

Subsequent precipitous population declines have occurred but the cause for this is not known (Wayne et al. 2015). Several microorganisms hosted by the *B. penicillata* have been identified recently (Smith et al. 2008, Bennett et al. 2010b, Parameswaren et al. 2010, Kaewmongkol et al. 2011, Botero et al. 2013, Pacioni et al. 2013, Thompson et al. 2013, Hobbs and Elliot 2016, Hulst et al. 2016, Ash et al. 2017, Skogvold et al. 2017), but whether they are related to the decline is not known (Wayne et al. 2015). Any activities involving woylie populations should follow disease risk management protocols (Chapman et al. 2011). Fortunately, periods in captivity for quarantine and disease assessment prior to reintroduction have no impact on body mass or post-release survival in bettongs (Batson et al. 2016).

The soil engineering role of Tasmanian brush-tailed bettongs have been incorporated into the Mulligans Flat Woodland Sanctuary restoration project in the Australian Capital Territory (Shorthouse et al. 2012, Batson et al. 2016). The reintroduction of Tasmanian bettongs into this predator free area appears to have been successful (Batson et al. 2016), but information on their effects on soil and plant communities have not been published.

Brush-tailed bettongs have not been recorded on Yorke Peninsula in historical times (Aitken 1973, Brandle 2008, Owens and Graham 2009), but they occur as subfossils in the Balgowan and

Innes National Park assemblage and west of Minlaton (Table 1.1; Figures 1.7 & 5.2)(Owens and Graham 2009, McDowell et al. 2012). Which species of brush-tailed bettong that occurred on Yorke Peninsula is not known (Haouchar et al. 2016; see above).

5.3 Knowledge gaps

Subfossils indicate that a species of brush-tailed bettong once occurred on Yorke Peninsula (see above). It is most likely to have been either (1) an undescribed extinct species that also occurred on Eyre Peninsula or (2) *B. gaimardi* which used to occur on Kangaroo Island (Haouchar et al. 2016). Studies of DNA from the available subfossil material may resolve this question. In practice, a decision will have to be made whether to continue translocating bettongs to South Australia from the sole remaining population of the critically endangered *B. penicillata* from Western Australia or change to sourcing them from the sole remaining population of *B. gaimardi* in Tasmania. The poorer conservation status and high tolerance to 1080 baits of *B. penicillata* suggest this would be the appropriate choice of bettong for translocation to Yorke Peninsula. Based on our current state of knowledge, the species of bettong chosen for translocation is unlikely to influence the ecosystem benefits they will provide.

The movements, diet and demography of southern brown bandicoots and brush-tailed bandicoots translocated to Yorke Peninsula should be monitored to document their adaptation to an environment from which both have been absent for over a century. Monitoring will allow adaptive responses to any unforeseen problems that may arise.

Both bettongs and bandicoots have broad diets, which occasionally include meat, suggesting they may prey on threatened ground-nesting birds or their eggs. Clay eggs in artificial nests on the ground have been attacked by brush-tailed bettongs (30% of 100 nests)(Fulton 2017) and southern brown bandicoots (3% of 142 nests)(Piper and Catterall 2006). Despite these instances of egg predation on artificial nests, there are no records of bettongs or bandicoots consuming real eggs of ground-nesting birds. A comprehensive analysis of bettong diet from across their range in Western Australia did not find egg or bird remains in faecal pellets or foreguts (Zosky et al. 2017). The natural co-occurrence of bettongs and bandicoots with native birds on Yorke Peninsula prior to European settlement suggest they should not have an overwhelming effect on key ground nesting or threatened birds such as hooded plovers, malleefowl or mallee whipbirds. Nevertheless, care should be taken to monitor these species to ensure translocated mammals do not threaten them.

The key prediction following from the soil engineering roles of bettongs and bandicoots are that they will (1) increase the rate of soil turnover, (2) improve soil nutrient content, (3) improve fungal diversity in the soil, (4) improve plant recruitment and native vegetation recovery, especially following disturbance such as fires. These predictions should be tested by comparing areas to which bettongs and bandicoots have been translocated using a combined before-after and replicated exclusion-control approach.

6 Re-establishing ecological processes: seed consumers and dispersal by native rodents

Dispersal is a key feature of population dynamics. It is how individuals move away from others of their own species to avoid intraspecific competition and by which disturbed patches of habitat are recolonised (Howe and Smallwood 1982). Seeds are the main means of dispersal for 'higher' plants, which are otherwise characteristically immobile (Silvertown 1982). The primary producers (plants) of most land-based ecosystems are maintained through a variety of abiotic and biotic mechanisms of seed dispersal. Seed dispersal by water (hydrochory) is rare in arid and semi-arid areas. Seeds dispersed by wind (anemochory) generally move only short distances from the mother plant, whereas seeds dispersed by animals (zoochory) can move much greater distances (Silvertown 1982).

Seeds that have been shed by plants and reside in the soil are referred to as the seed bank. The seed bank is crucial to maintain functional native plant communities (Silvertown 1982). Seeds remain viable in the seed bank for varying lengths of time, depending on the plant species (Forget and Vander Wall 2001). Soil removed from an eleventh century grave in Denmark contained viable seed after 850 years of dormancy (Odum 1978). There is no necessary relation between the species composition of the seed bank at a site and the above ground vegetation at the same site (Silvertown 1982).

Rodents are one group of mammals that commonly eat seeds and play a critical role in seed dispersal and the dynamics of plant communities. Even though rodents can consume a significant proportion of seeds produced by plants (30-80%; Nelson and Chew 1977), this need not decrease the population of growing plants. This can be so for several reasons. For example, granivores may leave a small excess of seeds produced each year to accumulate over time and have little impact on the total seed stored in the seedbank over the lifespan of long-lived plants (Silvertown 1982). Granivory may also result in a compensatory reduction in density-dependent mortality among seeds when they germinate (Silvertown 1982). Rodents are one group of granivores, along with birds and ants, that consume large amounts of seed (Brown et al. 1979, Andersen 1989, Mills et al. 2018).

The relationships between granivores and the plants whose seeds they consume can often be mutualistic. Ants can be such an important vehicle for dispersal that some plants produce special oil-containing appendages to their seeds, called elaiosomes (Beattie 1985). Elaiosomes attract ants, which carry seeds to their nest. The elaiosome is consumed and the seed is discarded to become part of the underground seed bank (Beattie 1985). Similarly, many plants produce seeds within sweet fruits that attract birds and mammal dispersers (Traveset et al. 2007). The fleshy fruits are consumed, and the seeds expelled by in the animal's faeces, which provide nutrient for germinating seedlings (Traveset et al. 2007).

Granivores may act as dispersers of any seeds they carry away but do not consume. In periods of seed abundance, seeds may be stored in caches for consumption later (Vander Wall and Beck 2012). Seed caching has been described in numerous rodents (Forget and Vander Wall 2001), some of which have specialized cheek pouches for transporting seeds to a cache. In contrast to *in situ* seed

consumption, cached seeds are buried intact and have the potential to germinate and be recruited into the above ground plant community (Campos et al. 2017). Fragments of large seeds discarded by granivorous rodents can retain the ability to germinate as long as the embryo remains intact (Loayza et al. 2014). Seeds handled and buried by animals are more likely to germinate than seeds that have not been handled (Zwolak and Crone 2012). Seed caching by rodents can move seeds to places that are favourable for germination (Hollander and Vander Wall 2004, Wang et al. 2004, Vander Wall and Beck 2012, Loayza et al. 2014), bury seeds at depths where they are protected from fires (Moore and Vander Wall 2015), and disperse seeds to areas of low plant density (Hirsch et al. 2012). Buried seeds are frequently dug up and re-cached multiple times, either by the original hoarder or by pilfering conspecifics. This can result in a step-wise increase in the distance that seed moves from its parent plant (Hirsch et al. 2012). Rodents are important primary seed dispersers, that also enhance nutrient cycling through ecosystems as prey to carnivorous birds, mammals and reptiles (Sarasola et al. 2016). These predators also act as secondary seed dispersers, by transporting seed-laden rodents (Sarasola et al. 2016, Reiserer et al. 2018). Survival from seeds to adult plants can be higher in the presence of rodents than in their absence (Borchert and Jain 1978).

Most of what we know about the ecological role of rodents as seed dispersers comes from research done abroad. Caching by rodents and its relationship with plant dispersal and seedling recruitment is well-understood in Europe and the Americas, in both tropical and temperate forests (e.g. Hollander and Vander Wall 2004, Vander Wall and Beck 2012) and arid zones (e.g. Beck and Vander Wall 2010, Waitman et al. 2012). The importance of rodents as seed dispersers has received very little attention in Australia.

Most studies of seed removal rates in Australia have been done in temperate and arid areas where the native mammal fauna has been severely depleted (Short and Smith 1994, Johnson 2006a, Gordon and Letnic 2015). Unsurprisingly, they found that mammals played a minimal role in seed predation compared to ants and birds (Morton 1985, Predavec 1997, cf. Auld and Denham 1999). The limited role of mammals as seed predators in Australia has been attributed to the lack of specialised granivorous rodents here, as most native rodents are omnivores (Murray et al. 1999, Breed and Ford 2007). More recent work in areas where native rodent populations have been able to recover has shown that mammals were the dominant vertebrate consumers of seeds (Mills et al. 2018).

For many years Australian rodents were not known to cache seeds (Watts and Aslin 1978), but we now know that some do. White-tailed tree rats (*Uromys caudimaculatus*) cache the seeds of a tropical rainforest tree (Theimer 2001) and seed germination is enhanced following removal of the fruit pulp by rodents (Rader and Krockenberger 2007, Elmoultie and Mather 2012). In arid areas spinifex hopping-mice (*Notomys alexis*) may bury seeds and recover them to consume later (Robinson et al. 2003). Studies of seed removal rates by mammals in (Morton 1985, Predavec 1997, Auld and Denham 1999, Mills et al. 2018) did not track the fates of the seeds, leaving the possibility that some of the removed seeds may have been cached. Nevertheless the absence of specialised cheek pouches for carrying seeds in Australian rodents (Watts and Aslin 1978) suggests seed caching is not as prevalent in them as it is among rodents elsewhere.

While they may not play a large role in seed dispersal, native rodents may consume enough seed of specific plants to influence vegetation. The loss of many medium-sized marsupials and native rodents has coincided with an increase in density of unpalatable woody shrubs in Australia (Short and Smith 1994, Noble et al. 2007a). This shrub encroachment is typically viewed as an undesirable environmental change by pastoralists and conservation agencies (Eldridge and Soliveres 2015). The

increase woody shrubs is usually attributed to combined pressures of overgrazing by livestock and rabbits, changed fire regimes and increasing CO₂ levels favouring the growth of shrubs over grasses (Archer et al. 1995). However, the functional extinction of rodents in many parts of Australia may have relaxed their role in limiting recruitment of woody shrubs (Noble et al. 2007a, Gordon et al. 2017, Gordon and Letnic 2019). Where native rodents persist, they are significant consumers of woody shrub seeds (Gordon and Letnic 2015, Mills et al. 2018, Gordon and Letnic 2019).

Five species of Australian rodents have been the subject of reintroductions for biodiversity conservation (Copley 1999, Moro and Morris 2000a, b, Anstee and Armstrong 2001, Moro and Bradshaw 2002, Short 2009, Short et al. 2018). The influence of these rodents on seed dispersal and/or plant recruitment has not been investigated.

Rodents have been important vehicles for rewilding projects in Europe and the Americas (Davidson et al. 2018, Willby et al. 2018). The construction of dams by beavers and extensive burrow systems by prairie dogs make them important ecosystem engineers (Davidson et al. 2018, Willby et al. 2018), but the role of specialist seed-caching rodents in seed dispersal has been less important (Cid et al. 2014, Blythe et al. 2015). Current evidence from Australia suggests that functionally extinct rodents may have played a role in suppressing woody shrubs by consuming their seeds. Evidence for a positive role in seed dispersal is limited but may reflect the historical absence of rodents in the landscape. All of the eight species of native rodents recorded on Yorke Peninsula appear to be locally extinct (Table 1.1; McDowell et al. 2012), although one appears to have been present at the time of European settlement. A mouse called ‘antubatu-’witekatja was reported by a ‘Nurangga informant in the 1890s (Tindale 1936). No native rodents have been found on southern Yorke Peninsula since then, despite considerable efforts to find them over the past forty years (Aitken 1973, Brandle 2008; Graham Medlin, personal communication to Greg Johnston, 2018). Any role in seed dispersal or suppression of woody shrubs played by native mammals has likely been absent from southern Yorke Peninsula for at least 120 years.

6.1 Knowledge gaps

Our knowledge of the role Australian rodents play in seed dispersal and plant recruitment is extremely limited. This reflects a lack of research in this area and should not be taken as evidence that Australian rodents do not perform this role. The limited evidence available suggests that native rodents play no significant role in seed dispersal but may play an important role in suppressing woody shrubs by consuming their seeds. The southern Yorke Peninsula Rewilding project offers an opportunity measure the extent to which Australian rodents influence the viability of native vegetation communities.

The reintroduction of native rodents has a potential to influence the population dynamics of native animals, particularly as occasional opportunistic carnivores. Predation is a major cause of nest failure among birds (Remes et al. 2012) and exotic rodents are a leading cause of bird species declines worldwide (Stattersfield et al. 2000). Australian birds have co-existed with native rodents for 1-2 million years (Smith et al. 2016b) and native rodents prey upon bird nests less frequently than do exotic black rats (*Rattus rattus*), perhaps because native rodents are less arboreal (Smith et al. 2016b). Introduced rodents prey on hooded plover eggs, and there has been one reported instance of a native rodent predating their eggs (Maguire et al. 2014). Rodent predation is considered a spatially localised and low severity threat to hooded plovers. The potential for rodent predation on hooded

plovers and mallee whipbirds should be considered in choosing which species of rodents to reintroduce.

The occurrence of subfossil pale field rat (*Rattus tunneyi*) in the Innes National Park and Troubridge Hill assemblages (McDowell et al. 2012) was based on incomplete dentaries without any teeth, making reliable identification difficult (Graham Medlin, personal communication to G. Johnston, 2018). The inclusion of this northern Australian species which has not been recorded elsewhere in South Australia (Owens and Graham 2009, Menkhorst and Knight 2011) needs confirmation before it is considered for reintroduction to Yorke Peninsula.

7 Re-establishing ecological processes: native predators

Predators play a key role in maintaining species diversity and ecosystem functions (Ripple and Larsen 2000, Soule et al. 2003, Glen and Dickman 2005, Ripple and Beschta 2007, Glen and Dickman 2014a). They do this by directly or indirectly regulating top-down processes at all trophic levels (Paine 1980, Terborgh et al. 1999). Keystone predators can influence the abundance, distribution and behaviour of species, including other subordinate predators, prey and primary producers, and in so doing facilitate nutrient cycling (Paine 1980, Schmitz et al. 2000, Courchamp et al. 2003a).

Much of the research on the role of predators in structuring Australian ecosystems has focussed on the role that introduced dingos may play in regulating the effects of introduced foxes and cats (Johnson et al. 2007, Letnic et al. 2009, Letnic et al. 2011, Fisher et al. 2014, Newsome et al. 2015, Morris and Letnic 2017, Rees et al. 2018). The ecological role of native predators has received less attention (Glen and Dickman 2014a). Nevertheless, there is evidence for predator-driven trophic cascades in temperate mammals (Jones et al. 2014) and reptiles (Sutherland et al. 2011b, Sutherland and Bryant 2014), and arid mammal communities (Dickman 2014b).

Invasive predators can have severe impacts on native wildlife and bring about fundamental changes in ecosystem dynamics (see chapter 2). This chapter considers the reintroduction of native mammalian predators that have gone extinct on Yorke Peninsula.

The loss of top-order predators has been identified as a key factor contributing to continuing species extinctions and global biodiversity loss (Estes et al. 2011, Ritchie et al. 2012). Dramatic re-organisation of ecosystems and losses of biodiversity frequently follow the loss of top predators in marine, aquatic and terrestrial environments (Soule et al. 2003, Ritchie et al. 2012). Consequently, restoring and maintaining the ecological function of predators is a critical component of restoration ecology in general (Hayward and Somers 2009, Ritchie et al. 2012) and of rewilding southern Yorke Peninsula in particular (Sharp 2018).

All ten species of native predators on Yorke Peninsula are locally extinct (McDowell et al. 2012). The red-tailed phascogale, western quoll and Tasmanian devil have been chosen for reintroduction onto Yorke Peninsula (Table 1.2; Figures 7.1) because they were identified as keystone predators that are likely to control mouse (phascogale and quoll) and rabbit (quoll) and possibly cats (devil) and would benefit from establishing additional populations for conservation (Taggart 2014, Sharp 2018; Figure 7.3).

7.1 Red-tailed phascogales

Red-tailed phascogales (*Phascogale calura*) are small, nocturnal, semi-arboreal marsupial predators. They have been recorded in the Innes National Park (181-235 years ago) and Troubridge Hill (797 –

994 years ago) fossil assemblages (Table 1.1; Figure 1.7)(McDowell et al. 2012), but have not been recorded on Yorke Peninsula historically (Aitken 1973, Brandle 2008). The species formerly occurred over much of semiarid and arid Australia (Figure 7.1), and now occupies < 1% of their former range in south-western Western Australia (Kitchener 1981, Short and Hide 2012). Fragmentation of habitat by clearing for agriculture, particularly of woodland habitats rich in suitable nesting hollows, and predation by cats and foxes contributed to this decline (Short and Hide 2012). Although large forest owls (*Ninox* spp.) prey on phascogales (Van Dyck and Gibbons 1980, Soderquist and Ealey 1994), they do not occur on Yorke Peninsula. Red-tailed phascogales have been successfully reintroduced to areas with natural populations of the owls that occur on Yorke Peninsula (Short and Hide 2015).

The life history of red-tailed phascogales has been described by Bradley (1997). Females (32.6 ± 0.9 g) are smaller than males (47.2 ± 2.7 g). They have a short (3 weeks), synchronised annual mating period in winter followed by the death of all males during July. In contrast females may live for up to three years. Young are born in August, so male cohorts do not overlap. Litter size is eight and show a male biased sex ratio (2 males: 1 female). Young are suckled during September-October and become independent from December onwards. Males disperse widely in search of mates during their short (up to 11.5 months) lives, whereas females are relatively sedentary. Males have larger home ranges than females (mean = 13.8 ha vs 6.7 ha)(Short and Hide 2015).

Field studies show that red-tailed phascogales prefer vegetation with a canopy of rock sheoak (*Allocasuarina huegeliana*) associated with large hollow-bearing Eucalyptus wandoo trees (Kitchener 1981, Short et al. 2011). Nest boxes placed in less preferred habitats were quickly occupied, suggesting that suitable hollows for shelter are key feature of suitable habitat, rather than other characteristics of the vegetation (Short et al. 2011, Short and Hide 2015). This opportunistic hunter feeds on invertebrates, birds, mammals and reptiles (Kitchener 1981, Stannard et al. 2010). Arthropods made up most of their diet (occurred in 93% of scats), but birds (52% and small mammals (33%) were also common. Introduced mice made of a significant part of their diet (11%)(Kitchener 1981, Stannard et al. 2010). Red-tailed phascogales are unlikely to be affected by baiting to control foxes and cats because they are active hunters, rather than scavengers (Kitchener 1981, Stannard et al. 2010). They also have a high tolerance for 1080 (approximate $LD_{50} = 17.5 \text{ mg.kg}^{-1}$)(King et al. 1989).

The three reintroductions of red-tailed phascogales documented to date have shown mixed results (Stannard et al. 2010, Short and Hide 2015): two were successful and one was not. Abundant quality nesting sites close to dense mid-storey vegetation for foraging and effective predator control were common features of the successful reintroductions. Long-distance movements of males following release have been reported for both species of phascogale (Soderquist and Serena 1994b, Young 2007, Short and Hide 2015), particularly if males are released without females (Soderquist and Serena 1994b). The home ranges of female red-tailed phascogales show considerable overlap (Young 2007, Short and Hide 2015) in contrast to the exclusive home ranges of female brush-tailed phascogales (Traill and Coates 1993, Soderquist 1995). This will probably allow red-tails to exist at higher densities, an advantage in the remnant native vegetation on Yorke Peninsula. Reintroduced red-tailed phascogales persist and breed during severe droughts (Short and Hide 2015). Short and Hide (2015) provide details of nest boxes that are readily used by phascogales. In the absence of sufficient natural hollows. Provision, maintenance and monitoring of nest boxes will be important both to the persistence of reintroduced populations and to being able to monitor their long-term fate (Short and Hide 2015). The results of the planned reintroduction of red-tailed phascogales to the Australian Wildlife Conservancy's Mount Gibson Sanctuary (Groom et al. 2015) should also be considered when planning the release of red-tailed phascogales to Yorke Peninsula.



Figure 7.1: The three native predators chosen for translocation to southern Yorke Peninsula: red-tailed phascogale (*Phascogale calura*); top), western quoll (*Dasyurus geoffroii*, middle) and Tasmanian devil (*Sarcophilus harrisii*, bottom). Photographs from Taggart (2014) and Wikipedia (<https://en.wikipedia.org>)

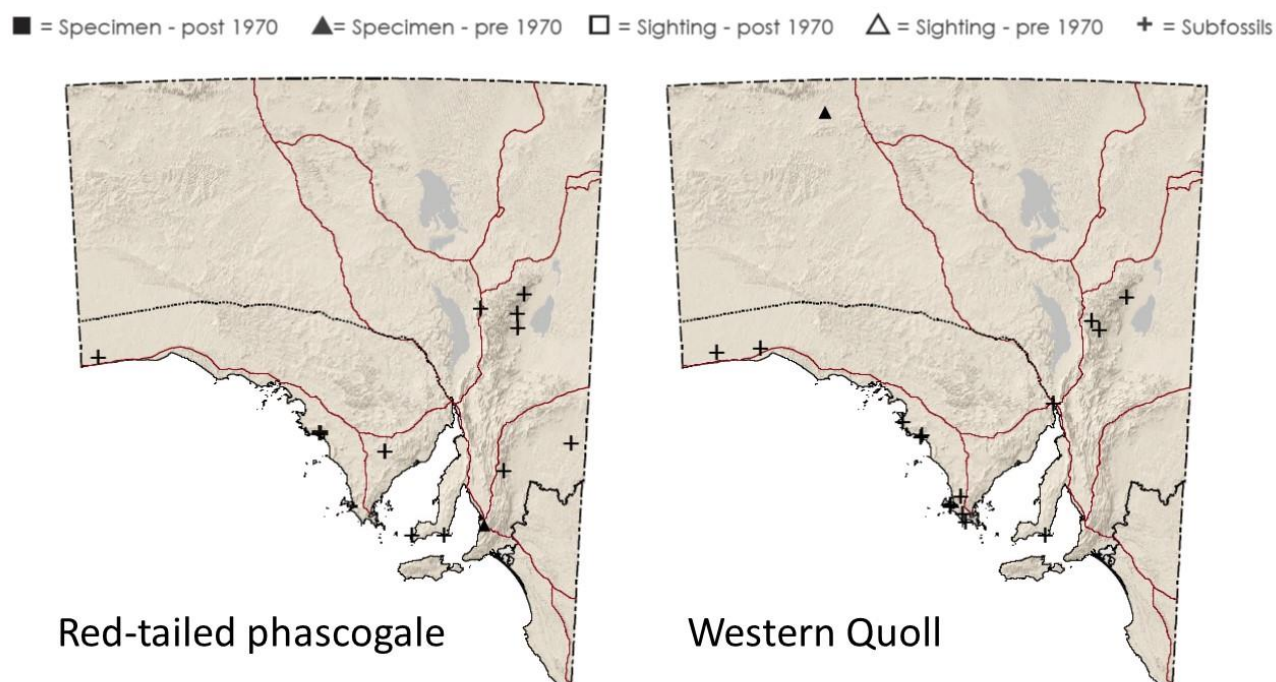


Figure 7.2: Known distribution of red-tailed phascogale and western quoll in South Australia from Owens and Graham (2009)

7.2 Quolls

There are six species of quolls (genus *Dasyurus*) in Australia and New Guinea and two are thought to have occurred on Yorke Peninsula (Woolley et al. 2015). The western quoll (*D. geoffroii*) definitely occurred on Yorke Peninsula, based on two specimens from the Troubridge Hill fossil assemblage (McDowell et al. 2012; Table 1.1, Figures 1.7 and 7.2). Three other specimens from the Troubridge Hill assemblage may be eastern quolls (*D. viverrinus*), though they lack key diagnostic features (Graham Medlin, South Australian Museum, personal communication to GJ, 2018). These specimens were listed as "*Dasyurus* sp. Indet." by McDowell et al. (2012). Historical and subfossil eastern quolls have been recorded nearby from Kangaroo Island (Haouchar et al. 2014), the Mount Lofty Ranges, southern Flinders Ranges and south-eastern South Australia (Owens and Graham 2009). Until the occurrence of eastern quolls is confirmed on Yorke Peninsula it seems reasonable to consider only western quolls for reintroduction there. The remainder of this account reviews the relevant information on western quolls. This species is also widely referred to in the literature as 'Chuditch, the 'Noongar name from south-western Western Australian (Morris et al. 2003).

Western quolls were the largest native predator on Yorke Peninsula at the time of European settlement (Table 1.1). Males weigh 1300–1600 g, but females are smaller (700–1200 g) (Serena and Soderquist 1989b, Rayner et al. 2012, McGregor et al. 2014b). They are swift runners that spend most of their time on the ground, even though they are capable climbers (McGregor et al. 2014b). Quolls are opportunistic, generalist predators of mammals, birds, reptiles and invertebrates, but also consume plant material, and human refuse where they live close to people (Soderquist and Serena

1994a, Gee 1999, Redner 1999, Glen et al. 2010b, Rayner et al. 2012). Individual prey items vary in size from individual termites to rabbits. Remains of mammals that are too large to be predated by quolls have been found in scats (e.g. large macropods and ungulates) and are probably the result of scavenging carrion (Glen et al. 2010b, Rayner et al. 2012). Mammals and invertebrates make up most of the diet of western quolls (32-95% and 75 - 81%, respectively), and birds make up 2-39% of the diet (Soderquist and Serena 1994a, Gee 1999, Redner 1999, Glen et al. 2010b, Rayner et al. 2012). In keeping with their role as generalist predators they have also been recorded consuming southern brown bandicoots (13-29%), brush-tailed bettongs (0-9%) and Phascogales (0 - 0.8%).

Western quolls are iteroparous. They begin breeding at one year of age (Soderquist and Serena 1990), and may continue breeding until they are 5 years old (Cuthbert 2010). Breeding is seasonal and mating usually occurs between April and June (Stead-Richardson et al. 2001). Gestation lasts 16 to 18 days (Serena and Soderquist 1988). The young are deposited in a den, usually a burrow or hollow log, after 64 days (Serena and Soderquist 1989a), are weaned at about 160 days, and eventually disperse after 170 days (November to January) (Soderquist and Serena 2000). Females have litters of up to six young, allowing a potential intrinsic rate of population increase (r_{max}) of 1.25 (Cuthbert 2010). Western quolls are generally regarded as solitary (Morris et al. 2003), but their use of communal latrines (Serena and Soderquist 1989b) suggests this may not be strictly true.

Western quolls once occurred over 70% of the Australian mainland, from the cool, moist forests of southern Western Australia to hot, dry inland areas. They are now restricted to a fragmented distribution in south-western Australia, representing less than 2% of the continent (Morris et al. 2003, Burbidge and Eisenberg 2006). Numerous factors may have played a part in this decline, including land clearance, degradation of habitat by introduced herbivores, persecution by humans, altered fire regimes, epidemic disease and the impacts of foxes and feral cats (Finlayson 1936, Jones 1969, Johnson and Roff 1982, Serena and Soderquist 1991, Abbott 2006, 2008a, Dunlop and Morris 2008, Glen et al. 2009, Peacock and Abbott 2014). Refuge sites (hollow logs, burrows, rock crevices, abandoned white-browed babbler nests, and concrete and metal waste piles) are a crucial component of the habitat for quolls because they supply shelter from adverse conditions and predators (Serena and Soderquist 1989a, Redner 1999, Rayner et al. 2012). Fire history and other sources of disturbance are also important in so far as they impact on the availability of refuges and exposure to predators (Glen et al. 2009, Rayner et al. 2012, McGregor et al. 2014b).

Predation by foxes is a major threat to western quolls (Orell and Morris 1994, Dunlop and Morris 2008, Glen et al. 2009, Department of Environment and Conservation 2012), although they can persist in areas of low fox density (Morris et al. 2003, Rayner et al. 2012). Western quolls have been predated or eaten as carrion by foxes and cats (Mathew 1996, Morris et al. 2003, Glen et al. 2009, Glen et al. 2010a). Early attempts to translocate western quolls to areas where there had been no fox control failed (Serena and Soderquist 1991, Morris et al. 2003), whereas populations increased following fox control (Morris et al. 2003, Glen et al. 2009, Wayne et al. 2017b). This makes clear the importance of foxes in controlling quoll populations, but the role of cats is less clear. Decreases in quoll populations after initial increases following fox control may be due to competitive release of cats (Glen et al. 2009), but this has not been demonstrated. In densely vegetation areas quolls (Serena and Soderquist 1991), foxes and cats (Mahon et al. 1998) often travel along roads and walking trails. This may increase the frequency of encounters between them (Glen et al. 2009), and offers an opportunity to monitor their relative abundance and frequency of encounter using sand pads. In addition to predation, the broad overlap in diets between western quolls, cats, foxes and Rosenberg's goannas suggests that exploitation competition may play a role in regulating quoll populations (Glen et al. 2009).

Western quolls have been successfully reintroduced to several locations in Western Australia (Morris et al. 2003, Dunlop and Morris 2008, Short 2009, Johnson et al. 2011, Morris et al. 2015), and to Ikara-Flinders Ranges National Park and Arid Recovery Reserve in South Australia (Moseby et al. 2016, Brandle et al. 2018). Therefore, procedures for transport, release and monitoring are quite well refined for western quolls. This includes the use of a special bait to allow specific monitoring of western quolls in the presence of other abundant native mammals (Wayne et al. 2008). Western quolls have some resistance to 1080 (LD_{50} approximately 7.5 mg.kg^{-1}), but it is not as high as that of native herbivores (King et al. 1989). This raised initial concerns that fox baits may inhibit population growth of western quolls (Soderquist and Serena 1993). However, no western quolls died following a field trial of toxic 1080 dried meat baits used to control foxes, even though they took some baits (Morris et al. 1995). Western quolls also consume sausage-style baits designed to control cats (ERADICAT and CURIOSITY etc) and the associated toxin-containing pellet (Hetherington et al. 2007, de Tores et al. 2011), but they persist in areas where these baits have been used (e.g. Flinders Ranges National Park; Robert Brandle, personal communication to GJ, 2018). Other species of quolls have also shown no negative response to fox, dog and cat baiting in the field (Belcher 1998, Kortner et al. 2003, Claridge and Mills 2007, Körtner 2007). Although limited, their resistance to 1080 is sufficient to protect western quolls from baits currently used to control foxes and cats.

As predators that include a large proportion of mammals in their diet, western quolls have the potential to regulate populations of overabundant, introduced rodents. Historical accounts suggest that quolls (*Dasyurus* spp.) were, at least partly, responsible for the failure of nearly 300 independent releases of rabbits in Australia between 1788 and 1900 (Peacock and Abbott 2013). These accounts were mostly from eastern Australia, involving eastern quolls and spotted-tailed quolls (*D. maculatus*), but enough of them involve western quolls to suggest that they also have the capacity to regulate rabbit population when they occur in the landscape. Western quolls also eat mice (*Mus musculus*) (Dickman 1992, Soderquist and Serena 1994a, Glen et al. 2010b, Rayner et al. 2012) and mice avoid the odours of quoll faeces, showing that predation pressure on them is sufficiently high to induce a behavioural response (Dickman 1992). The large proportion of mice in diet of quolls during a mouse plague (Rayner et al. 2012) demonstrates that quolls can take advantage of locally abundant prey and suggests quolls may play a role in moderating the effects of mouse plagues.

Most field studies of western quolls have been done in mesic jarrah forests (annual rainfall = 711-1400mm) (Hayward et al. 2005b) and are probably of little direct use in understanding how quolls may adapt to drier conditions. Rayner's et al. (2012) study of western quolls in semi-arid mallee (annual rainfall = 340mm) is probably indicative of how they may adapt following reintroduction on Yorke Peninsula (annual rainfall = 381-447mm; section 1.3.1 climate, above). The population density was $0.039/\text{km}^2$ (Rayner et al. 2012), almost three times lower than in Jarrah Forrest ($0.34\text{-}0.68/\text{km}^2$) (Serena and Soderquist 1989b, McGregor et al. 2014b). Home-ranges in mallee were substantially larger (662-3522 ha for male; 174-202 ha for females) (Rayner et al. 2012) than in Jarrah forest (410-791 Ha for males; 90-314 Ha for females) (Serena and Soderquist 1989b, Mathew 1996, Gee 1999, Redner 1999, McGregor et al. 2014b). As yet unpublished results from recent reintroductions of western quolls to the Arid Recovery Reserve and Ikara-Flinders Ranges National Park in South Australia (Moseby et al. 2016) will provide useful information that should be incorporated into plans to reintroduce this species to Yorke Peninsula.

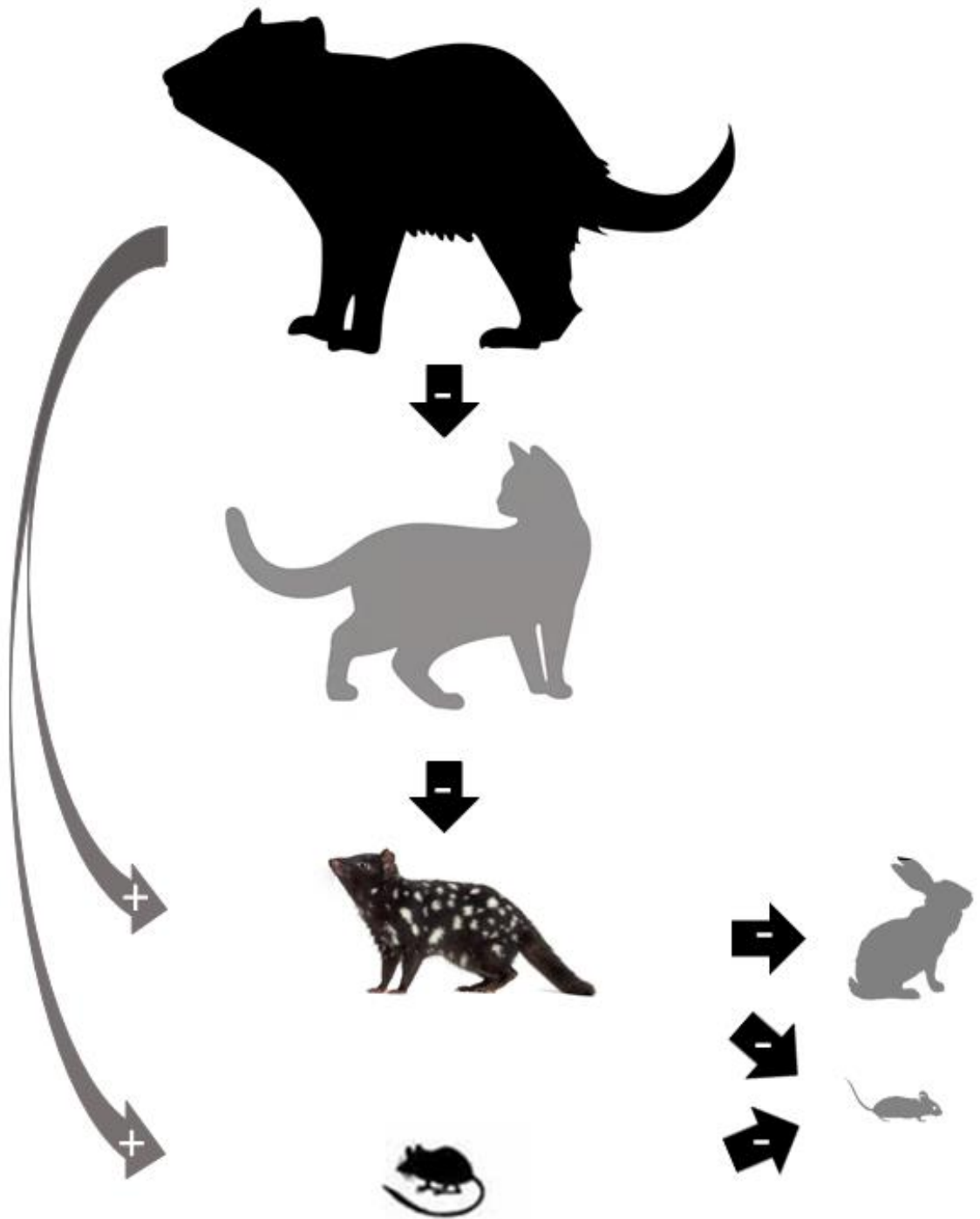


Figure 7.2: Hypothesised impact of reintroducing Tasmanian devils, western quolls and red-tailed phascogales (black silhouettes) on introduced (grey silhouettes) cats, rabbits and mice (Sharp 2018). Black arrows show direct negative effects, Grey arrows show indirect positive effects. Tasmanian devils are predicted to suppress the abundance and activity of feral cats. This will lead to an indirect positive effect on quolls and phascogales by reducing the impact that cats have on them. Quolls are expected to prey upon rabbits and mice, phascogales are expected to prey upon mice.

7.3 Tasmanian devils

The Tasmanian devil (*Sarcophilus harrisii*) is the world's largest extant carnivorous marsupial, with males weighing ~ 9kg and females ~6kg (Rose et al. 2017). Although devils are now restricted to Tasmania, they were widespread in mainland Australia until 3,200 years ago (White et al. 2018). The cause(s) of their extinction on the mainland are unclear, but may have included negative effects of human population growth, the introduction of dingoes to Australia, and increased variation in El Niño-Southern Oscillation cycles, (Gollan 1984, Mulvaney and Kamminga 1999, Johnson and Wroe 2003, Brown 2006, Prowse et al. 2014, Brüniche–Olsen et al. 2018), or some combination of these. Tasmanian devils have not been recorded from the fossil deposits on Yorke Peninsula (Pledge 1992, McDowell et al. 2012) but did occur on Kangaroo Island between 145,000 and 17,000 years ago, when the island was part of mainland Australia (Wells et al. 2006, McDowell et al. 2015b, Camens et al. 2018, Peacock et al. 2018). This suggests they also occurred on Yorke Peninsula.

Despite having been extirpated from mainland Australia, devils were regarded as common in Tasmania until the outbreak of an infectious cancer that only affects devils (Driessen and Hocking 1992). Devil Facial Tumour Disease (DFTD) is remarkable in that two different infectious cancers have arisen independently in wild devil populations over a relatively short period of time. Tasmanian devils with tumours and lesions on the face caused by the highly contagious DFT1 were first recorded in 1996 (Hawkins et al. 2006); the other cancer, DFT2, is thought to be less pathogenic and was discovered in 2014 (Pye et al. 2016). Both DFTDs are transmitted when devils bite each other during territorial disputes or courtship, making devils most susceptible as they reach sexual maturity (Hamede et al. 2009). Once the tumours are evident, the life expectancy of a devil is less than six months (Hawkins et al. 2006). DFTD occurs over 80% of Tasmania and has caused infected devil populations to decline by more than 75% overall, and more than 95% where the disease has been present longest (Jones et al. 2014, Lazenby et al. 2018). Devils are now endangered (Hawkins et al. 2008), and continue to decline (Jones et al. 2014, Grueber et al. 2018, Lazenby et al. 2018).

Tasmanian devils are iteroparous. Females begin breeding at two years of age, and may continue breeding until they are 7.5 years old (Cuthbert 2010). Females give birth to 3 young per year, allowing a potential intrinsic rate of population increase (r_{\max}) of 0.57 (Cuthbert 2010). Interestingly, devils are reaching sexual maturity earlier and having larger litters with female-biased sex-ratios due to strong selective pressure brought about by DFTDs (Jones et al. 2008, Lachish et al. 2009). Such changes in life-history should be considered in any population modelling associated with the planned translocation of Tasmanian devils to Yorke Peninsula in 2034 (Table 1.2). Apart from DFTD, the availability of food in the form of carcasses or suitable-sized, slow-moving prey (i.e. mostly mammal carcasses, but also young of ground and burrow-nesting birds) appears to limit population growth in devils (Jones et al. 2014, Rose et al. 2017).

Devils generally scavenge from vertebrate corpses, but can also be opportunistic predators (Rose et al. 2017). They have a remarkably catholic diet, including birds, rabbits, possums, echidnas, wombats, macropods, and sheep (Pemberton 1990, Jones 1997, Jones and Barmuta 2000, Jones and Barmuta 2002, Pemberton et al. 2008). This reflects their scavenging habit, rather than what they actively hunt. In keeping with this, devils consume hair and fragmented bones, as well as the flesh of 'prey', leaving only parts of large skulls and the often still-filled colon (Pemberton 1990, Pemberton et al. 2008). Following a meal devils leave scraps strewn over areas up to 500 m². Devils are gorge feeders, consuming one large meal every 4–8 days (Pemberton 1990). They can consume up to 40% of their body mass in a meal (Pemberton 1990). The scavenging mode of procuring most food,

together with their limited ability to run and jump suggests that devils do not generally function as true predators (Fancourt and Mooney 2016).

The impact of devils on sheep is a point of social controversy in Tasmania. The frequent presence of wool in devil faeces indicates that sheep are eaten by devils. Devils certainly scavenge moribund lambs and already dead lambs and sheep (Rose et al. 2017), but the extent to which devils actively prey on healthy sheep and lambs is unclear from the available literature.

In response to the threat of extinction posed to devils by DFTDs (McCallum et al. 2009) a captive population has been established by zoos on mainland Australia (Keeley et al. 2012, Farquharson et al. 2017, Harley et al. 2018) and translocation of DFTD-free devils to mainland Australia has been proposed to establish wild insurance populations (Hunter et al. 2015). It has also been suggested that translocation of devils to mainland Australia may restore top-down control of herbivores and smaller, introduced predators (Ritchie et al. 2012, Hunter et al. 2015). The potential for devils to suppress cat populations is an integral part of the rationale for translocating devils onto southern Yorke Peninsula (Sharp 2018).

The hypothesis that translocated devils may suppress the population of cats on mainland Australia (Ritchie et al. 2012, Hunter et al. 2015) is a corollary of the suggestion that the loss of devils from Tasmanian ecosystems results in an increase in feral cat abundance there via competitive release (Macdonald and Thom 2001, Jones et al. 2007). The primary evidence for this is a negative relationship between the abundance of devils and cats in north-eastern Tasmania (Hollings et al. 2014). However, not all sites in Hollings *et al.*'s (2014) study showed this negative relationship. Indeed, other studies have found no relationship between the number of devils and cats (Troy 2014, Fancourt *et al.* 2015, Lazenby *et al.* 2018), and some have found a positive relationship between them (Saunders 2012, Hollings et al. 2014 in central Tasmania). These contradictory results may reflect independent demographic responses by devils and cats to food availability. Alternatively, behavioural shifts in cats to diurnal activity in the presence of the nocturnal devils (Lazenby and Dickman 2013, Fancourt et al. 2015) may have reduced the likelihood of detecting cats in spot-lighting counts, the method used by most studies to document changes in cat and devil abundance. Fancourt *et al.* (2015) have speculated that differences in feeding ecology of cats (active hunting predator of small-medium-sized prey) and devils (scavenger/carnivore eating large 'prey') make competition between them unlikely. For the moment, the critical information required to objectively assess whether devils do or do not suppress cat population density is not yet available.

The ability of Tasmanian devils to regulate the abundance of feral cats is currently an area of active research. Should the impact of devils on sheep be sufficiently low and the impact of devils on feral cats be confirmed, then translocating them to Yorke Peninsula may provide benefits to agriculture by reducing the incidence of cat-borne diseases. It is appropriate that the translocation of Tasmanian devils to Yorke Peninsula is not planned until the later stages of the rewilding program (Table 1.2; year 20), providing sufficient time to allow these critical questions to be addressed and a considered discussion to be had regarding the efficacy of the proposal. By then we should have a clearer understanding of the costs and benefits of this component of the rewilding program.

7.4 Knowledge gaps

The importance of maintaining predator diversity (Finke and Denno 2004, Ritchie et al. 2012) and effective population densities of predators (Soule et al. 2003) to mitigate ecological degradation has

not been demonstrated in Australia (Hunter et al. 2015). The planned reintroductions of red-tailed phascogales, western quolls and Tasmanian devils to Yorke Peninsula (Sharp 2018; Table 1.2) present an opportunity to test this hypothesis.

It is important to keep in mind that all predators present in a system have the potential to interact with one another (Glen et al. 2009). For this reason, the effects of removing or adding predators to a landscape should adopt a whole-of-community approach. In the case of southern Yorke Peninsula this would ideally involve simultaneous monitoring of the three translocated species, resident native predators (Rosenberg's goannas, diurnal and nocturnal raptors), as well as cats and foxes. The responses of prey, vegetation communities, and agricultural and tourism productivity should also be measured.

The responses of reintroduced populations can be unique to each specific location. For this reason, movements, diet and demography of red-tailed phascogales, western quolls and Tasmanian devils translocated to southern Yorke Peninsula should be intensively monitored. This is particularly important the species in which there have been few (phascogales) or no (devils to the mainland) previous attempts to re-introduce them. Key variables associated with each reintroduced predator's hypothesised ecological function should be monitored (e.g. the density of mice, rabbits, Rosenberg's goannas and other translocated native carnivores and potential prey species) (Glen et al. 2009).

It is commonplace to use indices of relative density (e.g. capture rates) to assess responses of populations but this kind of information confounds demographic and behavioural changes. For example, it is not known whether changes in cat sightings during spotlighting counts reflect a change in population density or a tendency for cats to be more active during the day (a behavioural change) in response to Tasmanian devil abundance (Fancourt et al. 2015). Similarly, the mechanisms leading to the putative demographic effect of fox control on cats needs to be determined: Is it due to predation, competitive interactions, or some other mechanism unrelated to fox control (Morris et al. 2003, Morris et al. 2015)? Future research should attempt to measure demographic responses directly using traditional and genetic capture-mark-recapture approaches (Taylor et al. 1997, White and Burnham 1999, McLennan et al. 2018). By monitoring the resource use, survival and causes of death of putative competitors and prey, monitoring should seek to clarify not only *if*, but *how* species influence the abundance and distribution of others.

Birds can be a significant part of the diet of red-tailed phascogales and western quolls. Both may predate key endangered birds on Yorke Peninsula, most notably hooded plovers (*Thinornis rubricollis*), malleefowl and mallee whipbirds. Historically phascogales and quolls co-existed naturally with all three species. Red-tailed phascogales prey upon small passerines (Kitchener 1981, Stannard et al. 2010), but are unlikely to affect the larger malleefowl. Predation by quolls is not regarded as a threat to malleefowl (Benshemesh 2007, Johnson et al. 2011) or hooded plovers (Maguire et al. 2014, Department of the Environment 2018). Although we could find no specific record of phascogales or quolls preying upon whipbirds or their eggs, this remains a possibility (e.g. Fraser and Whitehead 2005, Burbidge et al. 2016). Whipbirds are a particularly important species on southern Yorke Peninsula. Following their extinction in Victoria, mallee whipbirds are now endemic to South Australia, and the population on Yorke Peninsula shows considerable genetic divergence from other extant populations of mallee whipbirds (Burbidge et al. 2016). Tasmanian devils prey on ground nesting birds opportunistically (Jones 1997, Jones and Barmuta 2000, Jones and Barmuta 2002). Devils do not naturally co-occur with megapodes (Menkhorst and Knight 2011, Menkhorst et al. 2017), so the possibility that devils may threaten malleefowl of Yorke Peninsula should be investigated. This information will also be important to allow better-informed planning of other potential translocation of devils to mainland Australia where endangered megapodes occur. Populations of malleefowl,

hooded plovers and mallee whipbirds on Yorke Peninsula should be monitored to allow adaptive management so their populations are not compromised by any of the planned predator reintroductions to Yorke Peninsula.

With respect to red-tailed phascogales, it will be important to determine whether there are enough tree hollows on Yorke Peninsula to sustain a viable population of red-tailed phascogales. Established procedures should be used to assess this (Short and Hide 2015). Should it be necessary, artificial nest boxes could be used to provide supplementary hollows. Nest boxes also provide a convenient method to monitor the phascogales (Short and Hide 2012, 2015).

With respect to devils, distribution modelling should be used to ensure that Yorke Peninsula provides a suitable climate for this species. A currently available species distribution model suggests that Yorke Peninsula is not suitable for devils (Hunter et al. 2015). However, that model is based solely on the devil's current distribution in Tasmania and may underestimate their potential distribution on the mainland. A more comprehensive distribution model should include additional information from survival of escaped captive devils on the mainland (White and Austin 2017), the fossil record and associated paleo-environmental conditions (White et al. 2018).

If devils are translocated to southern Yorke Peninsula, their movements, diet and demography should be intensively monitored. This will provide important information on how the species does in an environment which it has not inhabited for 3,200 (see 7.1.1.3). Key variables that indicate the devils hypothesised ecological function should also be monitored (e.g. the density of rabbits, cats, and smaller translocated native predators). This information will be important to allow adaptive management of the species on Yorke Peninsula, and better-informed planning of other potential translocation of devils to mainland Australia.

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