Threatened Species Recovery Hub

National **Environmental Science** Programme

A guide to surveying red foxes and feral cats in Australia

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Front cover image: Red fox Vulpes vulpes *in the Otway Ranges, Victoria. Image: Mark Le Pla*

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1. Introduction

1.1 Overview

Predation by European red foxes *Vulpes vulpes* (hereafter, 'foxes') and feral cats *Felis catus* (hereafter 'cats') poses a major threat to Australia's wildlife (Saunders et al. 2010; Woinarski et al. 2015). Foxes also prey on livestock, and both predators host pathogens that can cause disease in humans and livestock (Saunders et al. 2010; Legge et al. 2020). Consequently, Australia invests in fox and cat control across public and private land for environmental and agricultural purposes, including an increasing number of long-term, landscape-scale predator management programs (Allsop et al. 2017; Garrard et al. 2020).

A good understanding of the distribution, density and behaviour of these introduced predators is fundamental to measuring their impacts, deciding when management is required, and designing and refining effective management programs for local conditions. A wide array of field survey methods, data metrics and analytic approaches can be used to obtain this information.

This guide aims to provide a broad introduction to the different contemporary approaches for monitoring foxes and feral cats in Australia, highlight key trade-offs, and provide links to more specific resources, including examples of Australian studies that have applied these methods. The examples focus on monitoring introduced predators for environmental outcomes, but the advice can equally be used to monitor these species in an agricultural context.

The guide is divided as follows:

- Section 2: Introduction to the ecology of feral cats and foxes in Australia.
- Section 3: Overview of commonly used field methods for studying introduced predators.
- Sections 4–7: Guides to designing surveys for, and analysing data on, introduced predators.
- Section 8: Case studies of existing introduced predator monitoring programs to provide insights into practical approaches and key challenges.

Table 1 provides a summary of which field survey methods (described in Section 3) can be used to answer different types of questions about introduced predator populations (Sections 4–7).

Fox being fitted with tracking collar at Scotia Wildlife Sanctuary. Image: Australian Wildlife Conservancy

Table 1. Different field survey methods can be used to address different questions about introduced predator populations and behaviours. Monitoring programs often use more than one method at the same time. Field survey methods are described in Section 3, approaches for designing surveys and analysing data are addressed in Sections 4–7.

How could I measure it?		What do I want to measure?						
Field survey method	Field effort required	Bait take	Population index / occupancy	Population density	Habitat selection	Diet	Survival	Movement
Bait take	Set up, repeated checks	ground baiting (foxes)	X					
Track surveys	Set up, repeated surveys	X	X		X			
Spotlighting	Repeated or one-off surveys		X	X	X[1]			
Camera surveys	Set up, retrieve	X	X	X	X		+ (partially) marked populations	+ (partially) marked populations
Scat surveys	Repeated or one-off surveys ± dogs		X	+ DNA genotyping		X	+ DNA genotyping	+ DNA genotyping
Hair surveys (remote collection)	Set up, repeatedly survey		X[1]	+ DNA genotyping	X[1]		+ DNA genotyping	+ DNA genotyping
Biological samples (invasive collection)	Trap, shoot or collect carcasses	+ biomarkers in baits	X			X		+ DNA genotyping
Tracking collars	Trap, collar, release, retrieve data	X		+ camera surveys	X	+ collar-mounted video camera	X	X

[1]Potentially possible but very rarely done.

1.2 Terminology

Throughout this guide, we have used the following terminology:

- 'landscape' – a broad area inhabited by a population of predators.
- 'treatment landscape' – a landscape where a management intervention is (or will be) conducted.
- 'non-treatment landscape' – a landscape where management is not being conducted, typically included in a monitoring program as an experimental control.
- 'site' – the point or transect where a survey is conducted. Surveys are often conducted at several sites within a landscape.
- 'survey event' – a single survey occasion, e.g. one night of spotlighting, one survey for tracks or 24 hours of camera deployment.
- 'session' – a period during which repeat surveys are being conducted at each site (usually days – weeks).
- 'n' – a certain number, e.g. "n days" may be 1 day, 3 days or 10 days.

1.3 Regulations

Introduced predator management and research in Australia is strictly regulated: activities must be conducted as humanely as possible and in accordance with state and territory laws and guidelines, including animal ethics approval when relevant.

Model codes of practice and standard operating procedures for the humane capture, handling and destruction of foxes and feral cats have been developed by the NSW Department of Primary Industries, and are available from https://pestsmart.org.au/ (CISS 2021).

1.4 Occupational health and safety

Foxes and cats in Australia carry pathogens that can be transmitted to people, and can make you seriously unwell or kill you, including the parasites *Echinococcus granulosus* (which causes hydatid disease) and *Toxoplasma gondii* (which causes toxoplasmosis). People who handle live or dead foxes and cats, and their scats, are at risk of catching these pathogens. Wearing gloves and washing hands thoroughly after handling animals or their scats is essential. A face mask, safety glasses and/or lab coat may also be appropriate.

If collecting scats, wear gloves, and use a new ziplock bag turned inside out or fresh sticks to pick up each scat to avoid handling scats directly and avoid cross-contamination of samples. Remain upwind of the scat and keep it as far from you as possible. Further information on how to safely collect predator scats can be found at https://pestsmart.org.au/pest-animals/monitor-techniques/(CISS 2021).

Care should also be taking when handling lures and baits that may have been mouthed by predators.

Pregnant women and women planning pregnancy in the next three months should avoid exposure to cat scats due to the risk of catching *T. gondii* and transmitting it to their foetus. Foetal toxoplasmosis can cause miscarriage, stillbirth, blindness, deafness or brain damage in unborn babies. See https://www.pregnancybirthbaby.org.au/toxoplasmosis for more information.

Blood test screening for hydatid and *T. gondii* infections is available. Consult your doctor for further information.

All the normal hazards associated with fieldwork and handling wildlife also apply, including driving, working in remote places, working near roads, fatigue, venomous animals, animal bites etc. Risk assessments and risk management plans should be developed before commencing any monitoring program.

2. Ecology

2.1 Feral cats in australia

Domestic cats are thought to have been first introduced to Australia in 1788. They spread rapidly and are now found as feral populations across the entire mainland and many of the larger offshore islands.

Cat social organisation and behaviour is highly variable. In productive areas with rich food resources, such as rubbish tips, farms and seabird colonies, cats live in dense matrilineal societies, and dozens of cats may share a small area. Across Australia's more typical natural environments, however, cats are largely solitary. In these environments, densities vary from 0.01 to 7 cats per km^2, depending on food availability.

Female cat home range size is similarly variable, scaling from less than 1 km^2 to more than 6 km^2 (reviewed in Hamer et al. in press). Female cats have small, partially-overlapping home ranges when annual productivity is high, and large home ranges when productivity is low, provided that seasonal variation is also low; where seasonality is high, this relationship is less clear (Bengsen et al. 2016). The home range size of male cats varies with that of females, but is larger on average because dominant males typically overlap the ranges of two to three females.

Cats can travel far beyond their home range, particularly in response to rapid changes in food availability (e.g., because of fire or rabbit control); some do not have a fixed home range, particularly in arid regions. Both sexes have ranges that may partially overlap ranges of other cats of the same sex.

Female cats attain sexual maturity from as early as four months, but usually at 8-12 months, typically once they are at least 2500 g in weight. Females have one to three litters of ~4 kittens each a year. The winter non-breeding period is longer in cool climates than warm ones, and so the breeding season varies from four to nine months accordingly. Dispersal behaviour is not well documented, but male cats are likely to disperse more often and further than females.

Feral cat survival rates are poorly studied, especially in natural habitats. Kitten mortality rates are likely to be high, and most cats probably live no more than three to five years.

Cats eat approximately 200 g of prey a day. They use 'sit and wait' and stalking strategies to catch a wide variety of vertebrates and insects, and will also occasionally scavenge on dead animals. Individual cats may specialise on particular prey groups, such as birds or small mammals. For reviews of cat diet in Australia, see Doherty et al. 2015; Woinarski et al. 2017; Woinarski et al. 2018; Murphy et al. 2019; Woinarski et al. 2020; Woolley et al. 2020.

For comprehensive reviews of the ecology, distribution and management of cats in Australia, please refer to:

https://pestsmart.org.au/toolkit-resource/feral-cats-in-australia/ (CISS 2021).

Woinarski J.C.Z, Legge SM, Dickman CR (2019) *Cats in Australia: Companion and Killer*. CSIRO Publishing, Clayton South, Australia.

Legge S, et. al (2017) Enumerating a continental-scale threat: How many feral cats are in Australia? *Biological Conservation* 206: 293-303.

Doherty TS et al. (2016) Impacts and management of feral cats *Felis catus* in Australia. *Mammal Review* 47: 83-97.

Denny EA, Dickman CR (2010) *Review of Cat Ecology and Management Strategies in Australia*. Invasive Animals Cooperative Research Centre, Canberra.

2.2 European red foxes in Australia

European red foxes *Vulpes vulpes* became established in southern Victoria in the 1870s, and then spread rapidly across temperate and subtropical mainland Australia. Foxes live in family groups comprising a dominant pair and their juvenile cubs. If resource availability is high, subordinate adult offspring may also remain with the family group.

Fox density in Australia varies from less than 0.1 fox per km^2 in semi-arid regions to more than four foxes per km^2 in productive environments such as farmlands and wetlands. Fox home range size follows the opposite pattern, varying from less than 1 km^2 in urban and productive agricultural environments, to upwards of 20 km^2 in the arid zone. Foxes within the family group share a home range, but usually hunt alone. Fox home ranges are dynamic: they can drift through time, involve excursions to concentrated food resources, and change rapidly if resource availability alters or a neighbouring home range becomes vacant (it only takes 8 days – 1 month for a vacant home range to be taken over).

Foxes have a strongly seasonal lifecycle: mating occurs in mid-winter, cubs are born in spring, and offspring disperse in autumn and early winter, before breeding that season. Fox detection rates in surveys may vary with these seasonal behaviours. Survival and fecundity rates vary with age and local conditions, and litter size may increase after lethal control. Sex ratios at birth usually approximate parity. The proportion of young that disperse also varies, but males are more likely to disperse than females, and disperse further on average.

Foxes feed on a wide variety of vertebrates, arthropods, carrion and fruits, typically consuming about 380 g per day. For reviews of fox diet in Australia, see Fleming et al. 2021; Stobo-Wilson et al. 2021.

For comprehensive reviews of the ecology, distribution and management of foxes in Australia, please refer to:

- https://pestsmart.org.au/toolkit-resource/european-red-fox/ (CISS 2021).
- Hunter DO. et al. (2018). Not all predators are equal: a continent-scale analysis of the effects of predator control on Australian mammals. *Mammal Review* 48: 108-122.
- Devenish-Nelson et al. (2013). Demography of a carnivore, the red fox, *Vulpes vulpes*: what have we learnt from 70 years of published studies? *Oikos* 122: 705-716
- Saunders GR, Gentle MN, Dickman CR (2010). The impacts and management of foxes *Vulpes vulpes* in Australia. *Mammal Review* 40: 181-211.

Camera-trap image of a feral cat in recently burnt habitat, Otway Ranges, Victoria. Image: Craig Mildwaters

3. Guide to field survey methods

Many different field methods can be used to study foxes and feral cats. They vary in the technology required, the need for ecological expertise and training, their impact on the predators, and what they tell you about the predator population (Table 1).

Methods are described here in rough order of simplicity, and their key benefits and limitations are highlighted. The costs of each method also vary substantially, depending on the need to purchase equipment, the person-hours required to collect and process data in the field, office and/or laboratory, and sample processing costs, but are not a primary focus of this guide.

It is always helpful to talk with someone who has used similar methods in a similar landscape before beginning a new field program.

3.1 Bait take

Bait take rates tell you how many baits are being removed and are sometimes used as an index of predator activity (particularly for foxes). However, baits can be taken by species besides introduced predators, so it is informative to set a sand pad (i.e. a sandy area, smoothed so that animal tracks are easily visible) or survey camera at each bait station so that the species that visit the bait station or remove baits can be identified (e.g. Dexter & Meek 1998; Fancourt et al. 2021) .

Advantages. Bait presence/absence is simple to record as a routine part of an approved poison baiting program, if baits are being deployed at stations by hand (rather than aerially). In combination with sand pads or survey cameras, bait take can provide important information on whether and how quickly the target species is encountering and removing baits, and whether non-target species are removing baits. Bait take rates can show large declines after poison baits are deployed (e.g. 97% decline; Dexter & Meek 1998), and substantial differences between treatment and non-treatment landscapes (e.g. Robley et al. 2014).

Limitations. If the aim is to measure changes in the predator population in response to baiting, bait take rate is likely to overestimate the effectiveness of baiting, because:

- One predator may take (and possibly cache) numerous baits, particularly if the baits are spaced closely together.
- Other non-target species (currawongs, rats etc.) may also remove baits.
- Individual predators that avoid baits (or the habitats where baits are) will not be detected and will survive.
- Individual predators that receive a sub-lethal dose of bait may avoid baits in future, but not be detected again.

It is therefore often a good idea to also include an additional survey method that is independent of management. Nonetheless, trends in bait take data were roughly consistent with trends in fox track count data (Lindenmayer et al. 2018), and bait-take was moderately to strongly correlated with fox spotlight survey data (Marks et al. 2009).

Methods. Bait presence/absence is often routinely recorded as part of approved poison baiting programs. Surveys of sand pads or cameras set at the bait stations can be used to identify which species have visited the bait station and/or interacted with the bait.

Take of non-toxic baits can also be used as an index of predator activity. Variations on this method include:

- Deploying non-toxic baits and determining the proportion removed after n days (e.g. Thompson & Fleming 1994; Banks et al. 1998; Marks et al. 2009).
- Deploying non-toxic baits, determining the proportion removed after n days and waiting until this rate has stabilised (Robley et al. 2014). Target and non-target species are likely to learn where non-toxic baits are, increasing take rates over time.
- Comparing take rate of non-toxic baits over time, followed by take rate of toxic baits (Dexter & Meek 1998).

Data to record: Observer's name, GPS location of the bait stations, date the bait was deployed, type of bait, date bait checked, the presence (± condition) of bait, evidence of animals visiting bait station and/or interacting with the bait (from sandpads or cameras if available).

3.2 Track surveys

Foxes and cats have distinctive footprints, which can be used to survey predator occupancy (presence/absence) or as an index of predator activity. See Case Study 8.1 for an example.

Advantages. Track surveys are non-invasive and do not affect predator behaviour, making them ideal for recording visits to bait stations and traps, and monitoring populations that may be wary of humans or survey devices. If track surveys are conducted on suitable soils by experienced observers in good tracking conditions (i.e. slanting light, no wind or rain), they are unlikely to miss any animal detections. Track surveys do not rely on complex technology and are easily repeatable, making them suitable for monitoring predator populations over long periods. Track surveys also record the presence of other terrestrial fauna comparatively uniformly, making them suitable for investigating relationships between predators and prey. Finally, track surveys allow researchers to directly observe species' signs while collecting data, and this can help develop an intuitive feel for the study system. Allen and Engeman (2015) concluded that track surveys compared favourably with other predator survey indices.

Limitations. Track surveys require considerable field time. Predator detectability in track surveys can be impacted by the soil type and moisture, weather (particularly rain, wind, frost), interference by vehicles or heavy animal traffic, and observer expertise. Training is required to learn to distinguish the tracks of different predators. Fox and cat prints can sometimes be hard to distinguish from each other and other species (particularly young dingoes) in soft substrates. Track surveys measure predator activity, not predator density or abundance; the relationship between activity and density or abundance can vary through time and space, so changes in track data must be interpreted cautiously. Track data can be highly variable, meaning that a large sampling effort may be needed to identify signals. Data may also become 'saturated' at high densities, leading to underestimation of predator numbers (e.g. Thomson et al. 2000).

Methods. Survey transects or plots can be smoothed on naturally sandy soils (e.g. by dragging wire mesh behind a vehicle, a couple of heavy door mats behind a person, or simply using a large broom to sweep and smooth an area) or can be created using sand. Good hygiene practices should be followed to prevent the introduction of soil-borne pathogens. The transect or plot is revisited a day or more later, and walked or driven slowly to check for footprints, usually shortly after sunrise to maximise visibility.

If the aim is to provide an index of population size or occupancy, track survey sites should be located independently of management sites such as bait stations that may affect predator behaviour. Ideally, they should also be randomly located across the landscape (e.g. Southgate et al. 2007). However, they are often conducted along or beside roads for practical reasons and to increase detection rates.

Sites can be checked for tracks on several different days within a session to estimate detectability. A measure of 'imprintability' can also be taken each day – see Fleming et al. (1996) for method details.

Data to record: Observer's name, GPS location of site, date, time, tracking conditions (weather, light), species (presence/absence or count, depending on survey metric – see Table 2), imprintability.

3.3 Spotlighting

Reflective eyeshine means that foxes and cats can be readily seen at night using spotlights, particularly in flat, open environments. Spotlighting can be used to survey predator occupancy (presence/absence), provide an activity index, or estimate predator density. See Case Study 8.3 for an example.

Advantages. Spotlighting surveys are easily repeatable, only require simple technology, and can be used to obtain density estimates for unmarked populations. They can provide data on predators and prey at the same time. Some spotlighting surveys have been conducted in a similar manner for more than 40 years, enabling long-term population changes to be identified (e.g. Hollings et al. 2014). Spotlighting also allows researchers to directly observe species while collecting data, and so helps them develop an intuitive feel for the study system.

Limitations. Spotlighting is most practical in open, flat environments, and requires considerable field time (with 2-3 people per survey). The detectability of predators in spotlighting surveys is strongly affected by the terrain, vegetation type, weather, time of night and animal behaviour, as well as observer expertise. Predators that have been exposed to shooting are likely to have learnt to avoid vehicles and bright lights and so be difficult to detect. Approximately 60-80 animals need to be sighted to estimate detectability with a distance-sampling approach (across the surveys, not per transect; see Section 5.4.2), making it difficult to derive density estimates for sparse populations.

Methods. Spotlighting survey points or transects are usually established along vehicle tracks, or occasionally across open country. Transects typically provide more precise estimates than point surveys The transect is walked or driven slowly at a steady pace (< 10-20 km/h), and one or preferably two observers scan the road and surrounding area. Each transect should be surveyed on several nights within a survey session.

A guide to spotlighting can be found at https://pestsmart.org.au/pest-animals/monitor-techniques/ (CISS 2021)

The time of night that surveys are conducted should target the predators' peak activity period. However, this may vary with seasons, removal of more dominant predator species etc. One option is to randomly the vary time that each transect is surveyed, within the known peak active period (e.g. between 2100h and 0200h; Newsome et al. 2014).

Identifying species during spotlighting can be challenging. Cats, foxes and dingoes have very bright eyeshine, brighter than all native species. Yet it requires experience to tell these three species apart. In general, cat eyeshine has a green-blue tinge while fox eyeshine is more yellow, but this can depend on the type of spotlight. Height and behaviour are better indicators: cats are lower and have more stop-start movements. There is no substitute for experience: after seeing ten or so individuals of both species most people will be confident in telling them apart.

Infra-red thermal imaging may be useful to augment spotlight surveys for predators. Thermal cameras are most effective when there is a large difference between the animal's body temperature and background temperatures (i.e. when the background temperature is less than 25˚C), and at intermediate distances. McGregor et al. (2021) found that in an open desert landscape, adding a thermal camera did not improve detection rates of feral cats, as they were already highly visible due to their reflective eyeshine. Many professional hunters who use thermal cameras typically use a spotlight to first detect a cat, then use the thermal camera as a scope.

If the plan is to estimate density using distance sampling, it is essential to record either the perpendicular distance of the predator from the transect, or the sighting distance and sighting angle so that the perpendicular distance can be calculated later. Distance can be estimated by eye but a hand-held laser range finder is preferable.

Data to record: Observers' names, survey transect name and GPS location, length of the transect, time survey commenced, time each predator seen, location when seen, predator species, predator distance (and angle if relevant), weather and other factors that may influence detectability (e.g. habitat type).

3.4 Camera surveys

"Trail cameras" are now a widely used tool for surveying foxes, cats and other wildlife. The cameras detect movement of an object and/or whether its temperature contrasts with the background ambient temperature. Cameras can be used to survey predator occupancy, provide predator activity indices, or estimate predator density or abundance (particularly if some individuals can be uniquely and repeatedly identified). See Case Studies 8.1 and 8.2 for examples.

Advantages. Relatively little training is required to deploy survey cameras effectively, and species identification is easier than many other methods. Cameras can provide data on predator and prey species at the same time, and can be left in the field for long periods with little maintenance, allowing populations to be monitored over months or years. Data can also be reviewed by independent observers (including citizen scientists) or artificial intelligence tools to support curation of the data.

Limitations. Each camera can only survey a small area relative to a predator's home range, and predators can detect and sometimes avoid cameras (Meek et al. 2016). High background temperatures in hot climates can also interfere with detections by reducing the contrast between the ambient temperature and the predator's body temperature. Hence cameras may have a substantial false negative rate (i.e. fail to detect predators when they are present). For example, studies that have marked individual foxes or cats with collars have found that some, or even most, individuals are never detected on camera, despite moving nearby (e.g. Comer et al. 2018; Carter et al. 2019). Cameras often collect high volumes of data, and so it may take a long time to process images and identify species after field surveys are complete; however, the development of artificial intelligence methods for identifying species is likely to reduce this.

Methods. Useful guides on choosing cameras, setting them up survey introduced predators effectively in Australian conditions and processing images include https://pestsmart.org.au/pest-animals/monitor-techniques/ (CISS 2021), Meek et al. (2012) and Gillespie et al. (2015).

Detectability is an important consideration when deploying cameras, regardless of the study purpose. Approaches for improving the detectability of predators at each site include:

1. Setting the camera so that the detection zone is trained correctly on the lure or survey area, with a high trigger speed, minimal/no delay between trigger events, and a large field of view (wide angle cameras are available). Good setup is crucial and cannot be over-emphasised to field crews. The detection zone often does not cover the entire area visible in the image, and its configuration varies between camera brands and models – check the camera's manual and do some experiments to see how your cameras work. The alignment of the detection zone once a camera has been set up can be checked using the camera's "walk test" mode (if available); make sure you only move your hand at the height you would expect the animal to move at. Alternatively, the image position can be checked using a handheld viewer or digital camera.

2. Deploying several cameras at a site and combining their detection histories (e.g. Forsyth et al. 2019; Davies et al. 2020; Stobo-Wilson et al. 2020).
3. Increasing the number of survey nights that each camera is deployed at a site (see Geyle et al. 2020).
4. Using a lure, such as scent-based lures (e.g. tuna oil, synthetic fermented egg, commercial olfactory lures), food-based lures (e.g. chicken necks, tinned cat food, peanut butter), or visual lures (e.g. tinsel, white feathers). Predators may be attracted by, but also wary of, visible lures. Placing the camera 2-3 m from the lure helps increase the field of view and so ensure that wary predators are photographed.
5. Installing the cameras on roads, tracks or other linear features that may be frequently used by predators such as dry river beds (e.g. Carter et al. 2019; Geyle et al. 2020; Wysong et al. 2020b). Detection rates are highest when the camera is angled slightly obliquely across the road; CISS (2021) recommend 23° from perpendicular.

Each of these approaches has benefits but may also impact the cost of camera purchase or surveys; methods 3-5 may also potentially violate one or more statistical model assumptions, such as population closure, equal probability of detection through time, equal probability of detecting all individuals, or random selection of survey sites; see Sections 4-6 for further discussion.

Artificial intelligence methods for identifying species from camera trap images are being developed, and have great potential to increase data processing efficiency (Nguyen et al. 2017; Norouzzadeh et al. 2018).

Data to record: Person who set up camera, GPS location of site, camera model and settings, date deployed, date removed, lure and placement details (if relevant), whether the camera was on standard or daylight savings time. Triggering the camera by walking in front of it after setting it up and before removing is helpful to provide assurance that it was working throughout the survey period, as is configuring the camera's time-lapse function to take a photo each day.

3.5 Scat surveys

Fox and cat scats may be collected as an index of predator activity, to estimate predator density by identifying individual predators using DNA genotyping, for predator diet analysis, or to survey disease within the population. See Case Study 8.2 for an example.

Advantages. Scat collection is non-invasive and so does not affect predator behaviour. Collecting scats is straightforward; relatively little training is required to learn to find scats and (generally) to attribute them to predator species. When paired with DNA genotyping, can be a highly efficient way of collecting capture-recapture datasets for foxes.

Limitations. Cat scats can be difficult to find, compared to fox scats, as they may bury their scats. Scat detectability also varies between substrates and observers, and with individual predator behaviour (e.g. some foxes may deposit scats near bait stations or along roads, while others may not). Identification of the predator species from field or laboratory observations is not always accurate – see Section 3.5.1 below. In addition, genotyping (when relevant) is costly and genotyping success varies with microclimate and weather conditions, as well as between individual predators. Hence, scat counts and genotyped individuals may not be representative of the entire population. It can require long periods in the field to collect sufficient sample sizes for density estimation.

Methods. Fox scats can usually be found relatively easily by walking or driving slowly along roads and tracks, and so scat survey transects are often aligned with these features. However, if the aim is to use scat counts to provide an index of predator activity, Güthlin et al. (2012) found that searching along linear but randomly-oriented transects provided a more precise estimate of the number of scats, and hence more power for detecting changes in relative scat abundance, even though more fox scats were found by searching along roads.

If the aim is to monitor predator activity or diet over a known period, or extract DNA from the scats, all old scats need be removed from the survey site and discarded prior to the collection period so that collected scats are of known age.

Trained scat-detection dogs can also be used to help locate fox and cat scats, as well as threatened species; see the Australian Conservation Dog Network (https://conservationdognetwork.com.au/) for more information. Trained dogs can be highly efficient and typically find more scats than humans (Baker et al. 2021; J. Pascoe, pers. comm), and the predator genotyping success of the scats they find is substantially higher (A. Weeks, unpublished). However, dogs may be unable to survey for as long a period each day as humans. See Case Study 8.2 for further discussion.

Scats should be placed into individual bags and preserved by air-drying or freezing after collection as quickly as possible. In cold or humid climates, freezing is more effective at preventing mould, and does not substantially reduce DNA genotyping success.

More rarely, scats are collected from traps where predators have been caught.

Whitten et al. (1980) found that synthetic scents could be used to induce marking behaviour (sometimes including defecation) on snow by foxes during the mating season in Canada, but we are not aware of this being used for scat surveys.

Fox and cat scats can transmit pathogens that cause serious diseases to humans. Good hygiene protocols when handling, storing and processing scats are essential – see Section 1.2. A guide to safely collecting predator scats can also be found at https://pestsmart.org.au/pest-animals/monitor-techniques/ (CISS 2021).

Data to record: Collector's name, unique ID for each scat, GPS location for each scat, collector's opinion on predator species, evidence for (and confidence in) predator species identification.

3.5.1 Identifying the predator species

The scats of different eutherian predators (e.g. cats, foxes and dingoes) can often be distinguished in the field or laboratory by their shape, location and/or smell. However, this process is not entirely reliable or consistent between observers. Independent verification shows that error rates range from 8 to 100% (reviewed by Wysong et al. 2019).

Another way to identify predators is to swab or scrape the surface of the scat. The sample is then sent for genetic testing to identify the predator species, e.g. through melt curve analysis (Berry & Sarre 2007). However, sometimes DNA cannot be extracted or the species still cannot be confidently identified. Wysong et al. (2019) provide a decision framework on how to identify and resolve potential errors in predator scat identification, using independent assessments of predator species identity in the field and laboratory, followed by DNA amplification for scats with uncertain identity. This approach provides a compromise between maximising identity certainty and minimising survey costs.

3.5.2 Identifying the predator individual

Individual foxes can be identified from DNA extracted from swabs or scrapes of scats, using microsatellite markers (Piggott et al. 2008; Marlow et al. 2015a; Le Pla et al. in review). Published microsatellite genotyping success from fox scats in Australia ranges from 26 to 65%, and varies with scat age, weather conditions, soil moisture and among individual predators.

Single Nucleotide Polymorphism (SNP) markers have also been developed for foxes in New South Wales (Watson et al. 2021) and Victoria (Weeks et. al, in prep.), and assays for individual identification of foxes from scat samples are currently being trialed.

These genotyping approaches are also potentially applicable to cats. We are not aware of any examples of cats being individually identified from scat samples, but microsatellite markers are available for cat tissue samples (e.g. Cowen et al. 2019).

3.5.3 Identifying the prey contents

The prey contents in scats are usually identified by hand, often by outsourcing to an expert. Scats are first processed by oven-drying at high temperatures (>60˚C) for a minimum of 24 hours to kill parasites. They are washed in individual nylon bags, so that only indigestible prey fragments remain (e.g. hair, bones, exoskeletons). Prey species are then identified by their macroscopic and microscopic features. Key reference material for identifying prey from predators scats in Australia include: Brunner & Coman 1974; Watts & Aslin 1981; Triggs & Brunner 2002.

3.6 Hair surveys (remote collection)

Hair samples can be collected from predators remotely using "sticky traps". Hairs can then be used to provide DNA samples from hair follicles for genotyping, so that density can be estimated (e.g. Berry et al. 2012; Berry et al. 2014; Marlow et al. 2015b). Sticky traps could also be used to simply survey predator presence/absence and estimate occupancy, but this is rarely done and other methods are likely to be more effective (Vine et al. 2009).

Advantages. Remote collection of hair provides a way of obtaining genetic samples from individuals without directly trapping or killing them. The impacts on the animal are therefore relatively small, as is the level of training required.

Limitations. Predators need to interact with survey device, and so surveys may miss wary individuals. This method may be more successful with foxes than cats (all examples above are for foxes). Multiple individuals and different species may leave hair at a single trap, making processing time consuming and could also potentially cross-contaminate DNA. Small lizards, birds and invertebrates may also become entrapped on the sticky trap surface and die.

Methods. For detailed descriptions on how to design sticky traps to collect predator hair samples, see Berry et al. (2014) for foxes and Johnston and Algar (2020) for cats.

The species the hair comes from can be identified using a microscope from whole mounts or cross-sections (Vine et al. 2009); see Section 3.5.3 for reference guides. Alternatively, DNA can be extracted from the hair follicle and the species identified using melt-curve analysis (Berry & Sarre 2007).

If hairs are being collected for DNA analysis, it is important to minimise DNA contamination and degradation. Survey points should be checked daily, hairs removed with sterilised forceps, and immediately placed in sealed containers, labelled and frozen at –20˚C. With this protocol, DNA was then successfully extracted from fox hair samples within four weeks (Berry et al. 2014). Alternatively, Vine et al. (2009) did not freeze the samples; in this case, DNA amplification rates were highest if extracted within 7 days.

Data to record: Observer's name, GPS location of sticky trap, date trap set, date checked.

3.7 Biological samples (invasive collection)

A variety of biological samples can be collected from foxes and cats that have been shot or collected dead (e.g. poisoned or roadkill). Trapped animals that will be released alive can also provide plucked hairs, whiskers, small amounts of ear tissue, and tooth observations, given ethics approval. Along with observations on breeding status (e.g. lactating) and health (e.g. mange), these samples can provide additional insights into the state of the predator population.

- **Ear tissue**. A small amount of ear tissue can be collected from live or freshly dead predators to provide DNA for genotyping, for example to identify individuals, relatedness or population impacts of control programs (e.g. Cowen et al. 2019; Watson et al. 2021). The sample should be collected using gloves as soon as possible after death (if relevant), stored in 95% ethanol and kept cool prior to DNA extraction.
- **Hair or whiskers**. Can be plucked from live or dead predators. These can be used for various purposes, including to identify bait consumption if biomarkers were deployed in baits (e.g. Fisher et al. 1999; Thomson & Algar 2000; Marks et al. 2003). They can also be used to quantify fox or cat diet using stable isotope analysis (e.g. Killengreen et al. 2011), or to provide genetic samples (see Section 3.6, above)..
- **Reproductive information**. The reproductive tract of female predators can be dissected and examined for information on reproductive status, breeding season, and litter size, for example by counting corpora lutea in the ovaries and uterine scars and embryos within the uterus. (e.g. McIlroy et al. 2001; Algar & Brazell 2008; Marlow et al. 2016). Scars from sequential litters are visible, and can be differentiated by experienced observers. Placental scar counts may overestimate litter size due to embryo resorptions, prenatal mortality or stillborn litters, or underestimate litter size if uterine tissues have regenerated postpartum.
- **Stomachs**. Stomachs may be dissected from dead predators and frozen until processing. They are then defrosted, the contents removed, placed in alcohol (optional), rinsed through a sieve, sorted and identified. This should be done either outdoors or near a fan as the smell is horrendous. Also, prepare where remains will be disposed of before beginning dissection. Data are typically recorded as the minimum number of individuals of each identified prey item, or as a frequency of occurrence (number of samples as a percentage of all samples). Prey may also be weighed, or weights assigned based on published values.

- **Teeth**. Foxes and cats can be aged by their teeth. For live foxes, wear on the upper incisors can be used to class animals as juvenile (<12 months) or adult (e.g. Kay et al. 2000). Dead foxes can be aged using the same method. In addition, for both foxes and cats, a canine tooth can be extracted from dead animals, cut diagonally and examined under a microscope to count the annular cementum layers (dark bands) – see Coman (1988) and Nakanishi et al. (2009) for method details. Foxes can live for 7+ years but often it is only possible to age individuals as less than or greater than 1 (or 2) years old. In Australia, foxes are born in September–October (Ryan 1976; Kirkwood et al. 2014), which can help inform age estimates.
- **Tissue (liver, muscle etc)**. A small amount of tissue can be sampled to examine diet over different time periods, using stable isotope analysis. This approach has been used successfully for foxes and cats overseas (e.g. Meckstroth et al. 2007; Killengreen et al. 2011) and is being trialed in Australia.

Data to record: GPS location of the collection site, the animal's sex, weight, age (juvenile/adult), breeding condition (e.g. lactating), health condition (mange, injuries), the method of collection, and the name and contact details of the collector.

3.8 Trapping

Foxes and cats may be trapped as part of a lethal control program, or for release alive after fitting with tracking collars (Section 3.8). See Case Studies 8.1 and 8.3 for examples. Legal restrictions on trapping, and on releasing, foxes and cats vary between states and territories, and on whether trapping is being conducted for management or research. As for all survey methods, it is imperative that all approvals and correct equipment are obtained before commencing a trapping program.

Methods. Standard operating procedures for trapping foxes and feral cats using cage traps and padded jaw traps (sometimes called foot-hold traps, soft-jaw traps) in Australia can be found at https://pestsmart.org.au/ (CISS 2021).

The likelihood of trapping foxes and cats, and the class of animals trapped, will depend on the trapping method, season, environmental conditions, lure and the skill of the trapper. The trapped sample may not represent underlying structure of the population (Kay et al. 2000).

If the aim is to trap adult foxes to fit collars, non-dispersing adults can be most easily trapped after the autumn dispersal season but before cubs become highly mobile in early summer, i.e. between May/June and November (Roshier & Carter 2021; B. Hradsky, pers. obs.). If using this strategy, it is best to target early in the period prior to the commencement of lactation in breeding females as they are more prone to stress, injury or death once trapped. Else, the traps should be checked frequently or constantly monitored (e.g. using VHF signal). Other studies have targeted the opposite season, i.e. January – July, to avoid the breeding season (e.g. Kay et al. 2000; Saunders et al. 2002).

Food lures may be most effective when prey availability is low, e.g. cage trapping success rates for cats in central NSW peaked when rabbit numbers were lowest (Molsher 2001).

Cage traps can be successful for trapping feral cats that are accustomed to scavenging around human infrastructure, such as urban or agricultural populations, or animals living around rubbish tips or mining camps (e.g. Molsher 2001; Short et al. 2002; Cowen et al. 2019). Cages are also a good option in areas with non-target species that are particularly vulnerable to foot-hold traps, such as small hopping macropods (Short et al. 2002). Cage traps may be closed during the day to reduce captures of carnivorous birds, such as ravens. Cage traps tend to catch younger, more naïve individuals, than footholds (Short et al. 2002).

Foot-hold traps with offset, rubber-lined jaws, swivels and springs (e.g. Victor soft Catch trap #1 ½) are generally much more successful than cages for trapping introduced predators living in natural environments. Nonetheless, foot-hold trapping success rate for foxes and cats is typically in order of 0.01–2% (Kay et al. 2000; Saunders et al. 2002; Short et al. 2002), and may decline as predator density declines (Kay et al. 2000). Trap efficacy can also be affected by trapper expertise, the tension of the pan-spring, season, environmental conditions (e.g. heavy rain), interference by non-target species, and previous exposure of targeted individuals to trapping (Surtees et al. 2019).

Birds, reptiles and other native and non-native mammals may be captured in traps set for predators either incidentally or after being attracted to the trap, particularly if food is used as a lure. Elevated platforms are sometimes used to reduce bycatch encounters and inference, with both cage and foothold traps. Birds, smaller mammals and hopping macropods can be severely injured or killed if caught in foot-hold traps (Surtees et al. 2019). The decision on whether to use foot-hold traps will depend on context. See Surtees et al. (2019) for detailed recommendations on how to reduce the likelihood of trapping and injuring bycatch when trapping predators.

Data to record: Trapper's name, GPS location of trap, date, time, trapping method, species, sex, weight, age (from tooth wear and size), breeding condition (e.g. lactating), health condition (e.g. mange). Hair or ear tissue samples may also be collected if approved (Section 3.7).

3.9 Tracking collars

Tracking collars can be fitted on foxes and cats to study their interactions with baits, survival, habitat selection, movements, and (in combination with other survey methods) their density or diet. See Case Studies 8.1 and 8.3 for examples.

Advantages. Tracking data can provide information on predator ecology that cannot be obtained in any other way, such as high resolution information on habitat use at different times of day and precise movement rates. This data can be fundamental to designing effective predator monitoring and management programs.

Limitations. Each step in the process of obtaining appropriate permits and approvals, catching enough individuals to have a representative sample, fitting them with functional collars for long enough to collect useful datasets, and retrieving the data, can be difficult and time-consuming. The latter are also expensive and require some expertise. Further, trapped predators may not be representative of the whole population (e.g. young or vulnerable predators may be more likely to be attracted to food lures at traps, while male cats may be overrepresented in foot-hold traps).

Methods. The predator first has to be trapped (Section 3.8), or captured using a spotlight, net and/or dog (McGregor et al. 2014). Tracking collars may be fitted onto alert but restrained predators (Bengsen 2014), sedated predators (Hradsky et al. 2017b), or a mix of both depending on the situation (McGregor et al. 2014).

Three primary types of collar are used to track foxes and cats. The choice of collar type will depend on the study question and the primary field constraints. In all cases, the total weight of the collar needs to be no more than 5% (or ideally <3%) of the predator's body mass. Some key considerations are highlighted below:

- **VHF-only**. VHF collars have long been used to track foxes and cats (e.g. Meek & Saunders 2000; Molsher et al. 2005; Marks & Bloomfield 2006; Carter et al. 2012; Molsher et al. 2017). They are comparatively cheap, allowing larger numbers of collars to be deployed, for example in survival studies. Their batteries also last a long time and the collars are relatively light. However, it can be very challenging to regularly collect good location data in rugged environments that impede signal transmission, or if predators range across large distances relative to the signal strength of the collar (which is often less than 1 km). Each predator has to be found every time a location needs to be recorded, which is time-consuming; as a result, the estimated home range sizes tend to be much smaller from VHF-collared than GPS-collared animals in the same region (Towerton et al. 2016b). Animals that have dispersed or made forays outside their home range can also be difficult to find. Approaching the animal to collect data may also affect its behaviour.
- **VHF + GPS**. These collars have become increasingly common over the last decade. The collar collects and stores the predator's location at programmed intervals. Data recovery depends on the collar's features, options include:
    - Killing the predator at the end of the study (e.g. by trapping, shooting, poison baiting or lethal implant). This is the most economical in terms of battery-life and collar weight. However, it runs the risk of not being able to find and/or recapture the predator. If shooting, there is also a risk that the shot will destroy the collar.
    - Using a mechanical 'drop-off' feature, so that the collar falls off the predator. This may occur at a pre-programmed date and time, or be triggered remotely using a UHF signal. The drop-off adds weight to the collar and reliability varies. The collar needs sufficient battery for the drop-off to work. Other methods include using corrosive bolts (Thalmann 2013) or stitching collar material with degradable dental sutures. You still need to be able to find and recover the collar using its VHF signal.
    - Tracking the predator using VHF signal and then remote-downloading the data. This has the advantage that data can be recovered mid-deployment, which helps you locate the predator again in the future, and ensures that at least some data are recovered if the predator later disperses. However, you need to be able to find the predator's VHF signal, and data transfer uses battery power.
    - Automatically transmitting data via the mobile phone network. As this does not depend on finding the VHF signal, it can really help with locating missing individuals; however it adds weight to the collar, consumes battery power and depends on coverage being available.
- **Satellite**. Satellite collars have recently become small enough to be practical for foxes and cats that weigh at least 3 kg, and have the advantage of automatically transmitting the location data, making it possible to find animals that have gone on long forays. They are therefore a particularly good option in arid environments where predators move long distances (Roshier & Carter 2021), or when trapping animals is very difficult (and so every data point is precious). However, their battery life is typically shorter than VHF + GPS collars.

Other features can be added to the collar such as an accelerometer (to record motion and infer behaviour) or thermal logger (to record microclimate). Unique reflective symbols can also be fitted to the collar (Comer et al. 2018) or ear tags (B. Hradsky, unpublished), so that individuals can be recognized in camera survey photos. Collar-mounted video cameras are also possible (McGregor et al. 2015a) but are not readily available, and may need to be custom developed.

4. General guide to survey design

Here we cover some key points common to the design of many introduced predator surveys in Australia. However, foxes and cats are surveyed for a wide variety of reasons, and survey design is a big topic. There are many excellent, detailed guides on population monitoring available, such as Elzinga et al. (2001) and McComb et al. (2010); both are freely available online. Caley (2015) also raises some valuable topics.

There is no universal "best" approach for monitoring introduced predator populations. A series of decisions need to be made about the overall design, population metrics, survey method and analytic approach made, based on:

- The purpose of the monitoring program and how accurately the state (or change in state) of the population needs to be understood, for the given scenario. Where relevant, the survey program might form part of S.M.A.R.T. targets (Specific, Measurable, Achievable, Relevant and Timely) for a management intervention, to help evaluate whether the intervention is meeting its intended objectives.
- Whether the survey method would provide other useful information, e.g. on rate of population turnover, the presence or density of other predator and prey species, and/or predator diet.
- The suitability of the method for the study conditions, available expertise and resources.

Prior to commencing any survey program, we strongly recommend:

- Consulting with someone who has used similar methods to survey the same species, ideally in a similar type of habitat.
- Getting expert statistical advice on whether the design is likely to provide meaningful data that addresses the question(s) of interest (e.g., whether the study is likely to have sufficient statistical power).
- Planning well in advance to allow sufficient time to obtain permits and approvals, order equipment etc.
- Conducting a pilot study.

4.1 Measuring the effects of management

Many fox and cat monitoring programs aim to measure how the predator responds to a management intervention, such as lethal control of the focal predator (Fancourt et al. 2019), control of other predator or prey species such as dingoes or rabbits (Scroggie et al. 2018; McGregor et al. 2020), or other management activities such as prescribed burning (Payne et al. 2014; Hradsky et al. 2017a).

Plausible designs for quantifying the effects of management on predator populations and predator behaviour depend on whether non-treatment landscapes are available for comparison and whether management has already commenced.

The most robust overall approach is a replicated *before-after control-impact* (BACI) design (Stewart-Oaten et al. 1986; Christie et al. 2019). This involves conducting surveys repeatedly before and after management takes place, in several *treatment* landscapes (where management will occur, also known as the impact landscapes) and several *non-treatment* landscapes (where management will not occur, also known as reference landscapes or experimental controls).

BACI designs are an effective way of identifying the *direction* (increase or decrease) and *size* (by how much) of a response. They account for:

- **Spatial variation**: differences that already existed before management began (due to differences in habitat type, food availability, road density etc.).
- **Temporal variation**: changes that may take place over time anyway without management (because of rainfall, season etc.).

Increasing the number of landscapes helps improve the program's *statistical power*, i.e. the likelihood that the monitoring program will detect an effect of management if it really exists.

The choice of which landscapes receive the management intervention should be *randomly allocated*, to avoid other factors confounding the results (as in, causing differences between populations in the treatment and non-treatment landscapes that cannot be distinguished from the effects of management). For example, it can be problematic if the treatment landscapes all have higher rainfall or better-quality habitat than the non-treatment landscapes.

Strong examples of replicated BACI predator monitoring designs, with repeat survey sessions before and after management include Molsher et al. (2017). However, it is often impractical to conduct a fully replicated BACI design in a predator monitoring context. Compromises may be necessary, although their limitations need to be acknowledged. For example, management landscapes may need to be spatially clustered, preventing random allocation of treatments (e.g. Robley et al. 2014); management may need to target specific populations, preventing random allocation of treatments or limiting replication at the landscape scale (e.g. Moseby et al. 2009; Comer et al. 2018); or project timelines may limit the number of survey sessions that can be conducted pre- or post-management (e.g. Hradsky et al. 2017a; Fancourt et al. 2021).

If non-treatment landscape(s) are not available, a *before – after* design is a good option (e.g. Marks et al. 2009; Newsome et al. 2014). This involves monitoring populations in treatment landscape(s) before and after the management intervention; changes after management occurs are attributed to the management intervention. Conducting several survey sessions before management begins strengthens this design by providing baseline information on how much the system varies through time.

Alternatively, if management has already begun, a *control – impact* (also known as space-for-time, or impact versus reference) design is a good option (e.g. Fancourt et al. 2015; Moore et al. 2018). This involves comparing treatment and non-treatment landscapes after management has already taken place in the treatment landscape(s); differences between the landscapes are attributed to the management intervention. Having several treatment and several non-treatment landscapes strengthens this design by providing information on how much the baseline system and the response to management varies across space.

If non-treatment landscapes are not available and management has already begun, monitoring may just focus on assessing the population in the treatment landscape(s) at one point in time or how it changes over time (e.g. Field et al. 2005). This is the simplest design, but makes it difficult to know whether any changes observed are actually due to management.

In all cases, designs can be strengthened by:

- **Temporal replication:** Repeating the survey sessions through time to identify how much populations are changing (with several surveys both before and after management, if applicable).
- **Spatial replication**: Having several treatment (and if applicable, non-treatment) landscapes, to identify whether the effects of management are consistent.
- Ensuring that replicate landscapes are **spatially independent**; i.e., far enough apart that predators can't move between them and, if applicable, predators in the non-treatment landscape(s) are not affected by the management intervention.

Studies sometimes end up being replicated when different researchers address the same questions in different landscapes. Using well-established or comparable methods, recording the details of methods so that the approach can be reproduced, and collecting data on other factors that might influence results (such as rainfall, rabbit abundance or other management interventions that also took place) can therefore be very valuable.

4.2 Monitoring populations for other reasons

In other situations, the aim of a monitoring program is to assess the current state of the population or how populations vary over time or across space, without inferring anything about the effects of management. For example, a manager or ecologist may wish to know whether current suppression efforts have successfully reduced the introduced predator population to a target level, whether the population is approaching a pre-determined threshold that indicates the need to commence predator control, or just understand the baseline prevalence, density, distribution or behaviour of the predator population. The design may involve capturing a snapshot of the population at a single point in time, or monitoring how it varies through time. In such cases, an experimental design is not necessary but other assumptions associated with the statistical analysis still need to be met to ensure that the population metrics can be trusted.

A third category of monitoring programs involves establishing whether the predator population is at zero or not. Knowing whether any individuals are present is crucial during and after an eradication program (e.g. Rout et al. 2009; Kirkwood et al. 2014; Algar et al. 2020), and in situations where predators could be invading a new area where they did not previously occur (Sarre et al. 2013; Allen et al. 2017). In this scenario, high detection rates, high coverage of the area of interest and accurate species identification are crucial (e.g., Marks et al. 2014).

4.3 Survey effort and statistical power

A general challenge with monitoring foxes and feral cats in Australia is that these predators can occur at low densities, range over wide areas, are largely nocturnal, and are usually wary of humans. They may also avoid anything unusual, such as cameras or traps (or conversely, be attracted to them). The combination of low predator densities and poor detection rates mean that there can be a lot of zeros in the survey data, particularly when densities are very low, such as after a lethal control program. This can lead to high uncertainty around estimates, low statistical power to detect any changes, and an inability to draw any useful conclusions from the data. This can be very frustrating after spending a lot of time and money on field surveys.

Increasing the survey effort is a key way of addressing this problem, as it reduces the uncertainty in the data (i.e. increases the precision of the estimates). The survey effort can be increased in various ways, including increasing the number of study landscapes, the number of survey sites within each landscape, the number of survey sessions at each site, the number of surveys within each session at each site, and/or the likelihood of detecting the species in a single survey.

The relative value of each approach will depend on the study design (Christie et al. 2019). For example, there may be tradeoffs in cost and feasibility between increasing the likelihood of detecting the predator if it is really present at the site (e.g. by surveying each site more times within a session or buying more expensive survey-cameras) against surveying more sites (Southwell et al. 2019; van Hespen et al. 2019).

A *power analysis* can be used to help identify the best way of designing a study to maximise the chance of detecting a population change if one really occurs. For example, if the aim is to be able to detect an 80% decline in a predator population over six months, a power analysis can help decide whether it is better to invest effort in surveying more sites or surveying each site more times. For examples of power analyses using camera survey designs, see van Hespen et al. (2019) for a predator activity index, and Geyle et al. (2020) for predator occupancy.

Useful resources on designing predator studies and testing statistical power include Guillera-Arroita and Lahoz-Monfort (2012), Ellis et al. (2015), Southwell et al. (2019) and Southwell (in review).

4.4 Site selection

In general, survey sites within a study landscape should be chosen using probability-based sampling, such as choosing them totally at random, at random within different levels of stratifying covariate (such as habitat type), or at random but with a minimum distance between sites.

Nonetheless, a common approach for surveying predators is to selectively place sites in locations that predators also select. Bait stations, transects for spotlighting and track surveys, survey cameras, sticky wickets and live traps are all commonly set up along roads or other linear features such as dry waterways, spurs and ridgelines. Selectively placing sites in these locations can help increase the detection rate, an important consideration for statistical power (Section 4.2). It also practical: spotlighting, track surveys and scat surveys may be difficult to conduct off-road, unless in very open (and ideally sandy) country, and baits, sticky wickets and traps often need to be deployed near roads so that they can be checked regularly.

However, non-random selection of sites can potentially introduce another problem – *bias* – where the value estimated from the survey data is systematically higher or lower than the true value of the population (Anderson 2001). This can cause problems if:

1. Predator densities need to be estimated accurately or eradication of the population needs to be reliably ascertained; or
2. If the bias differs in an unknown way between treatment and non-treatment landscapes, or before and after management. For example, surveying along roads could cause problems if predators use roads more in some locations than others, if predators that use roads are more likely to be affected by management than predators that do not use roads, or if management affects how predators use roads (see Section 5.3 for a more detailed explanation of this issue).

The decision whether to place survey sites at random or selectively-chosen locations (or a mix of both) will depend on the context-specific tradeoff between the value of higher detection rates for more precision, the implications of potential bias for estimate accuracy and data interpretation, and the feasibility of conducting surveys off-road.

Of all the survey techniques, survey cameras are often the easiest to deploy at random locations, even in very rugged country; although it is generally more efficient to set them up along roadsides (Carter et al. 2019). In some cases, predator detection rates on survey cameras are many times higher when placed along roads than off-road, leading authors to strongly recommend roadside surveys in both temperate forests (Towerton et al. 2011; Geyle et al. 2020) and semi-arid regions (Carter et al. 2019; Wysong et al. 2020b). In others, road-side placement has little effect on detectability or variable effects at fine spatial scales (Hohnen et al. 2020), or authors have been able to obtain adequate detection rates by using other approaches such as lures, several cameras at a site or long survey sessions (e.g. Rees et al. 2019; Davies et al. 2020; Stobo-Wilson et al. 2020). For further discussion on options for improving detection rates in camera surveys, see Section 3.4.

5. Monitoring predator populations without identifying individuals

To monitor introduced predator populations across large landscapes or over long periods of time, people typically need to be able to track changes in populations without trapping and marking individuals. The most common and broadly applicable methods of conducting this type of survey involve either directly observing the predators in spotlighting or camera surveys, or indirectly observing predator signs by surveying for tracks or scats (Table 1). Section 3 introduces each of these methods, including their benefits and limitations.

If the aim is to monitor predator responses to a lethal baiting program with ground baits, bait take rate is widely used in ground-baiting programs for foxes. This method is efficient but particularly prone to bias (Section 3.1), and so is often used in combination with other monitoring methods. Other approaches specific to particular control methods include the number of poison ejectors triggered (e.g. Evans et al. 2021) and catch per unit effort from trapping (e.g. Marlow et al. 2016).

This section focuses on how to design surveys to monitor predator populations without identifying individuals ('unmarked' populations). However, non-invasive methods for identifying individual predators are also becoming increasingly affordable and practical; see Section 6.

5.1 Survey design

5.1.1 Site spacing

Most ways of analysing data from unmarked predator populations assume that sites within each landscape are spatially independent. Sites should therefore be spaced so that individuals cannot (or are unlikely to) move between them. If sites are too close, individuals can be detected at several sites, meaning that estimates will be unbiased but the uncertainty around them will be underestimated (MacKenzie & Bailey 2004).

Local information on predator movement is very useful if available. Fox and cat home range size varies by more than an order of magnitude across Australia, depending on resource availability (Section 2). As a rule of thumb, sites and transects are often spaced at least one or two times the diameter of an average predator's home range apart (e.g. Marks et al. 2009; Robley et al. 2016). However, such approaches do not entirely guarantee spatial independence (e.g. Marks et al. 2009). Home range size can also vary widely between individuals within a landscape, and some may not even have a fixed home range (Roshier & Carter 2021). Methods to adjust nonspatial model residuals for spatial autocorrelation (F. Dormann et al. 2007) and spatially explicit models have been developed (e.g. Johnson et al. 2013), but can be are computationally intensive and have rarely been implemented for invasive predator monitoring (although see Ramsey et al. 2015; Rees et al. in review).

5.1.2 Survey session duration

Surveying a site more than once within a survey session is often a good idea – it increases the likelihood of detecting predators if they are present and allows you to estimate detectability. However, most analytic approaches assume that the population was 'closed' during a survey session (i.e. no deaths, births or migrations during the session). Therefore, there is a tradeoff between conducting surveys over long periods to reduce false absences and over short periods (typically days – weeks) to better reflect the closed population assumption.

5.1.3 Pseudo-replication

Field studies often involve surveying sites that are clustered in space (e.g. several sites within each treatment landscape) or repeatedly conducting survey sessions at the same site (e.g. before and after management). This can introduce pseudo-replication into the data, as data from clustered or the same sites are more similar to each other than they would be if the sites were chosen entirely at random each time.

Pseudo-replication can be explicitly tested and accounted for when the data are analysed, for example by fitting random intercept effects (i.e. mixed modelling approaches), fitting autocorrelation terms or using dynamic occupancy models.

5.2 Data analysis: naive indices

The simplest way of monitoring unmarked populations is to use raw or naïve values from survey data as a population index, without adjusting for imperfect detection. Data can be aggregated in various ways, including:

- **naive occupancy**: proportion of sites where the species was detected at least once in the survey session.
- **activity indices**. These fall into two broad categories:
  - **detection-rate based indices**: proportion of repeat surveys (within a survey session) where the species was detected at least once.
  - **count-based index**: based on the number of 'independent' observations of predators per survey.

Table 2 provides examples of Australian studies that have used these types of approaches to summarise introduced predator monitoring data from track, spotlighting, camera or scat surveys. See also Case Study 8.1.

Naive occupancy and detection-rate based indices give binary data (0s/absences and 1s/presences), and may be reported as the proportion or percentage of sites where the species was detected.

Count data can be reported as the mean or median value. They can also be aggregated in various ways. For example, the General Index (also known as a 'modified form of the Allen Index') involves taking the mean count across all sites each survey, and then averaging across all surveys in the session (Allen et al. 1996; Engeman 2005). Another alternative is to assume the count data have a non-linear relationship with density and transform accordingly:

$$f = 1 - e^{-\bar{x}}$$

where f is the mean frequency and $\bar{x}$ is the mean density (Caughley 1977).

A special case is when count-based index surveys are conducted before and after a known number of individuals are removed over a short period (e.g., by shooting or poison-bait ejectors), and the index is calibrated to estimate absolute population size:

$$N_1 = I_1C / (I_2 - I_1)$$

where I_1 is the index before removal, I_2 is the index after removal and C is the number of animals removed (Caughley 1977). For examples, see Marks et al. (2009), Thompson and Fleming (1994).

Changes in indices over time or space can be analysed in a variety of ways, including generalised linear models or generalised linear mixed models. The latter include a random intercept to account for sites that are clustered in space and/or repeat surveys of the same sites.

Fox tracks surround a conspicuously cached rabbit. Image: Hugh McGregor, Arid Recovery

Table 2. Examples of studies that have used naive indices derived from different survey methods to monitor introduced predator populations in Australia.

	Naïve population index		
		Activity index	
Field survey method	**Naïve occupancy**	**Detection rate based index**	**Count-based index**
Track surveys	Southgate et al. 2007; Moseby et al. 2011; Towerton et al. 2016a	Payne et al. 2014; Norton et al. 2015	number of 'individual' tracks per unit distance or per plot. 'Individuals' are distinguished by their footprint size, movement pattern on and off the transect, distance between tracks etc. Burrows et al. 2003; Marks et al. 2009; Marlow et al. 2015b
Spotlighting	Vine et al. 2009	–	number of individuals seen per unit distance Banks et al. 1998; Kay et al. 2000; Saunders et al. 2002; Molsher et al. 2017 Or per unit area, if adjusted by width of surveyed area Marks et al. 2009
Camera surveys	Hradsky et al. 2017a; Fancourt et al. 2021	Payne et al. 2014; Davies et al. 2020; Hohnen et al. 2021	number of 'independent' trigger events per n day(s). 'independent' has been defined as anything from 5 to 60 min apart Towerton et al. 2011; Lazenby et al. 2015; Geyle et al. 2020
Scat surveys	–	–	Number of scats per unit distance Roberts et al. 2006; Le Pla et al. in review

5.3 The limitations of naive indices

While seemingly straightforward (and very commonly used; Table 2), naïve occupancy and activity indices need to be collected and interpreted with caution, if the aim is to compare populations through time or across space. This is because the likelihood of recording predators in a survey is influenced by not only by the *density* of predators (which is often the primary interest), but also the *detection process*, which can be affected by the observer, the predators' behaviour and the environment, as well as random variation (Anderson 2001). This can confound the index, leading to the interpretation that there is a difference in density, when actually there is a difference in detectability (Anderson 2001).

Some factors that affect detectability can be controlled relatively easily in the survey design, e.g. by using the same observer in all surveys, using observers with similar levels of expertise, or only conducting surveys in good weather conditions.

However, the effect of predator behaviour on detectability needs careful consideration. Three potential issues are particularly worth considering:

1. **Predator behaviour (and hence detectability in surveys) can vary between sites or landscapes or through time, for reasons unrelated to management**. For example, cats are twice as likely to use roads (compared to off-road areas) in spinifex grassland than in open woodland in central Western Australia, probably because they dislike being spiked by the spinifex; this making them more detectable in road-based surveys in this habitat type (Wysong et al. 2020a). Similarly, increased movement of foxes during the dispersal and breeding season can lead to peaks in fox detections at these times of years, e.g. on camera traps (Hradsky et al. 2017a; Carter et al. 2019) and spotlighting surveys (Vine et al. 2009). Using a BACI design (Section 2) can help account for this, provided that the variation is not caused by the management intervention or another factor that is the focus of the study (see below).
2. **Management may affect predator behaviour in a way that alters the likelihood of observing the predator (and/or other species)**. This will lead to the index overestimating or underestimating the effect of management. For example, foxes that survive a baiting program may rapidly learn to avoid bait stations, leading to a decline in bait take and an overestimation of how effectively the fox population has been suppressed. Or if cats avoid roads when dingoes are using them, but management causes a decline in dingo density, cats may begin using roads more and so become more detectable in road-based surveys, which could be misinterpreted as an increase in cat density.
3. **Some individuals may be more likely to be recorded than others, and the same individuals may be more (or less) likely to be affected by management**. This will lead to the index overestimating (or underestimating, respectively) the effect of management on the population. For example, if some foxes use roads more than others, these foxes might be more likely to be recorded in road-based surveys and more likely to take poison baits deployed along roads than foxes that avoid roads, and so would show a bigger response to baiting than the overall population. Similarly, predators that are seen in spotlighting surveys may also be more likely to be seen and shot during a control program. Using a survey method that is as independent of the management method is thought to help reduce this issue (e.g., using spotlighting to measure the effects of a baiting program, rather than bait take).

5.4 Data analysis: adjusting for imperfect detection

If knowing the true density, abundance or occupancy of the predator population is important, or variation in detection rates is of concern and cannot be eliminated in the survey design, repeat surveys of each site within a survey session can be combined with hierarchical statistical modelling approaches to explicitly distinguish differences in predator density (or occupancy) from variation in predator detection rates.

Estimating true density or occupancy through a hierarchical modelling approach is more computationally intensive than using a naïve index, and so usually requires more observations (aka more survey effort) and a greater amount of statistical expertise to fit the models. Hierarchical models can also be impossible to fit when there are too few observations, for example if predator densities are very low following management.

Here we cover the three broad approaches for adjusting for imperfect detection with survey data from unmarked predator populations.

5.4.1 Occupancy-detectability models

Occupancy-detectability models (commonly known as occupancy models) can be used to estimate the proportion of sites that are actually occupied by predators, as opposed to the proportion where they were detected (the "naive" occupancy described in Section 5.2). Occupancy models use repeat surveys of sites within a session to estimate detectability, and can test whether detectability varies with covariates of interest, such as management, habitat type, observer etc.

Occupancy models can be fitted to data from camera surveys (e.g. Lazenby et al. 2015; Hohnen et al. 2016; Robley et al. 2016; Comer et al. 2018; Wysong et al. 2020b), spotlighting surveys (e.g. Field et al. 2005), track surveys or scat surveys, provided that sites were surveyed several times within each session.

A *single season occupancy model* (MacKenzie et al. 2002) is used if only one survey session was conducted at each site. *Dynamic occupancy models* (MacKenzie et al. 2003) can also be used if sites were repeatedly surveyed in different sessions, e.g. before and after a management intervention; these models estimate the probability of sites becoming occupied or extinct between survey sessions, as well as detectability and initial occupancy.

See Popescu et al. (2012) for discussion on the benefits and limitations of using dynamic occupancy models to detect changes in wildlife populations with BACI studies.

See Neilson et al. (2018) for discussion on interpreting occupancy models fitted to camera survey data.

Models can be fit using the R package 'unmarked' (Fiske & Chandler 2011) or the software PRESENCE (https://www.mbr-pwrc.usgs.gov/software/presence.html)

5.4.2 Distance sampling

Distance sampling is a well-established method for estimating predator density from count and distance data in spotlight surveys. See Buckland (1985); Buckland et al. (1993); Buckland et al. (2004) for method details, and Case Study 8.3 and Newsome et al. (2014) for Australian examples. Distance sampling also has potential application to camera surveys (Howe et al. 2017); see Gilbert et al. (2021) for further discussion.

In the context of spotlighting transects, distance sampling models estimate the probability of the observer(s) detecting animals in relation to the animal's distance from the transect (and other factors that might affect detectability), then use these estimates to adjust the overall counts and derive density.

Important assumptions and considerations for estimating predator density using distance sampling include:

1. The analysis assumes that any predator on the transect is recorded with certainty and that all predators are detected at their initial location; failure to meet these assumptions leads to negative bias. These assumptions are often violated in predator surveys, as the predators tend to move as you approach.
2. For predators detected either side of the transect, the perpendicular distance of the animal from the transect needs to be accurately recorded (ideally measured with a range finder). For example, if someone conducting a spotlight survey sees an animal 50 m ahead, yet the animal is only 2 m from the road, the distance should be recorded as 2 m. Alternatively, the distance and angle from the observer can be recorded, and the perpendicular distance calculated later.
3. Numerous animals need to be sighted to generate estimates of detection probability. As a rule of thumb, 60–80 sightings for the overall survey dataset is adequate (Buckland et al. 1993). This makes distance-sampling approaches difficult to apply in areas with low densities of feral cats and foxes.
4. Factors that might bias detection rates of predators during the survey need to be considered. Such biases are not necessarily a problem provided they are either measured and incorporated in analyses or are sufficiently similar across sites and through time. Potential biases that should be recorded include the experience of the different observers, weather conditions, moon phase, distance to road and season.
5. Habitat should be considered, especially in terms of whether some habitats have higher densities of predators, or predators are more visible. For example, it may be appropriate to compare two transects, both of which were in a mix of open and forested habitats, but would not be appropriate to compare a transect in open habitat to one in a forest unless differences in detectability are explicitly accounted for.

Distance sampling models can be fit using the R package 'Rdistance' (McDonald et al. 2019), the R package 'unmarked' (Fiske & Chandler 2011) or the software DISTANCE (Thomas et al. 2010). All contain detailed user guides and tutorials. Thomas et al. (2010) also contains useful information on how to design surveys for distance sampling.

5.4.3 Estimating abundance from camera survey data

Being able to estimate the actual abundance or density of unmarked populations from camera trap data is highly desirable but challenging. Four main methods are currently being used to estimate the abundance or density of unmarked populations from camera survey data; all involve numerous assumptions and specific survey designs (Gilbert et al. 2021):

- The Royle Nichols abundance-induced heterogeneity model (Royle & Nichols 2003). Examples of applications to introduced predator studies in Australia include Bengsen (2014), Davies et al. (2020) and Fancourt et al. (2021).
- The random encounter model (Rowcliffe et al. 2008).
- N-mixture models (Royle 2004); see Case Study 8.1 for an example.
- Unmarked spatial capture-recapture modelling approaches. For methods, code and application to Australian foxes, see Ramsey et al. (2015) and Forsyth et al. (2019).

See Gilbert et al. (2021) for an introduction to these and other abundance-estimation approaches, and a review of their assumptions, use and limitations in monitoring wildlife populations.

6. Monitoring predator populations where (some) individuals can be identified

In many situations, it is useful to know the true density of a predator population (the number of predators per km^2) rather than occupancy or a population index. Spatial capture-recapture models with fully or partially-marked populations are currently the most reliable method to estimate animal density (Borchers & Efford 2008; Royle & Young 2008). They require information on individual encounters – where and when individual animals were recorded (e.g. in traps, on cameras, or from genotyped scat or hair samples). *Capture-recapture* refers to models where every detection is identified to the individual level, *mark-resight* is an extension of these models where only a portion of the population are identified to the individual level (for example, if cats with tabby or other unique markings are individually identified, but black cats are not). For simplicity, we use capture-recapture as a general term to encompass both in the following section.

Spatial capture-recapture methods can provide information on predator density and how it varies across space or with covariates of interest such as habitat type or bait intensity. They can also provide information on animal movement, resource selection, survival and population turnover (Royle et al. 2018), which can have further important implications for management. For example, if a lethal control program does not reduce predator density, it is useful to know whether this is because the original individuals are still alive (indicating that control was ineffective) or because new animals have moved into the management area (indicating that control was effective but needs to be conducted over a larger area) (e.g. Piggott et al. 2008).

Spatial capture-recapture methods can be used to estimate predator densities in a wider variety of scenarios than spotlighting with distance sampling (Section 5.4.2) and their assumptions are more likely to be met than those required to estimate density for completely unmarked populations (Section 5.4.3). However, they have intensive data requirements. Non-spatial capture-recapture methods are commonly used to estimate population size from equivalent datasets when data is insufficient (e.g. Marlow et al. 2015a). However, the area from which animals are sampled is usually poorly defined, making density estimation problematic, particularly for species that range widely such as foxes and cats (Otis et al. 1978; Borchers & Efford 2008).

6.1 Field survey methods

Spatial capture-recapture datasets for cats and foxes do not usually involve catching and marking individuals, unless they are already being collared for another purpose (e.g. Carter et al. 2019; see also Case Study 8.1). Instead, these predators are typically surveyed and identified non-invasively.

Feral cats

In any feral cat population, at least some individuals can be uniquely identified by their coat markings. Therefore, capture histories for cats can be obtained using survey cameras. See Section 3.4 for information on using cameras to survey predators, Section 6.2 for information on how to design this type of study, and Green et al. (2020) for a review of using camera traps to estimate densities of partially-marked populations.

Cats with tabby and other unique coat markings are identified by hand; there is no standard method for doing this, but we suggest keeping libraries of images or drawings for each individual and notes on how each individual was identified to enable re-evaluation. Black cats are typically relegated to the 'unmarked' category in mark-resight models; black cats can comprise anywhere from 0% to >40% of detections, depending on the study population. Ginger cats at night are also typically over-exposed on infrared cameras, making them extremely difficult to identify.

Studies that have used this approach to estimate cat density in Australia include McGregor et al. (2015b), Stokeld et al. (2016), Rees et al. (2019), Taggart et al. (2019), Hohnen et al. (2020) and Davies et al. (2021). See also Case Study 8.2.

DNA from hair samples or scats could also be used for to collect capture-recapture data for cats; this was attempted unsuccessfully by Marlow et al. (2015b), but several other studies are in progress.

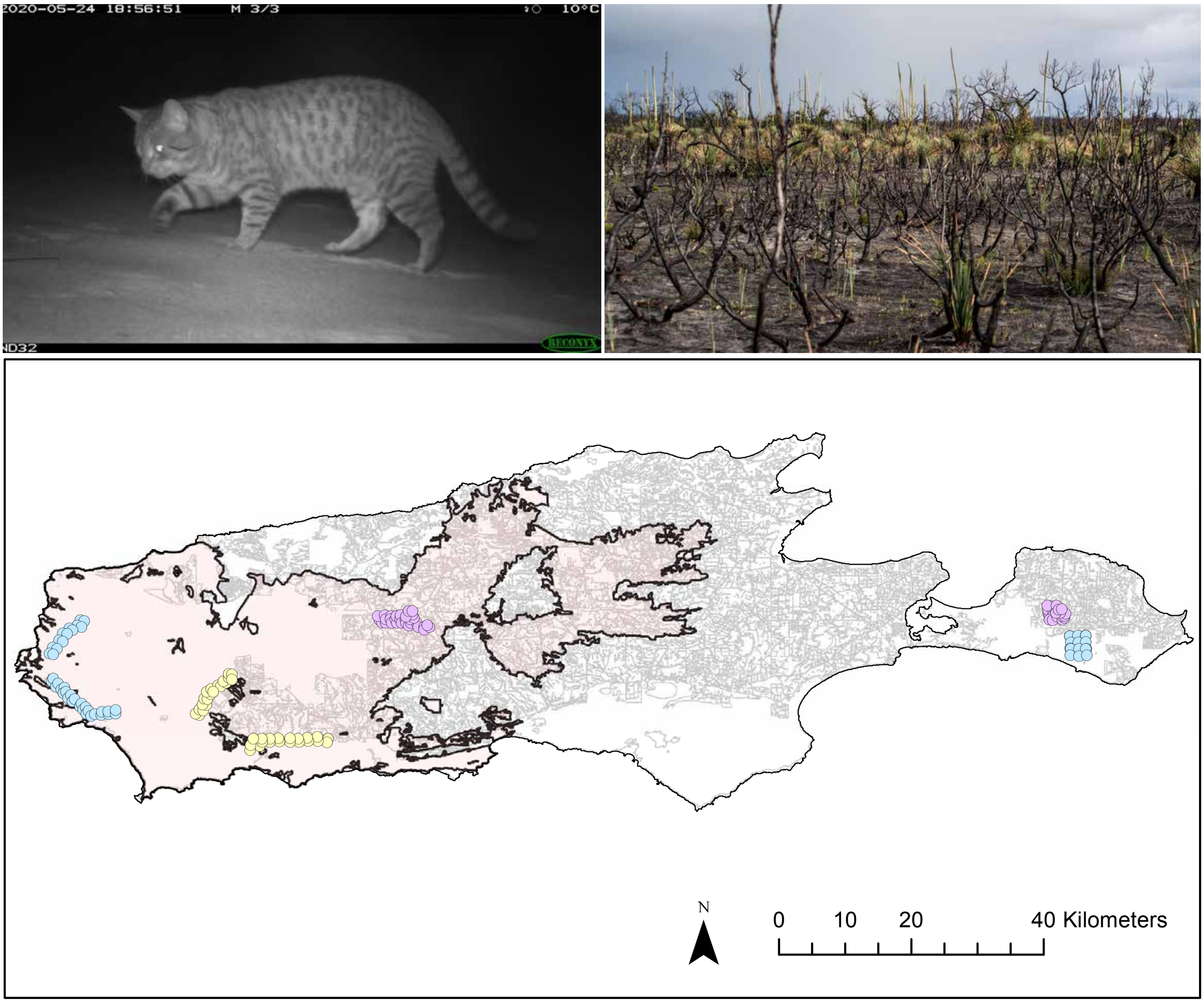

Figure 1. Arrays of camera-traps were used to estimate feral cat density before and repeatedly after a large, high-severity fire on Kangaroo Island, South Australia. The arrays spanned forest (blue), farmland (purple) and forest-farmland boundaries (yellow); burnt areas are shaded pink. Cat control also occurred at some arrays post-fire. Cats were individually identified where possible, and densities estimated using spatial mark-resight models. For more information, see Hohnen et al. 2020; Hohnen et al. 2021. Images credit Rosemary Hohnen.

Foxes

In general, foxes cannot be reliably identified from camera photos (Güthlin et al. 2014; although see Dorning 2019), unless augmented with GPS location data — see case study 8.1. However, individuals can be identified by genotyping scat or hair samples collected non-invasively (see Sections 3.5 and 3.6 for method details, respectively). This approach was used successfully to estimate fox density from hair samples by Berry et al. (2012); Berry et al. (2014); Marlow et al. (2015b); and from scat samples by Marks et al. (2009) and Le Pla et al. (in review); see also Case Study 8.2. Marlow et al. (2015a) successfully used both fox hair and fox scat samples, as well as saliva and tissues collected incidentally, to estimate fox density while minimising any potential sampling bias from variation in individual fox behaviour within the population.

6.2 Survey design

Estimating predator density through spatial capture-recapture methods involves deploying detectors such as cameras, sticky wickets (for hair collection) or traps at survey sites, or actively searching plots or linear transects for fresh scat samples that can be attributed to individuals through genotyping. During data analysis, plots or transects are commonly discretized into raster cells or transect segments, respectively, making them equivalent to detectors at sites, although models for polygon and linear transect datasets are also available (Efford 2011; Efford 2021).

The power and bias of alternative study designs for density estimation can be explored using the R package secrdesign (Efford 2020) or the secr design app (https://www.stats.otago.ac.nz/secrdesignapp/). Below we provide an overview of some key considerations.

Survey layout. The survey sites, plots or transects need to be spaced relative to the movements of individuals. For a given number of sites or search effort, the optimal design will involve a tradeoff between surveying across large area to detect many individuals and having closely-spaced survey sites to facilitate the repeat detections of each individual at several locations (known as *spatial recaptures*).

To obtain unbiased estimates, it is crucial that:

1. The area and maximum diameter of the total survey area are larger than the average home range size of animals (Efford 2011). Fox and cat home range sizes vary widely across Australia, so this should be based on location-appropriate information.
2. The survey sites or transect segments are close enough that more than five spatial recaptures (movements of individuals) are detected (Efford & Boulanger 2019).
3. If the predators have elliptical ranges (rather than perfect circles), the configuration of the survey sites or transects is not systematically aligned with their ranges (Efford 2019).

Designs that do not meet these criteria are likely to give biased density estimates and should not be used (Efford 2019; Efford & Boulanger 2019). Within these criteria, decisions about the number of survey sites (or search area or transect length), their spacing and configuration, the number of repeat surveys within each primary survey session, and replication of survey sessions through time will influence the number of new animals detected ('captures') and the number of repeat detections ('recaptures'), and hence the precision of the density estimate(s).

It can become impossible to estimate density when the number of individuals detected or recaptured is very low. Furthermore, to obtain density estimates that are precise enough to be useful (e.g., to compare densities between locations or through time), it is preferable that the relative standard error (RSE) of the density estimate is less than 20%, or ideally less than 10% (Efford & Boulanger 2019). The RSE for a given study design can often be approximated as 1 / sqrt(estimated number of captures) or 1 / sqrt(estimated number of recaptures), whichever is the smaller; see Efford and Boulanger (2019) for details. For an RSE of 20%, this is equivalent to at least 25 captures and 25 recaptures; an RSE of 10% would require at least 100 captures and 100 recaptures.

Density estimates will also be most precise when the estimated number of captures and estimated number of recaptures are equal, i.e. when the 'average' animal is detected twice (Efford & Boulanger 2019). For a given number of survey sites, the number of captures will increase as the sites are spaced further apart (because the total area surveyed is larger), but the number of recaptures will decrease (because there will be fewer survey sites within each animal's home range). Sigma (σ) is a movement parameter used in spatially-explicit models to describe how the distance from the centre of an animal's home range affects the likelihood of detecting it (σ is related to the home range radius but not directly equivalent). The optimal spacing of survey sites is usually in the range of 1 – 3 σ for a half-normal detection function. If sampling intensity is low (due to low detectability, few repeat surveys or a linear configuration of survey sites), the optimal spacing may be less than 1 σ (Efford & Boulanger 2019); however covering a large total survey area remains important (Nawaz et al. 2021). Estimates of σ are provided in most published studies that have used these approaches to estimate fox and cat density, albeit not always from the half-normal detection function; see Section 6.1 for references.

The survey sites or search areas do not have to be spread evenly across the study landscape. Clustered, irregular or even linear arrays can be used, as long as they meet the above criteria. If clusters are used, each cluster should be large enough to span a home range; there is a trade-off between the number of clusters and the number of survey sites within each cluster (Sun et al. 2014; Efford & Boulanger 2019).

The survey sites or clusters should provide a rigorous sample of the study landscape, e.g. be placed at random locations (Borchers & Efford 2008). Given that foxes and cats often select for and move along roads and waterways (e.g. Moseby et al. 2009; Hradsky et al. 2017b; Wysong et al. 2020a), the implications of only conducting surveys along these linear features requires further exploration, due to the potential to bias σ and density estimates (Sutherland et al. 2015; Nawaz et al. 2021). An alternative is to explicitly model density only in these linear features (e.g. Kreplins et al. 2020).

Survey duration. A single survey can be used to estimate density through spatial capture-recapture methods, if it provides enough recaptures of the same animal at different locations (e.g. scats in multiple grid cells or hair samples across a large area) (Efford et al. 2009b). However, this is very unlikely to be sufficient for cryptic, wide-ranging species such as foxes and cats: even with many repeat surveys within a survey session, studies often struggle to obtain sufficient sample sizes (Anile et al. 2014; Hohnen et al. 2021).

Repeating the survey several times within a session will increase the number of captures and recaptures. However, most spatial capture-recapture models require an assumption of population closure; that is, that there are no births, deaths, immigrations or emigrations during the survey session. A tradeoff therefore arises between conducting more repeat surveys and trying to meet this assumption by only surveying for a short period. Simulations by Dupont et al. (2019) suggest that, for species with an intermediate life history such as the fox (and presumably the similar feral cat), increasing the duration of the survey session improves the precision and relative bias of spatial density estimates up to approximately 0.4 years, after which relative bias increases. Survey sessions should also be timed to avoid periods of peak population change (Dupont et al. 2019); for seasonal species such as the fox, these are likely to be the autumn dispersal season and late spring when offspring are recruited into the detectable population.

Open-population spatial capture-recapture models have been developed (e.g. Gardner et al. 2010), but have not yet been extended to partially-marked populations.

6.3 Data analysis

Spatial capture-recapture models are hierarchical. They consist of two submodels: a state model describing the distribution of animal activity centres within home ranges, and an observation model that links the probability of detection to home ranges (Efford et al. 2009a; Royle et al. 2013). Detections and non-detections of uniquely marked individuals at known locations are used to estimate detectability and an activity centre for each individual. If unmarked individuals are also included (i.e. in a mark-resight model), they are typically assumed to be a random sample of the population, sharing the same detection process as marked individuals (Efford & Hunter 2018; Dey et al. 2019).

Capture-recapture models estimate population density by summing the number of activity centres across the surveyed area and surrounding buffer zone (the unsurveyed space around each survey site for which a detected individual could feasibly have an activity centre). A starting value for this buffer width can be based off ecological knowledge of predator movement, or estimated ad hoc as at least three times the root-pooled spatial variance (RPSV), then evaluated by calculating the effective sampling area as a function of increasing buffer size (Borchers & Efford 2008; Efford 2021).

Maximum likelihood models are an accessible way of modelling spatial capture-recapture and mark-resight data; these can be fitted in the 'secr' R package (Efford 2021) or the Shiny application 'secr app' – see https://www.otago.ac.nz/density/ for more information. Spatial capture recapture models are also commonly fitted within Bayesian software such as WinBUGS (Royle & Young 2008) or the 'nimble' R package (Turek et al. 2021). Bayesian methods with data augmentation offer considerable advantages for handling unmarked and unidentifiable detections (Jiménez et al. 2021).

7. Monitoring predator diet

Understanding which species and how many individuals of each prey type are being eaten by introduced predators is often important. Numerous diet reviews have recently been conducted for Australian foxes (Fleming et al. 2021; Stobo-Wilson et al. 2021) and cats (Doherty et al. 2015; Woinarski et al. 2017; Woinarski et al. 2018; Murphy et al. 2019; Woinarski et al. 2020; Woolley et al. 2020), based on scat and/or stomach contents analysis.

7.1 Scats

The predator species that deposited the scat can be determined based on scat location, shape and smell, or DNA analysis (see Section 3.5.1). However, differences in diet between predator individuals or subcategories (e.g. sex, age) can't be identified from scats, unless the scats are collected from trapped animals or sex and/or individual identity is determined using DNA (Section 3.5.2).

Biases in diet based on scat contents may occur due to differences in the digestibility of different prey types (e.g. beetles are less digestible than mammals). It can also be difficult to distinguish prey killed by the predator from scavenged roadkill etc., although the presence of maggots can be a helpful indication.

Scat content data may be reported as the proportion of scats in which the prey occurs, the proportion volume or mass of scat contents comprising the prey, the number of prey per total number of food items, or the proportion of prey per total biomass consumed. If feeding trials have been conducted, biomass calculations can be done to convert the weight of prey in the scat to the weight of original prey consumed; these are considered to provide the best approximation of a carnivore's true diet (Klare et al. 2011).

7.2 Stomach contents

Using the stomach contents of dead predators to analyse diet allows you to easily observe the characteristics of the individual predator (sex, age, reproductive status etc.) and so relate it related to the individual's diet, unlike for scat analysis (Catling 1988; Kutt 2012). Stomach contents are a function of time (for example, stomach contents are likely representative of ~10 hours in cats; Chandler et al. 1997), and so they can also be used to make estimates of kill rates. For example, Moseby et al. (2021) used stomach contents to estimate annual kill rates of rabbits for different age classes of feral cat. Some characteristics of the prey can also be evaluated with this approach, such as whether they were juveniles or adults. However, differences in digestibility mean that some prey, such as soft-bodied frogs, are less identifiable in stomach samples than others such as birds.

7.3 Isotope analysis

Tissue or hair samples from predators can be analysed for the ratio of heavy to light isotopes (most commonly carbon and nitrogen), relative to standard. The isotope ratios for different prey types also need to be determined. In contrast to scat and stomach samples, isotope analysis provides information on the predator's average diet in the past days to months (depending on the tissue source), rather than immediate consumption, and is less biased by differences in digestibility. Isotope analysis is particularly useful for identifying variation in diet between geographic regions, seasons or individuals, and quantifying the consumption of easily-digested foods such human foods. However, interpretation can be made more challenging by fractionation (differences in the isotopic composition in the predator's tissues and its diet) and differences in the isotype signatures of different prey sources. Isotope analysis has been successfully conducted for foxes and cats overseas (e.g. Meckstroth et al. 2007; Killengreen et al. 2011), and is now being trialed in Australia.

7.4 Predator-mounted video camera

This approach involves fitting predators with a GPS collar and a video camera to capture footage of animal behaviour. The video camera is typically placed on a collar, so that it sits under the chin, and may cycle through recording/non-recording periods. Animal-borne video has recently been to record domestic, stray or feral cats hunting a variety of prey (Loyd et al. 2013; McGregor et al. 2015a; Hernandez et al. 2018; McGregor et al. 2020). This method can provide the most accurate data available in wild settings on predators' hunting techniques, the micro-habitats around hunting events, and consumption rates. Video cameras can also capture data on other aspects of animal behaviour, such as interactions between adults and kittens. The major drawback is that animal-borne video can only record a small amount of footage per deployment; currently less than 12 hours for cats and foxes. This results in small sample sizes, typically at great expense of time and money. It also means that a reliable method for capture and recapture of animals is needed to achieve valid sample sizes. Other limitations include difficulties in identifying the prey item and classifying the predator's behaviour from the video, and the potential to miss prey items if the handling time is less than the non-recording interval (Brockman et al. 2017).

8. Case studies

8.1 Scotia Wildlife Sanctuary

Written by David Roshier

8.1.1 Context

Australian Wildlife Conservancy commenced a 4-year study at Scotia Wildlife Sanctuary in far western NSW in 2015, with the aim of:

- Comparing 'best-practice' statistical methods for density estimation with methods that are less demanding in terms of data requirements and analysis, such as activity indices derived from camera surveys and track surveys.
- Understanding more about the spatial ecology and interactions of feral cats and foxes in this environment.
- Informing the reintroduction of regionally extinct mammals into predator-controlled landscapes.

8.1.2 Methods

The study landscape is semi-arid woodland, with low and highly variable rainfall. Lethal control of foxes across the study landscape commenced after two years as part of a related project.

We monitored fox and feral cat populations across an area of 14,000 ha, using 60 sand plots (Section 3.2) and up to 107 individual camera traps with passive-infrared sensors (HC600, Reconyx; Section 3.4). Surveys were conducted in monthly sessions, with 43 sand plot sessions and 48 camera-trapping sessions from October 2015 until September 2019. Lethal control of foxes commenced from Session 25 and not all methods were available subsequently because marked individuals were killed and removed from the population.

Sand plots were spaced approximately 500 m apart on the existing road network and monitored daily for a four-day period each month, coinciding with the camera-trapping sessions. Cameras were deployed for 24 days per month at 98 sites, spaced 750-2000 m apart. Seventy-two cameras were placed on roads for the duration of the study in various configurations (grids or transects), and the remainder placed off road (Carter et al. 2019).

We trapped cats and foxes and fitted them with GPS collars between 2015 and 2018 (Sections 3.8 and 3.9). In the first two years, all GPS collars had VHF signal and UHF remote-download functions; fox collars also had a remote drop-off mechanism. In 2018, cat collars instead transmitted data via the Iridium satellite network. Location data were collected at 20 min intervals overnight and 96 min intervals during the day. See Roshier and Carter (2021) and (Roshier et al. 2021) for details.

Data from the sand-plots and/or camera surveys were analysed using three different approaches, as well as comparison of methods within each approach:

- Count-based and detection-rate based indices of activity (Section 5.2). The **Allen–Engeman Index:** the average number of tracks per sand plot, per day (or photographs per camera, per day) (Engeman 2005), and the **Catling Index**: the percentage of sand-plot nights with tracks (or camera-trap nights with photos) (following Catling and Burt 1994).
- Abundance estimates for unmarked populations, using **N-mixture models** (Royle 2004) (Section 5.4.3).
- Density estimates for partially-marked populations, using **spatial mark-resight and maximum likelihood estimation methods** (Borchers and Efford, 2008; Royle et al. 2014) (Section 6.3). In addition, camera trap data were augmented with telemetry data and analysed within a **Bayesian framework**, following the approach of Sollmann et al. (2013) and Whittington et al. (2018). Individual cats could not be reliably identified from camera images due to their distance and orientation. Densities could only be estimated for sessions when GPS collars were deployed (Carter et al. 2019).

8.1.3 Results

Fox population estimates.

Twenty-six foxes were trapped and fitted with GPS collars. Foxes were detected on camera traps in every survey session across the 4-year study except one session post-control (e.g. Figure 2). In the first two years prior to the commencement of lethal control, there were 2,781 total detection events on survey cameras. Yet on average, 26% of collared foxes were not detected in any given month during this period, even though GPS data indicated they were resident within the study area. In the same period, there were 795 detections on sand plots.

Figure 2. *A fox with a GPS collar walks past a survey camera at Scotia Wildlife Sanctuary. Image credit Andrew Carter, copyright Australian Wildlife Conservancy.*

The Allen–Engeman and Catling indices of fox populations were near-identical for each session when applied to camera or sand plot data. However, indices derived from sand plots were generally higher and more variable than those from camera traps.

The N-mixture model had much wider confidence intervals (i.e. greater uncertainty) than the Allen–Engeman index, likely reflecting the lack of spatial independence of camera survey sites.

Fox detection rates were much higher at cameras placed on roads compared with off-road cameras. Yet, in most sessions, camera placement had relatively little effect on fox density point estimates from spatial mark-resight methods. However, more detections of unmarked foxes, or fewer detections or resightings of marked foxes increased uncertainty around the estimates. Generally, for spatial mark-resight methods, density estimates derived from grids had narrower confidence limits (or less uncertainty) than those derived from transects. See Carter et al. (2019) for details.

Feral cat population estimates

Twenty-eight feral cats were trapped and fitted with GPS collars. Cats were detected in all sessions, even as detections approached zero in the latter part of the study (Figure 3).

The comparison of methods was more problematic for feral cats, as overall detections and resightings of marked animals were fewer, despite deploying more GPS collars. A change of technology to satellite-based GPS collars revealed that individual cats were traveling up to 160 km from their point of release, which explains why many were not seen again in the camera surveys (Roshier & Carter 2021).

Space use

There were large differences in the distribution and movement ecology of foxes and feral cats. Foxes mostly occupied defined territories, while many feral cats roamed apparently independent of each other. Among individuals that were range-resident, the home ranges of feral cats were 3–3.5 times larger than those of foxes (Roshier and Carter 2021). A portion of the fox population used extraterritorial artificial water sources (Roshier, Signer and Carter 2021). No such dependency on artificial water sources was evident in cats.

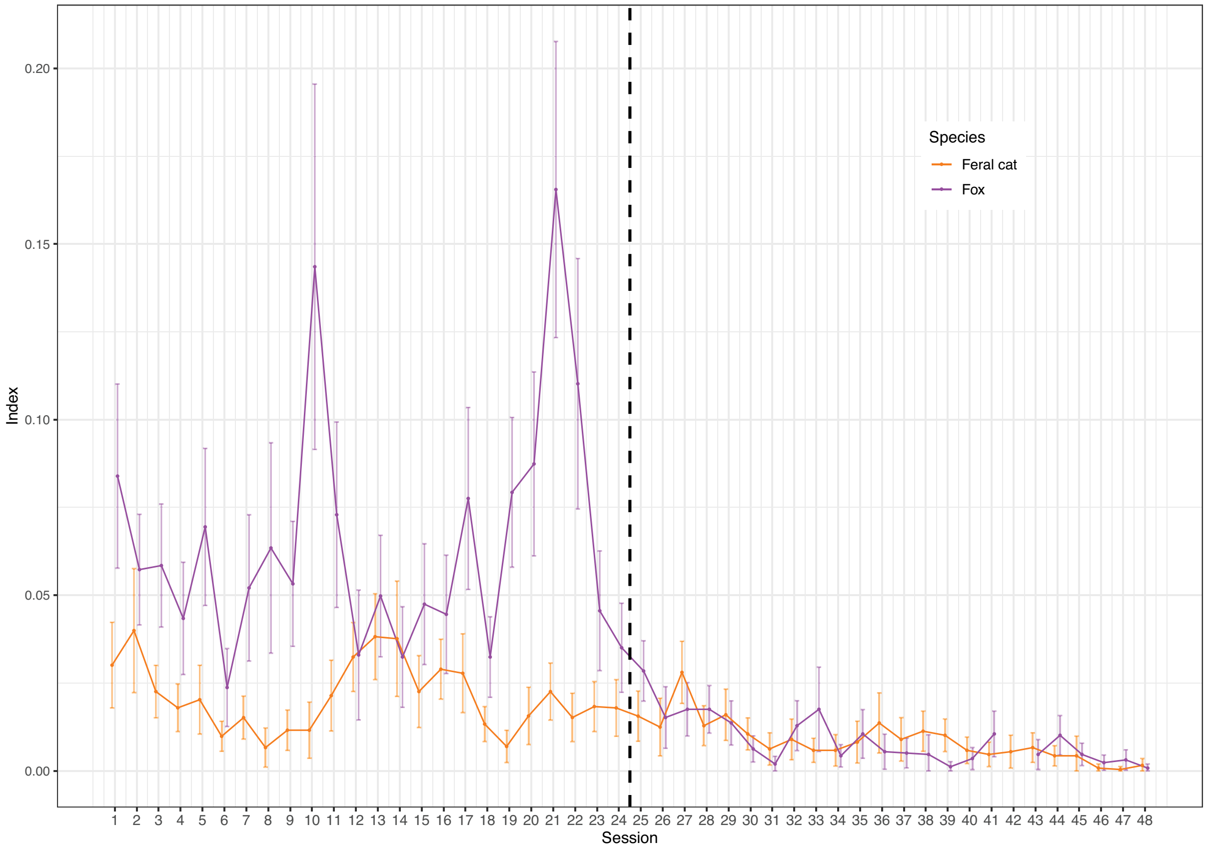

Figure 3. *Allen-Engemen indices of activity with confidence limits for red fox and feral cats derived from camera trap data at Scotia. Note: lethal control of foxes commenced in session 25 (after Roshier 2020)*

8.1.4 Reflections

This study showed that the precision of population density estimates using spatially explicit mark–resight models is sensitive to resighting rates of identifiable individuals. Camera surveys with and without telemetry data from GPS collars can provide reliable estimates of population density using such models. However, ensuring frequent resightings of uniquely-identifiable individuals on cameras outweighs concerns about the magnitude of unknown biases associated with placing cameras on roads, tracks or trails (Carter, Potts and Roshier 2019).

Indices of activity such as the Allen–Engeman index provide a useful proxy for density when resources, time and/or expertise are more limited.

Observed differences in movement behaviour of feral cats and foxes from the GPS collar data underscores the importance of incorporating species and location-specific information on spatial behaviour when planning camera surveys. To achieve the same level of precision, the extent, intensity or duration of monitoring will differ for each population. Methods also differ in whether they assume that survey sites are spatially independent (e.g., N-mixture models; Royle 2004) or require redetections of individuals between survey sites (e.g., spatial capture-recapture and mark-resight methods; Borchers and Efford, 2008). As a result, the precision of all abundance estimation methods will be influenced by the grain of the spatial array deployed to detect individuals in the population (Roshier and Carter 2021).

Knowledge of resource use and movements derived from GPS tracking is also useful for informing lethal control programs. For example, this study showed that there are opportunities to target foxes around water sources during periods of drought or heat accumulation (Roshier, Signer and Carter 2021). It also showed that a greater portion of the feral cats in this study were non-sedentary and, therefore, likely less amenable to local control efforts than foxes. Individual cats familiar with the study area left for extended periods and returned up to 11 months later.

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8.2 Otway Ranges

Written by Matthew Rees, Mark Le Pla and Bronwyn Hradsky

8.2.1 Context

In 2017, Parks Victoria began a broadscale fox control program to help conserve threatened native fauna in the Otway Ranges on Gadubanud and Wadawurrung country, south-west Victoria. The program involves continuous 1080 poison-baiting at 1-km intervals along accessible roads and tracks.

As part of an associated monitoring program, the Conservation Ecology Centre and University of Melbourne led studies to quantify the effects of the fox control program on fox density and feral cat density, respectively. The aim was to (1) quantify the reduction in fox density caused by the fox control program and (2) investigate any potential increase in feral cat density (mesopredator release hypothesis).

8.2.2 Methods

Both studies had a Before After Control Impact design (Section 4.1), with one treatment landscape and one non-treatment landscape monitored concurrently before and for two years after baiting commenced.

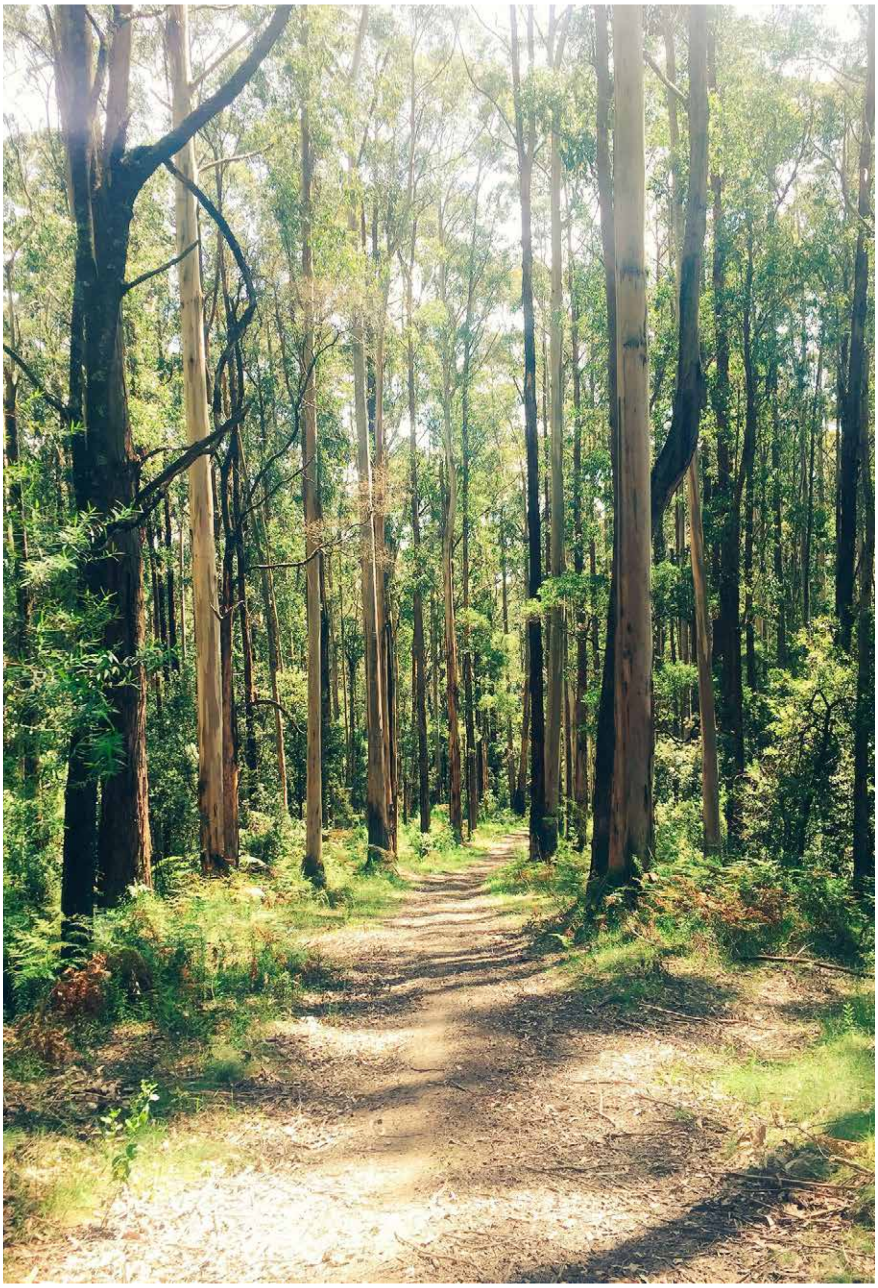
Our study landscapes were located in the rainforests and wet forests of the western Otway Ranges. Foxes and feral cats are the dominant terrestrial mammalian predators here, allowing us to focus on the interactions between these species.

Fox density surveys

In each landscape, we established a 16-km continuous scat survey transect along forest tracks (Figure 4; Sections 3.5 and 6.1). The treatment and non-treatment transects were approximately 5-km apart. We collected fox scats simultaneously along both transects over eight sessions: two prior to baiting and six after baiting had begun in the treatment landscape. Each session involved five surveys of each transect by multiple observers, over a 10-day period. All fox scats were removed from the transects prior to the first survey to ensure that all scats collected were less than 48 hours old. In some sessions, trained scat-detection dogs were also used to help find scats.

Fox scats were frozen immediately after collection. They were then sent to experts at CESAR Australia for swabbing, screening for fox DNA and genotyping using microsatellites to identify individuals (Section 3.5.2).

Once data were returned, we used spatial capture-recapture models to estimate fox density and test the effects of fox control (Section 6).



RIGHT: ***Figure 4.*** *We surveyed for fox scats along forest roads and tracks, Otway Ranges, Victoria. Image credit: Mark Le Pla.*

Feral cat density surveys

In each landscape, we established a camera-trapping grid of 63-103 cameras; cameras within each grid were spaced approximately 500 m apart, based on our knowledge of cat ranging behaviour for this region (Sections 3.4 and 6). The total area of each grid was approximately 49 km^2, and the grids entirely overlapped the fox scat survey transects. Cameras were placed off-road, with lures of tuna-oil and white feathers (Figure 5). Treatment and non-treatment landscapes were surveyed at the same time each year, once prior to fox control and twice afterwards. Cameras were deployed for two to three months in each survey session.

We identified individual feral cats from the camera trap images, based on their coat markings. Cats were initially sorted into five categories: black, mackerel tabby, classic tabby, ginger and other (cats with multiple colour blends or other distinctive coats). We did not attempt to identify black cats, treating them as "unmarked" in the spatial mark-resight models. Within the other four coat categories, multiple observers identified individual cats based on their unique coat patterns where possible.

We used spatial mark-resight models to estimate feral cat density (Section 6), and test the effects of the fox control and spatial estimates of fox occupancy (derived from the camera surveys).

Figure 5. *A "marked" tabby cat investigates a tuna oil lure, Otway Ranges, Victoria. Image credit: Matthew Rees*

8.2.3 Results

Fox density surveys

Through microsatellite genotyping, we were able to attribute an individual fox identity to 55% of the 614 scats collected across the 60 surveys. In total, 50 individuals were identified: 29 on the treatment transect and 21 on the non-treatment transect. In general, more female foxes than males were detected in each survey session. These data enabled us to produce the first direct estimates of fox density in the wet forests of south-eastern Australia.

Changes in the raw fox activity index (scats per km) suggested the lethal control program was effective at reducing fox activity. However, this was not reflected in density estimates or individual survival: there was no evidence of a significant reduction in fox densities post-baiting in the treatment area and many individual foxes survived throughout the monitoring period. No individuals were detected moving between treatment and non-treatment landscapes.

Feral cat density surveys

Over the three years of surveys, we recorded 970 feral cat detections from 36,272 camera-trap nights. Feral cats were detected in 60% of camera deployments; 40% of these detections were of cats with black coats. We were able to identify 53% of cat detections to the individual level; a total of 93 cats (20–30 individuals per landscape). No individuals were detected moving between treatment and non-treatment landscapes.

Feral cat densities were much higher than expected (Rees et al. 2019). We found evidence that fox control affected feral cat density and behaviour (Rees et al. in review). Over the three year survey period, cat density increased in the treatment landscape and decreased in the non-treatment landscape. At a finer scale, feral cat density and detectability were highest where spatial fox occupancy estimates were lowest (Rees et al. in review).

8.2.4 Reflections

Overall we were pleased with the effectiveness of our survey approaches: we obtained enough data to generate density estimates for foxes and feral cats, monitor fox survival during a lethal control program and provide new information on the interactions between foxes and feral cats. Our work provides a useful baseline to compare future changes in introduced predator density, particularly once fox control has occurred over a longer time frame, and complements the broad-scale surveys of predator and prey occupancy led by Parks Victoria.

Our studies generated the first estimates for fox and feral cat densities in Australian wet forests. Contrary to expectations for this habitat type, we found that fox densities were low and feral cat densities were high even before fox control began. It is not clear why fox density was low, given that prey were abundant. However, this may help explain why feral cat density was high, particularly as there was a negative correlation between fox occupancy and feral cat density at a fine spatial scale.

Scat genotyping was a useful and non-invasive way of obtaining mark-recapture data on foxes, even in wet environments where scats might be expected to degrade quickly. Data on fox survival and population turnover during baiting programs is especially likely to be useful when reviewing the design of management programs. Outsourcing the genotyping to experts makes this an accessible approach for management agencies and field survey teams. There is potential for detection dogs to dramatically improve the number of fox scats found per survey, but these benefits need to be balanced against the higher survey (and potentially sampling processing) costs and the practical limitations around using dogs to survey long distances in remote locations. The impacts of dog-presence on fox behaviour also need further investigation.

By deploying camera-traps for two to three months across landscapes with high cat densities, we obtained one of the largest sample sizes used to estimate feral cat density in Australia. The survey duration was also critical to ensure sufficient recaptures at different sites. However, individual identification was tedious and time-consuming. For nocturnal images, identification from white flash camera-traps was considerably easier from infrared. Camera-traps also allowed us to easily contrast the activity and behavior of foxes and cats, including spatial and temporal overlap. This was particularly beneficial for cats because we could contrast responses to foxes in terms of activity, occupancy and density, all from the same data. For example, feral cats had high spatial overlap with foxes, but occurred at lower densities and ranged further when fox occurrence was high. This highlights the benefits of estimating density directly and accounting for changes in detectability when assessing predator interactions.

Indices and absolute density estimates provided conflicting results about population change. We found that fox control affected the number of fox scats (fit with generalised linear models) but there was little evidence that it affected fox density (fit with spatial capture-recapture models using the same scats). The scat count data may have overestimated the effects of baiting, if susceptibility to baiting varied among individuals and/or baiting affected fox behaviour. However, low fox densities prior to baiting also led to relatively small numbers of captures and recaptures of individual foxes and consequently wide confidence intervals around density estimates, limiting our statistical power to detect population change.

A major benefit of the spatial capture-recapture approach is that robust comparisons can be made even when survey designs are modified. Future fox density estimates could be improved by prioritising different sections of road to search for fox scats (genotyping success was very low in wet sections of the transect near waterways) and increasing the number of surveys per session to increase the number of recaptures. Similarly, we had fewer recaptures of feral cats than expected in our first survey session, and so we added additional camera sites between established sites, which improved the precision of the density estimates.

Open population models, which share information on individual activity centres across sessions, are also a priority for future research to improve the precision of density estimates.

8.2.5 Further reading

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8.3 Arid Recovery

Written by Hugh McGregor

8.3.1 Context

The impacts of cats and foxes on native animals can be amplified when rabbits are abundant, because this readily-available food source boosts cat and fox density, with flow-on impacts for prey. This means that landscape-scale rabbit control (such as the release of bio-control agents such as calicivirus) which reduces rabbit numbers and thus cat and fox numbers, can have long-term benefits for native wildlife. However, sudden drops in rabbit numbers might also cause severe prey-switching, particularly in the short-term, if now-hungry cats and foxes switch to killing more native wildlife.

Prey switching by feral predators has been suspected before, but the available evidence has been difficult to interpret. For example, after the original calicivirus markedly reduced rabbit populations around 1995, the incidence of native wildlife in cat scats increased. However, this pattern could have arisen either because cats prey switched, or because they killed the same numbers of native animals but fewer rabbits (thus increasing the relative proportion of everything else).

Experimental evidence of prey switching at landscape scale is extremely difficult to achieve. First, in open landscapes, immigration and emigration can obscure changes in the original population. Second, experimentally altering animal populations at large scale is expensive. Third, catching and tracking free-ranging predators in natural landscapes can be very challenging. Fortunately, all these criteria for an experiment on cats and rabbits were addressed at our project site.

Arid Recovery Reserve in central South Australia is Australia's largest feral predator-proof fenced reserve. This massive reserve (123 km^2 total) has a fox, cat and rabbit proof fence surrounding it, and is also divided into six 'paddocks'. Four paddocks are feral-free with populations of several species of endangered mammal. Two paddocks are dedicated to landscape-scale ecological research; in these two paddocks, feral animals are present but closely managed. In 2017, one of these paddocks had large populations of rabbits and cats, almost at pre-calicivirus levels, making it the perfect place to investigate prey switching in a quasi-experimental setting.

With all the ducks lined up, and a major investment in a rabbit-shooter planned, we had an opportunity to do a Before After Impact Control study to get the first landscape-scale experimental evidence on whether cats prey-switch. We needed to get our monitoring and measurements of the key ecological factors exactly right to capitalise on this once in a lifetime opportunity. Our aims were to obtain:

- Population trends of all key animal populations: cats, rabbits and native wildlife
- Detailed understanding of cat ecology before and after rabbit control, particularly their diet, kill rate and survival.

8.3.2 Methods

First, we needed to measure the rabbit and cat population closely to understand the mechanics of what happened during and after knockdown. The measure needed to be easily repeatable and not require individual identification, so we chose spotlighting (Section 3.3). We drove a 20-km transect inside and outside the experimental paddock every month, and used distance sampling to estimate density (Section 5.4.2). After a few surveys, we suspected that the method was under-estimating density, at least for rabbits, so we trialled using a thermal camera to improve our estimates: we had someone with a thermal camera sit in the back seat independently looking for animals (Figure 6). On cold nights, when the camera was most effective, the person using this thermal camera would see 30% more rabbits within the same range as the spotlight, suggesting densities acquired by normal spotlights surveys should be multiplied by 30% to be more accurate.

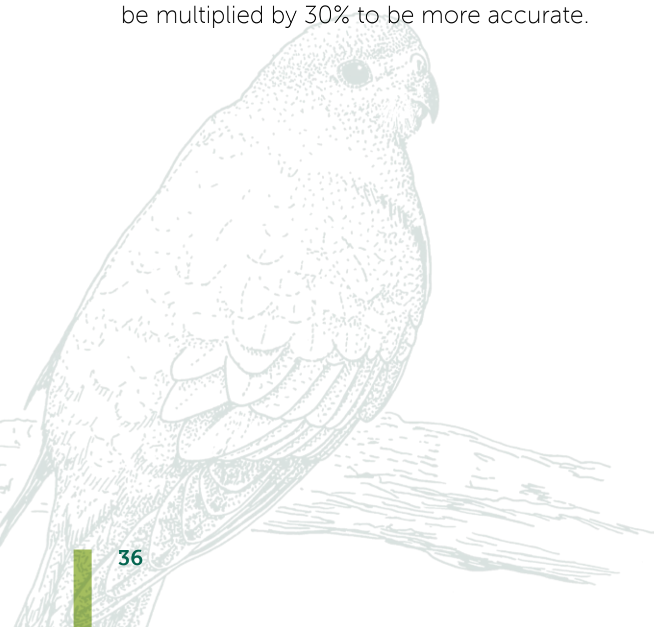

Figure 6. *Feral cat seen through a thermal camera, Arid Recovery. Image credit: Hugh McGregor.*

Second, we needed to measure prey abundance. Spotlight surveys were not good for surveying the native animals of interest in this study, such as spinifex hopping mice and plains mice. So we added a track-counting component to our research (Section 3.2). Fortunately, there is much sand through study area, so this was relatively easy (Figure 7). We made around 70 transects each around 200 m long, wiped them in the afternoon, and came back the next morning to count all tracks. This gave an index of small mammal populations, along with an extra line of evidence for understanding the cat and rabbit activity, as their tracks could also be counted.

Figure 7. *The sandy soils and open vegetation at Arid Recovery mean this is ideal country for conducting track and spotlighting surveys. Images credit: Hugh McGregor*

Last, to understand how cats responded to a reduction in rabbit numbers, we followed the lives of individual feral cats before and after rabbit knockdown. We caught 36 cats and fitted them with wildlife tracking collars. Cats were caught by leg-hold and cage trapping (Section 3.8). All cats were all fitted with VHF collars, and on some we also added either a GPS unit for position data, or a video-collar to measure kill rates (Section 3.9). Six cats were caught outside the experimental fenced paddock, in order to provide a comparison to the 30 cats collared inside the paddock.

8.3.3 Results

Inside the experimental fenced paddock, the professional shooter killed 2215 rabbits within a month. The rabbit activity in our track-count surveys concurrently dropped by 80% in the paddock, but remained unchanged in our surveys outside the paddock. On the other hand, spotlighting surveys before the knockdown suggested the rabbit population inside the paddock was around 2100 rabbits, lower than the number of rabbits killed! Clearly spotlighting was underestimating the size of the rabbit population. Based on our learnings from the thermal cameras, we multiplied this number by 30%, meaning that around 2730 rabbits were present in the paddock before the rabbit knockdown. Satisfyingly, 80% of this estimate is 2184, very close to the body count (2215) from the shooting effort.

After the rabbit knockdown, 40% of the collared cats died within a month. The survivors lost body condition. Cats outside the experimental paddock remained stable in numbers and condition. Cat detections on track-transects and spotlighting inside the experimental paddock also declined by around 40%.

Prey switching was observed from the video collar data, we started seeing cats eat things they were not touching before rabbit reduction. For example, cats started eating carrion, more lizards and caterpillars! Small mammal tracks declined in the experimental paddock, however they also declined outside the paddock to a lesser degree, so this decline could not be confidently attributed to prey switching.

Unfortunately, we could not learn much from the six cats collared outside the paddock, who were much harder to follow than the 30 inside the paddock. Two cats were shot soon after release during routine management activities by land owners, and the other four disappeared. This prevented us from comparing survival and movement of cats inside and outside the experimental paddock. But amazingly, two years after the research was completed Ceduna City Council captured one of the four cats that had gone missing, over 350 km away as the crow flies! It had clearly gone on a massive dispersal journey.

8.3.4 Reflections

We were able to design a Before After Impact Control study of prey switching at a landscape-scale, by capitalising on a unique management situation and opportunity. We showed that cats quickly declined after rabbit control, and that prey switching occurred, with cats more likely to eat prey they wouldn't eat when rabbits are plentiful. However, the prey-switching did not cause a clear decline in native small mammal populations, possibly because the event happened over a very short period, and because individual cats were reluctant change hunting preferences. The study shows that prey switching does need to be considered when conducting landscape-scale rabbit control. Controlling rabbits in tandem with feral predator control would be the safest approach. It is also likely that poison baiting for cats would be far more effective than usual after rabbit knockdown, because they are genuinely hungry.

One key lesson was the importance of diversifying monitoring methods. For our study, spotlighting data was much more useful when combined with thermal cameras and track-counts. The agreement between estimates derived using different methods gave us more confidence about the results. For example, it was heartening that cat survival, cat spotlight rate and cat track counts all decreased by about the same magnitude (40%) after rabbit knockdown. The detailed methods we used here were time-consuming, but the fact we were able to have a quasi-experimental setting and put so much effort into catching and collaring a large number of cats within one area really helped maximise ecological understanding.

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Hugh McGregor and his field survey equipment for capturing and tracking feral cats. Image: Nicolas Rakotopare

Further information:

http://www.nespthreatenedspecies.edu.au

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