Mammal Rev. 2007, Volume 37, No. 3, 191–205. *Printed in Singapore*.

Non-target impacts of poison baiting for predator control in Australia

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ABSTRACT

1. Mammalian predators are controlled by poison baiting in many parts of the world, often to alleviate their impacts on agriculture or the environment. Although predator control can have substantial benefits, the poisons used may also be potentially harmful to other wildlife.
2. Impacts on non-target species must be minimized, but can be difficult to predict or quantify. Species and individuals vary in their sensitivity to toxins and their propensity to consume poison baits, while populations vary in their resilience. Wildlife populations can accrue benefits from predator control, which outweigh the occasional deaths of non-target animals. We review recent advances in Australia, providing a framework for assessing non-target effects of poisoning operations and for developing techniques to minimize such effects. We also emphasize that weak or circumstantial evidence of non-target effects can be misleading.
3. Weak evidence that poison baiting presents a potential risk to non-target species comes from measuring the sensitivity of species to the toxin in the laboratory. More convincing evidence may be obtained by quantifying susceptibility in the field. This requires detailed information on the propensity of animals to locate and consume poison baits, as well as the likelihood of mortality if baits are consumed. Still stronger evidence may be obtained if predator baiting causes non-target mortality in the field (with toxin detected by *post-mortem* examination). Conclusive proof of a negative impact on populations of non-target species can be obtained only if any observed non-target mortality is followed by sustained reductions in population density.
4. Such proof is difficult to obtain and the possibility of a population-level impact cannot be reliably confirmed or dismissed without rigorous trials. In the absence of conclusive evidence, wildlife managers should adopt a precautionary approach which seeks to minimize potential risk to non-target individuals, while clarifying population-level effects through continued research.

Keywords: 1080, invasive species, vertebrate pest

Mammal Review (2007), **37**, 191–205
doi: 10.1111/j.1365-2907.2007.00108.x

INTRODUCTION

Mammalian predators such as the red fox *Vulpes vulpes*, wild dog *Canis lupus* ssp. and feral cat *Felis catus* are recognized as important pests in Australia due to their impacts on agri-

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culture and on native species (e.g. Rolls, 1969; Saunders *et al.*, 1995; Dickman, 1996a,b; Glen & Short, 2000; Fleming *et al.*, 2001). The red fox in particular has been implicated in the decline of a wide range of native fauna (e.g. Burbidge & McKenzie, 1989; Short, Kinnear & Robley, 2002) and is also recognized as a significant predator of lambs and goat kids (Saunders *et al.*, 1995). Feral cats have been implicated in many faunal declines (Dickman, 1996a,b). Wild dogs are significant predators of livestock and are therefore considered pests of the grazing industry (Rolls, 1969; Fleming *et al.*, 2001).

Conflict between humans and mammalian predators is not unique to Australia and introduced predators have established themselves as pests over much of the globe. For example, feral cats have had devastating effects on a number of countries, including the extinction of several species of birds in New Zealand (e.g. Atkinson & Bell, 1973; Dickman, 1996a) and the extirpation of iguanas from some of the Galapagos Islands (Phillips *et al.*, 2005). Native predators can also become pests. For example, culpeo foxes *Pseudalopex culpaeus* in Patagonia and coyotes *Canis latrans* in parts of North America are considered agricultural pests due to their depredation of sheep (Sacks, Blejwas & Jaeger, 1999; Travaini, Peck & Zapata, 2001). A recent meta-analysis by Salo *et al.* (2007) has confirmed that introduced predators have generally detrimental impacts on populations of native species. This study also revealed that impacts on prey are much greater in Australia than in other parts of the world, and thus emphasized the need for particularly effective management of predators in the island continent.

Since the early days of European settlement in Australia, control of predators has been attempted using a variety of methods, including shooting, trapping, fencing and poisoning (Rolls, 1969). Similar measures are taken in many parts of the world, e.g. New Zealand (Murphy *et al.*, 1998), South America (Travaini *et al.*, 2001), North America (Sacks *et al.*, 1999) and parts of Europe (Fairley, 1971; Kauhala, 1996; Packer & Birks, 1999). While the precise methods used vary according to local resources and conditions, control programmes everywhere should adopt the best practice principles of reducing the damage caused by pests, while minimizing the risk of harm to non-target species (Braysher, 1993).

The most widely used method of controlling exotic predators in Australia is the distribution of poison baits which contain sodium monofluoroacetate, commonly known as 1080. Baits consist either of fresh or dried meat, offal, chicken eggs or commercially produced mixes (Saunders & McLeod, 2007). Baiting is the only method currently available for predator control that can be used successfully over broad areas (Gentle, 2005). By comparison, other methods are ineffective, expensive, labour-intensive or area-specific (Saunders *et al.*, 1995). Despite the efficacy of poison baiting for predator control, concern has been expressed by a number of authors over the potential effects of poisoning programmes on native, non-target animals (e.g. McIlroy, 1981a,b; Sinclair & Bird, 1984; Calver *et al.*, 1989a,b; Marks, Nijk & Gigliotti, 1996; Belcher, 1998, 2003; NPWS, 2001; Glen & Dickman, 2003a,b). Here, we review the evidence for both positive and negative effects on native Australian mammals of 1080 baiting for predator control. The purpose of our review is to explain the potential risks that 1080 baiting has for non-target species and to describe previous attempts to evaluate or minimize these risks. In doing so, we develop a logical framework which may be used both within and outside Australia to assess and manage the risks involved in using poisons for predator management.

EFFECTS OF 1080 BAITING ON TARGET AND NON-TARGET ANIMALS

Baiting with 1080 has been successful in reducing populations of vertebrate pests in a range of locations. For example, Thomson *et al.* (1998) achieved a reduction of 95% in the density of foxes, and Burrows *et al.* (2003) reported reductions of up to 100% in indices of feral cat abundance, using aerial baiting. Numerous other studies have reported large reductions in estimated abundances of wild dogs (e.g. Thomson, 1986), foxes (e.g. Thompson & Fleming, 1994) and feral cats (e.g. Short *et al.*, 1997) after baiting.

A number of studies indicate that reductions in the density of introduced predators can be highly beneficial to populations of native animals. For example, Kinnear, Onus & Bromilow (1988) reported increases of up to 223% in numbers of the black-footed rock-wallaby *Petrogale lateralis* following fox control (see also Kinnear, Onus & Sumner, 1998). Similarly, Friend (1990) found that the recovery of numbat *Myrmecobius fasciatus* populations was more rapid in areas where foxes were baited than in areas where no baiting occurred. Large increases in the local abundance of several other species of medium-sized native mammals following 1080 baiting of foxes have been reported (e.g. Saunders *et al.*, 1995; Morris *et al.*, 2003).

Baits used in such programmes do not always reach their target. Various studies have recorded the removal of toxic or non-toxic baits by non-target animals including birds (e.g. Allen *et al.*, 1989; Dexter & Meek, 1998), spotted-tailed quolls *Dasyurus maculatus* (e.g. Körtner, Gresser & Harden, 2003; Murray & Poore, 2004; Claridge *et al.*, 2006), native rodents (e.g. McIlroy, 1982b; Fairbridge *et al.*, 2001), bandicoots (Fairbridge *et al.*, 2001) and reptiles (e.g. Short *et al.*, 1997). Similarly, 1080 baits for brushtail possums *Trichosurus vulpecula* in New Zealand have been removed by non-target species including native birds, domestic animals and livestock (Spurr, 2000). Such non-target removal potentially poses a risk of poisoning native fauna and also reduces the effectiveness of the control programme. There is also a potential risk of secondary poisoning if the carcasses of poisoned animals are eaten by scavengers (McIlroy & Gifford, 1991). The effects of 1080 baiting on non-target animals are difficult to quantify as Australian native species show wide variation in their sensitivity to the toxin (McIlroy, 1986) and in their propensity to locate and consume baits (McIlroy, 1981a). Thus, assessment of the potential risk to a particular non-target species (or population) requires knowledge of its sensitivity to 1080 and the likelihood that individuals will find, ingest and receive a lethal dose from poison baits. Other issues such as toxic loading, location of baits and duration of exposure may also need to be considered.

Sensitivity and susceptibility

The sensitivity of an animal to a particular toxin refers to the quantity of toxin required to produce a lethal effect. Sensitivity is usually expressed in terms of the median lethal dose (LD_{50}). The LD_{50} is a statistical estimate of the dose required to kill 50% of a test population (McIlroy, 1981a).

There is wide variation in the sensitivity of Australian species to 1080, with canids and felids being considerably more sensitive than most other taxa (McIlroy, 1981b, 1986). Several explanations have been offered for this disparity. First, fluoroacetate (of which 1080 is the sodium salt) is a naturally occurring compound in a number of Australian plants, including several species of Gastrolobium (McIlroy, King & Oliver, 1985) and at least one species of Acacia (Oliver, King & Mead, 1977). Plants containing high concentrations of fluoroacetate are most common in Australia's south-west, where many native animal species have markedly higher tolerances to 1080 than introduced animals, or native animals from eastern Australia (McIlroy, 1986). This is most likely a result of exposure to fluoroacetate over many generations, which has generated selective pressure for high tolerance to the toxin (King, Oliver & Mead, 1978; Oliver, King & Mead, 1979; Twigg & King, 1991). It has also been suggested that the low basal metabolic rate of a number of species of native Australian vertebrates confers greater tolerance to 1080 relative to many exotic species (McIlroy, 1984, 1994; Twigg, 1994).

Estimates of sensitivity are generally obtained under laboratory conditions, with the toxin being administered by oral gavage rather than being incorporated in a bait. Also, conditions such as ambient temperature, diet and stress levels of animals may differ between the laboratory and the field, so that captive trials may not accurately reflect the sensitivity of wild animals. A potentially more useful measure of risk is the susceptibility of a species to poison baits. Susceptibility is defined by Korn *et al.* (1992) as the likelihood of an animal being fatally poisoned in the field. This measure not only takes into account the animal's sensitivity, but also includes other relevant factors such as the propensity of the animal to locate and consume poison baits, and defensive mechanisms such as bait aversion (e.g. Sinclair & Bird, 1984). For example, an animal that is strictly arboreal and herbivorous, even if it is highly sensitive to 1080, may face little risk from poisoned meat baits laid at ground level. It is therefore possible that a species highly sensitive to 1080 can have low susceptibility to the poison in the field. Thus, the sensitivity of a species to 1080 may be used only as a very rough indication of the potential risk posed by baiting. A more reliable risk assessment requires that other factors be taken into account, most of which can be quantified only by field studies (Beck & Stein, 1979; McIlroy, 1994; Martin *et al.*, 2002).

Non-target species of potential concern

A number of species have been flagged as being potentially at risk from 1080 baiting programmes in Australia. McIlroy (1999) listed three threatened species considered to be of potential concern: the spotted-tailed quoll, eastern quoll *Dasyurus viverrinus* and brush-tailed phascogale *Phascogale tapoatafa*. Of these, the brush-tailed phascogale is mainly arboreal and therefore unlikely to consume ground-placed baits (McIlroy, 1999), while limited evidence from Tasmania suggests no effect of 1080 baiting on the persistence of individual eastern quolls, including females with pouch young (Mooney, Emms & Bloomfield, 2005). However, the spotted-tailed quoll is known to consume toxic and/or non-toxic baits in the field (Murray & Poore, 2004; Körtner & Watson, 2005; Claridge *et al.*, 2006) and smaller individuals may theoretically receive a lethal dose of 1080 from a single 3 mg bait (McIlroy, 1999). Such observations led to concern that baiting may be detrimental to populations of *D. maculatus*. However, more rigorous field trials have since suggested that, for a number of reasons, this theoretical risk is not realized in the field (Körtner, 2007).

Conclusions about the susceptibility of non-target animals based solely on bait removal should be treated with caution, because they do not take into account the fate of bait after it is removed (Körtner, 2007). An excellent example of this discrepancy between uptake and consumption is presented by Körtner *et al.* (2003) where 19 from 20 fox baits removed by spotted-tailed quolls were found uneaten a short distance from the bait station. These findings suggest that it is misleading to infer susceptibility from bait removal alone without supporting evidence. In other cases, animals may survive despite ingesting a theoretically lethal dose, suggesting that the amount of 1080 absorbed in the animal's gut may be less than the nominal dose contained in the bait (McIlroy & King, 1990).

In contrast to the results of Körtner *et al.* (2003), two separate trials have confirmed that spotted-tailed quolls did consume aerially deployed non-toxic meat baits containing the systemic marker Rhodamine B (Murray & Poore, 2004; Claridge *et al.*, 2006). Around 60% of quolls captured following each of these simulated baiting operations tested positive for Rhodamine B, indicating that they had found and consumed non-toxic baits. Subsequent trials have since been conducted using toxic meat baits containing 6 mg of 1080, also deployed from aircraft. In two trials conducted in northern New South Wales, only one quoll from a total of 45 radio-collared individuals was confirmed to have been killed by 1080. A further 18 quolls trapped after baiting (including individuals with and without radio-collars) tested positive for Rhodamine B, indicating that they had consumed bait and survived. Several of these individuals had consumed bait on more than one occasion (DEC, 2005; Körtner & Watson, 2005). A third trial in southern New South Wales recorded no deaths from 1080 among 16 radio-collared quolls, although six of the animals consumed baits (DEC, 2005). In summary, these baiting events had no significant short-term effect on the populations of quolls being studied.

Recent trials in southern Queensland have had similar outcomes. Over a 4-year period, the fate of 72 quolls was monitored during a series of coordinated wild dog baiting campaigns. During this time, quolls were exposed to fresh meat baits containing 6 mg of 1080 for over 700 bait nights. A total of five buried fresh meat baits may have been removed by quolls across all campaigns (less than 0.6% bait uptake), with two quolls confirmed to have died from 1080 poisoning. Regardless, the mortality rate from poisoning was very low compared with other causes of mortality, and the quoll population remained numerically stable over this period (P. Cremasco, Queensland Department of Primary Industries and Fisheries, pers. comm., 2005). Similar trials in Western Australia showed that 43% of western quolls *Dasyurus geoffroii* trapped after aerial deployment of sausage baits containing 4.5 mg of 1080 had consumed bait, but none of 15 radio-collared animals died as a result (Morris, Johnson & York, 2005).

The results of these trials suggest that the potential impacts of 1080 baiting on spotted-tailed quolls have been over-estimated by earlier trials using non-toxic baits. One possible explanation for this disparity is that the presence of 1080 reduces the palatability of baits to quolls as inferred by Körtner & Watson (2005), and as has also been reported for a number of other species including brushtail possums (Morgan, 1982), fat-tailed dunnarts *Sminthopsis crassicaudata* (Sinclair & Bird, 1984) and several species of rodents (Calver *et al.*, 1989a; O'Connor, Moriss & Murphy, 2005). However, this hypothesis does not account for the fact that some quolls in recent trials consumed poison bait on multiple occasions (DEC, 2005; Körtner & Watson, 2005). Alternatively, *D. maculatus* may simply be less sensitive to 1080 than suggested by the limited laboratory data available, or the digestive tract of quolls may be inefficient at extracting 1080 from meat-based baits. Whatever the reason, the potential risk posed to quolls by 1080 baiting programmes is apparently low.

Although individual animals may potentially be at risk, no studies published to date have found significant reductions in populations of non-target animals following 1080 baiting (King, 1989; McIlroy, 1999; Körtner *et al.*, 2003; Körtner & Watson, 2005; Morris *et al.*, 2005). Significant and detectable changes at the population level are of primary concern in the management of wildlife (McIlroy, 1982a; Soulé, 1985; Caughley & Sinclair, 1994).

Individual and population-level effects

Information on the effects of 1080 baiting on individuals may be of limited value as the occasional death of non-targets will not necessarily lead to an overall reduction in a population (McIlroy, 1982a; Hegdal *et al.*, 1986; Sinclair, 1989; Choquenot & Ruscoe, 1999; NPWS, 2001). With less competition among conspecifics for available resources, there may be compensatory demographic responses, such as enhanced survival, enhanced reproduction or immigration (Sinclair, 1989; Choquenot & Ruscoe, 1999). Given the benefits to many non-target species of removing feral predators by baiting (e.g. Kinnear *et al.*, 1988, Kinnear, Sumner & Onus, 2002; Morris *et al.*, 2003), the net effect on their populations is more likely to be positive, despite the loss of some individuals. Therefore, the true effect of baiting on populations of non-target species is best gauged by measuring changes in the density of these species (McIlroy, 1983; Choquenot & Ruscoe, 1999). Using track pads across roads, Körtner & Watson (2005) sampled the activity of foxes, wild dogs and spotted-tailed quolls before and 2–3 weeks after the aerial deployment of meat baits containing 6 mg of 1080. Indices of fox and dog activity dropped significantly after baiting, while indices for the spotted-tailed quoll showed a highly significant increase. Such a rapid response by quolls suggests release from interference competition (Frye, 1983; Cypher & Spencer, 1998; Maitz & Dickman, 2001; Glen & Dickman, 2005). Thus, despite the fact that at least one quoll was apparently killed by the poisoning operation, the overall effect on the population was apparently beneficial. However, the long-term effects remain unknown and the lack of an unbaited control area means that alternative explanations for the change in quoll activity cannot be discounted.

A population may sustain short-term reductions in density but accrue long-term benefits from the control of exotic predators. For example, numerous studies in Western Australia have demonstrated dramatic recovery of threatened native mammals as a result of broad-scale fox baiting (e.g. Friend, 1990; Morris *et al.*, 2003). In such situations, a theoretical risk of occasional non-target mortalities may be readily justified. Thus, it may be possible to set a maximum acceptable level of mortality in the non-target population when planning pest control programmes (Choquenot & Ruscoe, 1999). The magnitude of this limit would depend on the resilience of the population. In turn, the resilience is dependent on the intrinsic rate of increase, and whether the population is regulated by density-dependent or stochastic factors, or a combination of both (Choquenot & Ruscoe, 1999).

Although some level of mortality may be acceptable in populations of non-target species, ethical pest control programmes are seeking increasingly to minimize mortality of non-target animals (Olsen, 1998; NPWS, 2001). Various modifications to baiting methods have been devised in order to reduce the potential risk to non-target species and these are discussed in the following section.

TARGET-SPECIFIC PREDATOR CONTROL

The target-specificity of a baiting programme is dependent on a number of factors. As discussed below, these include the type and amount of toxin used, the type of bait used, the density of baits and the way in which the bait is presented.

Choice of toxin

The use of 1080, as opposed to other toxins such as strychnine or cyanide (Rolls, 1969), offers some degree of target-specificity, particularly in Australian environments. First, 1080 naturally degrades to form harmless by-products and therefore does not accumulate in the environment (Kelly, 1965; Bong *et al.*, 1979; King *et al.*, 1994). Second, most native Australian fauna tend to be less sensitive to 1080 than are introduced foxes, dogs or cats (Oliver *et al.*, 1977; King *et al.*, 1978; McIlroy, 1981b, 1986; Twigg & King, 1991). However, this advantage could theoretically be negated by the smaller body weights of many non-target animals (McIlroy, 1981b). Although 1080 is disproportionately toxic to exotic predators, a potential risk exists for a number of non-target species, assuming they can consume sufficient quantities of bait. However, such risks can be diminished where necessary by using target-specific methods of bait delivery, as discussed in the following section.

Methods of delivering poison

Many broad-scale baiting programmes for wild dogs and foxes, as well as some for feral cats, distribute baits from aircraft (e.g. Thomson, 1986; Thomson *et al.*, 1998; Burrows *et al.*, 2003). Target-specificity of this method may be compromised because the baits are potentially encountered by a wide range of species (Murray & Poore, 2004; Claridge *et al*., 2006). The burial of baits has been found to reduce their uptake by non-target animals (Allen *et al*., 1989; Glen & Dickman, 2003a). However, some non-target individuals have been observed to excavate buried baits (e.g. Fleming, 1996; Belcher, 1998; Dexter & Meek, 1998; Glen & Dickman, 2003a,b) and burial may also reduce the consumption of baits by target animals (Thomson & Kok, 2002).

Little is known about the effects of different burial depths on bait uptake. Field studies have recorded the removal of baits by spotted-tailed quolls from various depths, ranging from 1 to 10 cm (Fleming, 1996; Belcher, 1998; Williams & Marshall, 2000; Glen & Dickman, 2003a,b). In captivity, Belcher (1998) found that spotted-tailed quolls located and excavated non-toxic baits buried at a depth of 10 cm, but not at 20 cm, whereas eastern quolls excavated baits from a depth of up to 20 cm. Burying baits at an increased depth of 15–20 cm may reduce the potential risk to non-target animals; however, the effectiveness against target animals of baits buried at these depths has yet to be tested in the field (NPWS, 2001).

Bait density, or the number of baits laid per unit area, may also affect the potential risk posed by baiting operations to non-target animals. The bait density required to effectively target predators is likely to vary with predator density, home range size and habitat use, and the method of bait presentation (Saunders & McLeod, 2007). Obviously, at the landscape level, bait density must exceed predator density to ensure that sufficient baits are presented for each animal. However, excessive bait density may result in a surfeit of baits being available to each individual, thus wasting resources and perhaps encouraging problems such as caching (Thomson & Algar, 2000) and increased uptake by non-target animals (e.g. Glen & Dickman, 2003a). Caching is a behaviour typical of many carnivores, including foxes and dogs, which involves burying surplus food items for future consumption (Macdonald, 1976; Vander Wall, 1990; Fleming *et al*., 2001). Caching of toxic baits may increase non-target risk as cached baits are difficult for human operators to find and remove, and thus potentially remain available to non-target animals after the completion of a baiting programme (Thomson & Kok, 2002). Such baits may be cached a considerable distance from their original location, potentially in areas where the risk to non-target species is greater (Saunders, Kay & McLeod, 1999; van Polanen Petel, Marks & Morgan, 2001; Gentle, 2005).

In addition to the number of baits laid in an area, consideration must be given to the pattern of bait placement, as this can affect the frequency with which they are encountered by target and non-target animals (McIlroy, 1994). For example, dogs and foxes frequently utilize roads or tracks for movement (Bennett, 1990; Claridge *et al*., 1991; Mahon, Banks & Dickman, 1998; Meek & Saunders, 2000). Placement of baits along such tracks may therefore increase target-specificity of canid control programmes by increasing the chances that dogs and foxes will encounter the baits (May & Norton, 1996).

Placing baits at widely spaced intervals may further reduce non-target impacts by decreasing the likelihood of individual non-target animals encountering more than one bait (e.g. NPWS, 2001; Glen & Dickman, 2003a). This should also reduce the incidence of bait caching through reducing multiple bait uptake by individual predators (Gentle, 2005). However, the home ranges of predators vary in size according to factors such as habitat, resource availability (Bubela, 1995; Banks, 1997) and social status (Meek, 1997). In areas where the home ranges of target animals are relatively small, it may be necessary to place baits at shorter intervals to ensure that a high proportion of the target population is exposed to the baits (NPWS, 2001).

The potential risk to non-target animals may be further reduced by incorporating lower doses of toxin in each bait (NPWS, 2001). According to label instructions, baits for wild dog control contain 6 mg of 1080, while fox baits contain 3 mg of the toxin. These doses are significantly higher than the estimated lethal dose for average-sized dogs and foxes (McIlroy, 1981b; McIlroy & King, 1990). There are several reasons for the use of large doses of 1080 (Korn *et al.*, 1992). Individual animals vary in their sensitivity to 1080 and exceptionally large dogs or foxes may not receive a lethal dose if the amount of toxin used was based on animals of average size. Furthermore, the amount of 1080 contained in baits generally decreases after the bait is deployed (Saunders, McLeod & Kay, 2000). 1080 is lost from baits due to leaching by rain, defluorination by microbes and conversion to inorganic fluoride compounds, and baits may also be partially eaten by insects (Korn & Livanos, 1986; McIlroy *et al.*, 1986; Kramer, Merrell & Burren, 1987; McIlroy, Gifford & Carpenter, 1988; Fleming & Parker, 1991; Parfitt *et al.*, 1994; Saunders *et al.*, 2000; Twigg *et al.*, 2000, 2001; Twigg & Socha, 2001). There is also evidence that 1080 contained in some meat baits may bind to the bait material, preventing its easy absorption in the gut following ingestion (Kramer *et al.*, 1987). Larger doses are employed to compensate for these possibilities.

Another method of delivering toxins to predators which may be more target-specific than conventional baiting is a device known as the M-44 ejector (Busana, Gigliotti & Marks, 1998; Marks, Busana & Gigliotti, 1999). This mechanical device delivers a dose of toxin (in powder form) into the mouth of an animal as it bites the trigger mechanism (Busana *et al.*, 1998). Activation of the ejector requires an upward force equivalent to lifting 1.6–2.7 kg (Connolly & Simmons, 1984), meaning that only relatively large animals are likely to be capable of releasing the trigger (Busana *et al.*, 1998). Because the ejector is anchored in position, the risk of bait caching is also eliminated (Busana *et al.*, 1998). Recent research in Australia has investigated modifications to the design of M-44s to prevent them being triggered by non-target species (Nicholson & Gigliotti, 2005).

Alternative bait materials

A variety of different bait types have been utilized in predator control programmes around the world, including fresh or dried meat, chicken eggs, wildlife carcasses and commercially produced meat-based baits (e.g. Calver *et al.*, 1989a; Saunders & Harris, 2000; Travaini *et al.*, 2001; Twigg, 2001; Twigg *et al.*, 2001; Bencini *et al.*, 2005; Morris *et al.*, 2005). Many of these have sought to increase target specificity either by increasing their attractiveness to the target species or by decreasing their attractiveness or palatability to non-target animals. Increased attractiveness to target animals has been attempted using various chemical or visual lures (e.g. Short *et al.*, 1997; Saunders & Harris, 2000), and increased palatability through presenting favoured bait materials (e.g. Gentle, 2005). Presenting bait that is highly palatable to the target species may not only increase the efficacy of baiting practices but also should reduce the impact on non-target species. Caching is inversely proportional to the palatability of the food (Vander Wall, 1990), so the use of highly palatable bait will mean that fewer baits are cached, remaining available to non-target species after the completion of poisoning operations (Gentle, 2005).

Diminished attractiveness to non-target species can be achieved using dyes or chemical deterrents (Hone & Mulligan, 1982; McIlroy, 1994). Palatability to non-target species may also be reduced by using baits that are too hard or large to be consumed by animals smaller than the target species (e.g. Saunders *et al.*, 1995; Fleming, 1996). A number of studies have investigated the attractiveness of different bait materials (e.g. Soderquist & Serena, 1993; Short *et al.*, 1997; Saunders & Harris, 2000; Martin *et al.*, 2002). Bencini *et al.* (2005) also investigated the possibility of making cat baits more target specific by incorporating the toxin into a capsule. Non-toxic trials suggest that cats are much more likely than non-target animals to consume the capsules (Bencini *et al*., 2005). However, with the exception of Staples & McPhee (1995) and Travaini *et al*. (2001), there has been little direct comparison between different bait types in terms of their uptake by both target and non-target animals in the field.

Free-feeding

Many ground-based predator control programmes have sought to maximize their impact on pest populations and avoid non-target impacts by incorporating an initial ‘free-feed’ period and monitoring the uptake of baits (e.g. Dexter & Meek, 1998; Travaini *et al*., 2001). This method utilizes bait stations, which consist of a buried bait surrounded by an area of smoothed sand to allow the identification of animal tracks. The laying of toxic baits is preceded by a period of free-feeding, during which non-toxic baits are deployed. If the tracks of any non-target animal are identified on a bait station, that station is not used when toxic baits are subsequently laid.

There are arguments for and against the practice of free-feeding. Target animals may become habituated to visiting bait stations, so that the rate of bait uptake increases during the free-feed period. Thus, target animals may be removed more quickly and efficiently when toxic baits are eventually laid (Dexter & Meek, 1998). However, non-target species may also become habituated to bait stations (e.g. Glen & Dickman, 2003b). Another potential disadvantage is that free-feeding appears to increase the incidence of bait caching, thereby increasing the potential risk to non-target species (Saunders *et al*., 1999; NPWS, 2001; Gentle, 2005).

Monitoring non-target impacts

The accuracy of monitoring using tracks on bait stations has also been questioned by a number of authors. This method requires that bait stations are checked daily by observers who are experienced in identifying animal tracks (Belcher, 1998; Fairbridge *et al*., 2001; Glen & Dickman, 2003b). If monitoring is carried out less frequently, tracks will often become obscured by elements such as wind and rain (McIlroy, 1986; Fairbridge *et al*., 2001; Glen & Dickman, 2003b). Another potential problem arises from the fact that dogs and foxes are known to investigate areas of freshly disturbed soil, and will often dig on sand plots, even in the absence of any bait. It is possible therefore that the diggings of a dog or fox may obscure the tracks of a non-target animal that has previously removed a bait. Identification of tracks would lead to the false conclusion that the bait had been removed by a pest animal, causing the incidence of non-target bait uptake to be underestimated (Belcher, 1998; Glen & Dickman, 2003b). These limitations mean that monitoring is impractical in many situations.

CONCLUSION

The aim of any predator control programme should be to ameliorate the economic and environmental impacts of the target species (Braysher, 1993; NPWS, 2001). It is also imperative that control methods are designed to minimize potential impacts on populations of non-target species. However, in doing so, the effectiveness against predator populations must be maintained. Many of the precautions described in this review are unlikely to be effective, practicable or indeed necessary in all areas at all times. For example, fox baiting operations in south-west Western Australia are conducted at a broad spatial scale that precludes techniques such as burying baits. However, the higher tolerance to 1080 of many Western Australian species, as well as the clearly demonstrated biodiversity benefits of these operations, mean that such precautions are unnecessary. In areas where the target-specificity of toxins is less well-established, individual predator control operations should be planned based on the principles and examples presented here, taking into account such factors as choice of toxin, bait material and method of delivery. Methods should be tested and monitored for each new programme until their efficacy and target-specificity have been established.

In evaluating the potential risk posed by poison baiting, a spectrum of evidence may be considered, ranging from relatively weak evidence through to unequivocal proof that baiting has an undesirable impact on non-target species at a population level. Weak evidence comes from estimating the sensitivity of non-target species to the toxin in laboratory studies. Such information is readily available in many cases, but may provide little indication of the actual or realized risk faced by animals in the field. Thus, where non-target species are found to be sensitive to a toxin, this does not necessarily indicate that poison baiting is unsafe for these species, but does mean that field testing is required. More convincing evidence may be obtained by quantifying the susceptibility of a species to poison baiting. Such an assessment requires detailed information on the propensity of animals to locate and consume baits, as well as the likelihood of mortality if baits are consumed. Still stronger evidence may be obtained if mortality of non-target animals is recorded in the field as a direct result of poisoning. However, such evidence still does not account for the possibility of compensatory demographic responses, or benefits accrued from the removal of pest animals. Thus, conclusive proof that predator baiting has a negative impact on non-target populations may be obtained only if any observed non-target mortality is followed by sustained reductions in population density.

Baiting operations should not be assumed to be harmless in the absence of proof to the contrary. The difficulty of obtaining such proof means that it is unlikely to be available in most cases, even where a negative impact may exist. Therefore, a precautionary approach is required. The possibility of a population-level impact cannot be dismissed without rigorous trials. For this reason and from an ethical point of view, the death of non-target individuals should be avoided wherever possible.

FUTURE RESEARCH

As discussed by Choquenot & Ruscoe (1999), some level of mortality among non-target populations may be deemed acceptable if it is balanced or outweighed by the benefits of pest control. For non-target species of potential concern, future research should aim to quantify such levels of ‘acceptable mortality’. First, it is necessary to know whether any poison-induced mortality is additive, or if there is a compensatory response. Then, using detailed demographic data, mathematical models may be created allowing population trends to be projected according to varying levels of mortality due to poisoning.

Further research is also required to clarify the mechanisms behind the responses of wildlife populations following predator control. For example, the rapid response observed by Körtner & Watson (2005) in activity levels of quolls after poison baiting suggests a behavioural change. Abundance is unlikely to have changed so rapidly. The demographic response of quolls to canid removal must be monitored in the longer term to determine whether the apparent release from interference competition results in increased abundance; release can be expected to occur commonly in such situations (Glen & Dickman, 2005; Glen *et al*., 2007).

Further research is also required to develop more target-specific methods of poison delivery, as even occasional deaths of non-target species should be avoided wherever possible. For example, although laboratory trials suggest that some bait materials are more-target specific than others, direct comparisons in the field would increase the applicability of such results.

ACKNOWLEDGEMENTS

Research funding was provided by the Pest Animal Control CRC, Forests NSW, NSW Department of Environment & Conservation, Australian Geographic Society, Australian

Academy of Science, Wildlife Preservation Society of Australia, Foundation for National Parks & Wildlife and Royal Zoological Society of NSW. Thanks to Peter Cremasco, Queensland Department of Primary Industries and Fisheries for access to unpublished data, and to Andrew Claridge, Glen Saunders and two anonymous referees for comments that helped greatly to improve the manuscript.

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Submitted 4 January 2006; returned for revision 6 June 2006; revision accepted 27 February 2007
Editor: GSa