

The dingo and biodiversity conservation: response to Fleming *et al.* (2012)

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Abstract. Several authors have recently argued that dingoes could be used to help conserve biodiversity in Australia. Fleming *et al.* (2012) [*Australian Mammalogy* 34, 119–131] offer the alternative view that restoration of dingo predation is unlikely to help native species, and is more likely to do harm. We think many of the arguments used by Fleming *et al.* to reach that conclusion are either unsound or beside the point, and we explain why.

Additional keywords: *Canis lupus*, ecological restoration, feral cat, mesopredator release, red fox, trophic cascade.

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Introduction

In 2001, Dr Alan Newsome of the CSIRO argued that part of the dingo fence should be moved: he wanted the long north–south section that keeps South Australian dingoes out of the comparatively dingo-free west of New South Wales shifted eastwards. His vision was that dingoes, by controlling pest animals, would help repair the battered ecosystems of far western New South Wales (Newsome 2001). Ten years on, the dingo fence is still in the same place but the evidence in support of Newsome's view of the ecological impact of dingoes has grown stronger. As a result, positive management of dingoes is being seriously discussed as a tool to help prevent further loss of biodiversity from this continent, and assist recovery of degraded landscapes. In its boldest form this 'positive management' could consist of the reintroduction of dingoes to places where they have been extirpated (Dickman *et al.* 2009), but is more likely to involve relaxation of control of dingo populations that are currently held below their ecological potential by poisoning, trapping and shooting. This strategy could offer a low cost (in fact, a cost-saving) pathway to restoration of ecological resilience in degraded and fragile ecosystems.

These ideas are controversial, because dingoes threaten livestock and are treated as pests over most of Australia (Fleming *et al.* 2001). Management is further complicated in some areas by extensive hybridisation with domestic dog breeds, and replacement of dingoes by feral dogs. These factors limit the extent to which wildlife managers would be able to use dingoes to help conserve other wildlife species. Fleming *et al.* (2012) go further, arguing that positive management of dingoes might not achieve conservation aims at all, and would be fundamentally misguided. They lay out their critique in the form of seven 'considerations' that, in their view, affect the

potential for positive management of dingo populations to have beneficial effects.

This response does not present a comprehensive review of the evidence and arguments that support the concept of biodiversity conservation through positive management of dingoes, because that has recently been done elsewhere (Letnic *et al.* 2012; Ritchie *et al.* 2012). Instead, we respond to the critique of that concept by Fleming *et al.* at points where we think their arguments are unsound and unsupported by published research. Fleming *et al.* concentrate on the role of the dingo in suppressing populations of the red fox (*Vulpes vulpes*) and feral cat (*Felis catus*) so we do likewise, although we emphasise that regulation by dingo predation of populations of generalist herbivores is also significant to conservation of biodiversity, and provides a separate and compelling justification for positive management of dingoes. Our response follows the structure of Fleming *et al.* (2012), responding to their seven considerations in turn.

Any discussion of wild *Canis lupus* in Australia raises terminological difficulties. We use the term 'dingo' to encompass animals that could be assigned to the subspecies *C. l. dingo*, as well as wild dogs with mixed ancestry but which have morphological and behavioural characteristics considered typical of *C. l. dingo*, and so can be presumed ecologically similar to 'true' dingoes.

One, two and three: nature of the dingo, and altered Australian landscapes

The first three considerations raised by Fleming *et al.* (2012) can be summarised as follows. The dingo is Australia's largest terrestrial predator, but it is a mistake to assume that it plays the same ecological role in Australia as apex predators in other parts of the world; it does not have a long evolutionary history

in the Australian environment, and is the largest predator nowadays only because of changes to the Australian mammalian assemblage caused by people; it is not a native species, having been transported to Australia by people; moreover, the dingo experienced genetic and behavioural change due to partial domestication early in its history, meaning that Australian dingoes ‘... may not be readily likened to apex predators elsewhere (including grey wolves) that have not undergone such extensive genetic and phenotypic changes’; further, Australian ecosystems have been fundamentally altered by European impact, so we should not imagine that the dingo could recreate ecological relationships that might have prevailed before European arrival.

Most of what Fleming *et al.* say about the history of the dingo and modification of Australian environments is true, but these observations are not helpful as a guide to management. We should base our management of the dingo on an understanding of its interactions with other species in contemporary Australian environments, rather than on its past or the closeness of its fit to the theoretical ideal of an apex predator. Fleming *et al.* (2012) imply that much of the recent discussion of the dingo has been based on the naïve assumption that because it is Australia’s largest terrestrial predator, the dingo will have the same effects in Australian ecosystems as apex predators do elsewhere in the world. To put that another way, Fleming *et al.* seem to believe that many people are applying a label to the dingo – it is an ‘apex predator’ – then using that label to construct arguments of this kind: the ecological literature tells us that apex predators suppress mesopredators; the dingo is an apex predator and foxes and cats are mesopredators; ergo, positive management of dingoes will help control foxes and cats. This would explain why they repeatedly claim that ecological reality is being ignored in the current debate, and why they sometimes characterise the positions of others as being derived ‘automatically’.

But the research that has stirred current debate over the role of the dingo in conservation of Australian wildlife has been predominantly empirical, not theoretical. It has aimed to answer questions about the interaction of dingoes with other predators and prey in contemporary Australian environments, and to understand the ecological effects of variation in dingo abundance. It does not consist of the uncritical translation to the dingo in Australia of ecological concepts developed elsewhere. And it is worth stressing that this research effort has been motivated by some painful encounters with ecological reality, such as the attempt to help one of the last two populations of the mala (*Lagorchestes hirsutus*) on mainland Australia by poisoning dingoes, which action was quickly followed by invasion of red foxes and the extinction of the mala population (Lundie-Jenkins *et al.* 1993).

Four: dingoes threaten biodiversity too

Fleming *et al.* argue that because dingoes are predators they potentially threaten prey species, just as foxes and cats do; there is extensive diet overlap among dingoes, foxes and cats, so many of the species that we want to protect from foxes and cats will also be eaten by dingoes; dingoes, being versatile, medium-sized predators – in fact, archetypal mesopredators – are capable of

imposing high intensities of predation on some prey species, and may often turn out to be just as destructive as foxes and cats, so ‘... one may ask if the biodiversity conservation outcomes are any greater if a species is extinguished by a dog instead of a fox or feral cat’.

Here, Fleming *et al.* are at risk of falling into the error that they warned against in the first part of their paper: applying a label to a species (‘dingoes as dogs are and always were mesopredators’) and assuming that this label will predict its interactions with other species and its effect on ecosystem processes. But it is a mistake to treat designations such as ‘mesopredator’ as if they represent fixed ecological types that always produce the same effects in ecosystems (Ritchie *et al.* 2012). What matters is the way in which particular behavioural and ecological traits of co-occurring species affect the outcomes of interactions between those species, in particular environmental contexts. We emphasise this point by considering the conditions that could result in the interaction of a larger with a smaller predator providing net benefit (that is, reduced total intensity of predation) for prey species vulnerable to predation, even when those prey species are eaten by both predators. The most important of these conditions are:

1. If the smaller predator lives at much higher densities than the larger predator

In that case, replacement of the smaller by the larger predator can result in a lower total predator biomass, and therefore lower predation rates on vulnerable prey. For example, the switch from fox-dominated to lynx-dominated communities in Finland reduced predation rates on hares (*Lepus timidus*), even though both lynx (*Lynx lynx*) and red foxes kill hares, because the typical density of red foxes in Finland is ~40 times that of lynx (Elmhagen *et al.* 2010).

2. If the larger species is highly aggressive to the smaller

Interspecific killing is common among mammalian predators, and it often involves larger species maliciously killing smaller ones (Palomares and Caro 1999; Ritchie and Johnson 2009): lynx suppress fox populations by killing foxes, but they often leave their bodies uneaten (Helldin *et al.* 2006). Such malicious killing increases the strength of the effect of the larger on the smaller predator in two ways. First, the rate of killing is limited not by the food requirements of the aggressor, but only by its motivation and ability to attack its victims. Second, the fear of attack motivates the smaller predator to avoid places where it is at high risk of encountering its larger enemy. This avoidance behaviour constrains activity and habitat choice, and thereby limits the impact of the smaller predator on its prey. Both factors allow a low-density population of a larger predator to exert strong control over a potentially much more abundant smaller predator.

Red foxes and feral cats typically reach much higher densities than dingoes in a wide range of Australian environments (Fig. 1). Depending on rainfall, which has a strong effect on fox density but, possibly, less impact on dingo density (Fig. 1), foxes may be 20 or more times more abundant than dingoes. This difference in density overwhelms the roughly three-fold difference in mass between the two species. Therefore,

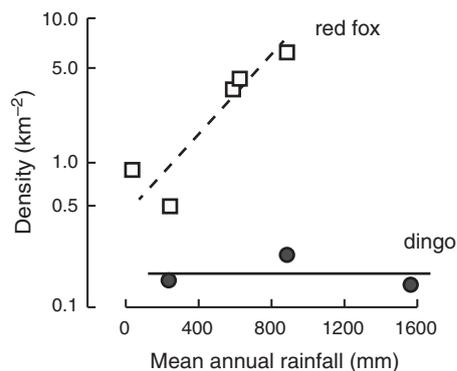


Fig. 1. Estimates of population density of dingoes and foxes across Australia in relation to mean annual rainfall. The estimates of density of dingoes are for populations not subject to control; the estimates of fox density are from populations that may have been subject to some control, but mostly of unknown intensity. Data on dingoes are from Thomson (1992), Corbett (2001) and Fleming (1996b, cited in Fleming *et al.* 2001). Data on foxes are from Coman *et al.* (1991), Marlow (1992, cited in Saunders *et al.* 1995), Thompson and Fleming (1994), Marlow *et al.* (2000) and Saunders *et al.* (2002).

even partial replacement of foxes by dingoes could bring large reductions in predator biomass. Recent research has shown that dingoes maliciously kill foxes and feral cats (Moseby *et al.* 2012). So, ecological differences between dingoes, on the one hand, and foxes and feral cats, on the other, and the nature of the interactions between them, make it plausible that suppression of mesopredators by dingoes could provide relief from predation even for species that are eaten by dingoes as well as foxes and cats.

Fleming *et al.* cite three pieces of evidence to support their view that dingo predation is currently a significant threat to native species, or has caused declines in the recent past. First, dingo predation has been identified as a threat to the surviving populations of two species, the northern hairy-nosed wombat (*Lasiorninus krefftii*) and the bridled nail-tail wallaby (*Onychogalea fraenata*). But these populations are tiny, containing as few as 100 individuals or so. For a population so small, any predation is potentially catastrophic. It does not follow that widespread populations of the same species would be equally vulnerable to predation, and there is no evidence that dingoes contributed to the initial declines of the wombat or nail-tail wallaby. Second, ‘dog predation’ is listed as a ‘known or potential’ threat in 14 national threatened species recovery plans. We regard this evidence as so weak as to be negligible. Species recovery plans often cite a wide range of threats without solid evidence that they act on the species in question or that their magnitudes are significant. Third, Fleming *et al.* rely on Corbett (2001) to assert that ‘... following the introduction of rabbits, permanent water and livestock in central Australia, subsequent high dingo densities were ultimately responsible for the demise of nine native mammals following the extended droughts in 1925–30 and 1958–65’. But Corbett’s interpretation of these events was entirely conjectural: there is no evidence linking dingoes to those extinctions (Johnson 2006), nor is there even good evidence that dingo densities increased in the way that Corbett proposed.

Five: the evidence for mesopredator release

The fifth consideration of Fleming *et al.* (2012) is the most important, because it concerns the evidence that dingoes do suppress foxes and cats, so that reduction of dingoes could lead to release of fox and cat populations. They argue, first, that manipulative experiments show that control of dingoes does not lead to mesopredator release, and, second, that there is no credible evidence of any kind for mesopredator release involving dingoes, foxes or cats in Australia. In support of their first point, Fleming *et al.* cite four studies: Eldridge *et al.* (2002), Claridge *et al.* (2010), Fleming (1996) and Allen (2005). What do these studies show?

Eldridge *et al.* (2002) set out to test the effects of reduction of dingo abundance on the activity and abundance of other predators and wild herbivores, and rates of predation on livestock. They worked on three central Australian cattle stations, distributing poison baits over half of each property and using the other half as a control. The study was well designed, but it did not achieve its aims because the baiting program had very little effect on dingo activity, which was reduced by only ~20% on treatment areas. This reduction was clearly too small to affect any of the target variables, which did not differ between treatment and control areas. Eldridge *et al.* attributed the failure of their baiting program to the exceptionally good seasons that prevailed during their study.

Claridge *et al.* (2010) monitored the relative abundance of predators and prey in three forest areas of south-east Australia over a ten-year period. In two areas, poison baiting was being used to control fox populations; the third area was unbaited. Baiting for foxes was successful: fox abundance declined steadily over the 10 years of the study in the two baited areas, but not in the unbaited area. However, baiting had no discernible impact on dingoes. Their abundance increased in one of the baited areas, fluctuated with no long-term trend in the other, and declined slightly in the unbaited area. The design of this study does not lend itself to tests of the effects of changed dingo abundance on foxes and cats, and Claridge *et al.* did not analyse relationships among predator populations; nonetheless they concluded that ‘... the relative abundance of wild dogs, foxes and cats are clearly far more complex than that implied by mesopredator release theory’. However, they did not formally test for interactions among predator species, and their data show two intriguing trends that do suggest suppression of foxes by dingoes. First, the rate of decline of foxes was greater in one of their baited areas where dingoes increased than in the other baited area where there was no trend in dingo abundance. Their fig. 3 suggests rates of fox decline of ~3.4 versus 1.5 units of their abundance index per year in these two areas. That is, more than half of the fox decline in the first area could be attributed to the effect of an increasing dingo population, rather than to fox baiting. Second, there was an intriguing inverse relationship of the abundance trends of dingoes and foxes on the unbaited site (Fig. 2): during periods when dingo abundance was rising, fox abundance fell, and *vice versa*. This cannot be explained by differential responses to rainfall, because rainfall showed no corresponding pattern over the 10 years of the study, nor by shifts in habitat structure, which did not occur.

Fleming (1996) reported a six-week poison baiting trial that successfully reduced both dingo and fox activity by 76.1% and

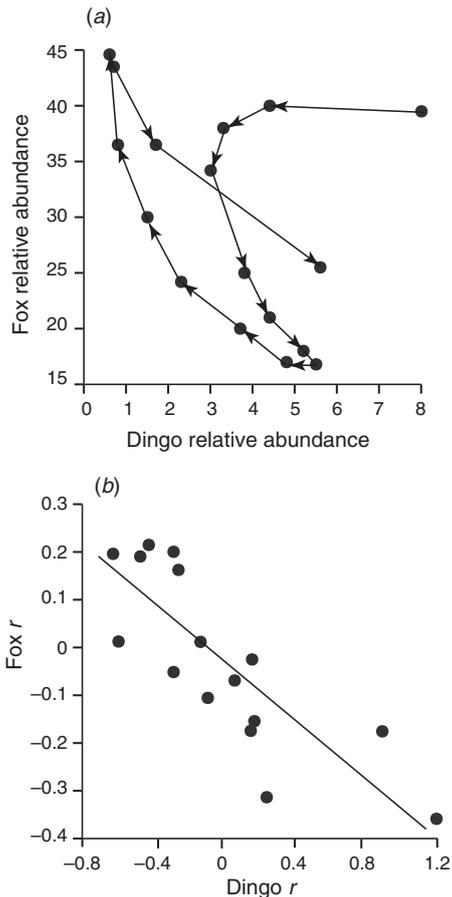


Fig. 2. Reanalysis of data from Claridge *et al.* (2010) on the population dynamics of foxes in relation to dingoes in a south-eastern New South Wales forest area not subject to poison baiting: (a) relationship of relative abundance measures of the two species through time, and (b) inverse relationships of exponential rates of increase of the two species. Claridge *et al.* used regression splines to describe fluctuations in relative abundance of each species, and we transcribed values from these fitted splines.

90.8% on two experimental sites, with no reductions on a control site. The study was not designed to test the effect of changes in dingo activity on foxes, but again there is an interesting trend in the data that does suggest such an effect. The prebaiting indices of abundance of dingoes and foxes on the three sites were inversely related (Fig. 3). Possibly, this could be explained by habitat differences: the activity of foxes was less in sites furthest from cleared land. But in the control (unbaited) site, dingo activity increased in the postbaiting measurement period, possibly because of increased movement of immigrant animals through the area in response to removals at the two baited sites. This increase in dingo activity was associated with a decline in fox activity, the magnitude of which was consistent with the negative relationship observed for the prebaiting measurement (Fig. 3). This may be a hint that the original variation in fox activity across sites was a function of dingo activity, not habitat.

The study of Allen (2005) used a similar design to that of Eldridge *et al.* (2002). He worked on two cattle stations, one in semiarid southern Queensland and the other in dry-tropical

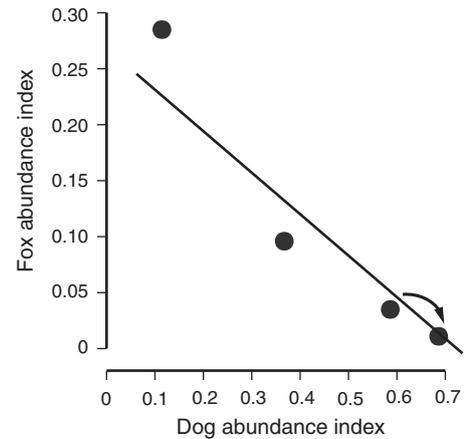


Fig. 3. Data from Fleming (1996) on the relationship between relative abundance of foxes and dingoes at prebaiting samples, with postbaiting samples for the unbaited control site (arrowed) also shown.

north Queensland. Half of each property was subjected to 1080 baiting directed at dingoes while the other half was left unbaited. Abundance indices – mostly, track counts on sand plots – were measured for dingoes and a variety of wildlife species on both halves. Baiting reduced dingo abundance, but abundance usually recovered quickly through immigration. There were no foxes in the tropical study area, and foxes were extremely rare in the southern area – only 19 fox tracks were seen in five years, compared with 2297 dingo tracks; this study tells us nothing about effects of changed dingo abundance on foxes. Feral cats were present in both areas, but track counts were consistently low. Allen (2005) could find no effect of dingo baiting on feral cat abundance, but he conceded that the track count index is not a sensitive measure of the relative abundance of feral cats. It is therefore possible that this study failed to detect variations in cat abundance that might have been ecologically significant.

None of these four experimental studies was designed to investigate interactions among predator species and, apart from Eldridge *et al.* (2002), the authors of these studies did not conduct formal analyses to test for such interactions (even though the results of two of the studies contained trends consistent with suppression of foxes by dingoes). Therefore, these studies add little to our understanding of interactions among predators in Australian ecosystems, and they do not provide guidance on whether biodiversity conservation might be better served by not attempting to control dingoes.

This collection of studies does show that control programs aimed at dingoes do not cause mesopredator release if they fail to reduce dingoes, and that foxes do not benefit from poison baiting of dingoes if they are also poisoned. Fleming *et al.* conclude from this that dingo control would not indirectly threaten prey species by releasing mesopredators, and therefore there is no reason to call for cessation of dingo control on conservation grounds. There are several problems with this. While it is true that foxes are susceptible to 1080 baiting, which is the main method used in dingo control, there are circumstances under which poison baiting can have larger impacts on dingoes than foxes. For example, in arid environments

dingoes concentrate their activity near water, and when dingoes are not controlled the activity of foxes and cats is correspondingly reduced near water (Brawata and Neeman 2011). Baiting at water points can therefore selectively remove dingoes and allow smaller predators to increase. The impact of poison baiting on a population can also depend on the density of baits in relation to the range size of individual animals. Individual dingoes may be more likely to encounter baits than foxes when baits are dispersed at low density, because dingoes have larger individual ranges than foxes. Poison baiting is often augmented by trapping and (especially) shooting of dingoes with the aim of preventing dingo recovery, and this could indirectly favour recovery of mesopredators. And control of either dingoes or foxes, or both, by poison baiting can still release cats, which do not readily take poison baits and can be as destructive as foxes when released from control by larger predators (Risbey *et al.* 2000; de Tores and Marlow 2012).

Fleming *et al.* then claim that there is, in any case, no strong evidence for negative effects of dingoes on foxes and cats. They base this mainly on a critique by Allen *et al.* (2011), who purported to find major methodological flaws in studies that have presented evidence of suppression of mesopredators by dingoes. However, the review by Allen *et al.* (2011) was itself deeply flawed. This is best shown by example. Southgate *et al.* (2007) surveyed the distribution of bilbies (*Macrotis lagotis*) over more than 207 sites in the Tanami Desert, and found that bilbies were more likely to be present in areas occupied by dingoes, where foxes were less likely to occur. Allen *et al.* (2011) rejected this result because the survey of Southgate *et al.* included a range of habitats, and predator activity may have been affected by habitat. But the objective of Southgate *et al.* (2007) was to compare the effects of habitat versus interactions with other species on the distribution of bilbies. To do this, they stratified their sampling by latitude, substrate type and fire history to give equal coverage of major habitats, then surveyed for all target species across all habitats, and used generalised linear modelling to remove effects of habitat so that they could analyse relationships among species independent of habitat. Sampling across a range of habitats was not a weakness of this study: it was essential for the study to achieve its aims, and the study design and the handling of the complexities of the data were appropriate to those aims.

Allen *et al.* (2011) also dismissed the study of Southgate *et al.* (2007) because while they registered the occurrence of animals on their plots by counting tracks, they converted these track counts to categorical presence/absence scores for each plot. Allen *et al.* regarded this as invalid, because continuous measures such as track counts should be preferred to categorical measures such as presence/absence scores. But the procedure used by Southgate *et al.* was sensible because it avoided inflation of counts due to recrossing of plots by individual animals. Because a presence/absence score is less sensitive to variation in animal abundance than is a total count of tracks, filtering the track-count data in this way is statistically conservative. That is, it makes it more difficult to detect a trend, such as the inverse relationship of dingoes to foxes that Southgate *et al.* found, when that trend is truly present. To reject evidence of a trend because it was detected using a conservative procedure is faulty statistical reasoning.

Allen *et al.* (2011) included 20 studies in their review, and rejected all but one of them as being methodologically flawed (the one exception was Allen (2005), which, as noted above, did not include foxes, and used a survey method with low sensitivity to cats). But, throughout, their assessments suffer from the problems noted above with respect to Southgate *et al.* (2007): superficial evaluation of study designs, and fundamental errors in statistical reasoning. In our view, Allen *et al.* (2011) should not be relied on as an evaluation of the state of evidence on the interaction of dingoes with foxes and cats in Australia.

Six: let's not be simplistic

Fleming *et al.* argue that a focus on top-down effects of large predators is inappropriate because it ignores bottom-up effects due to productivity and availability of refuges from predation, which may often have stronger effects on the abundance of prey species. We agree that both top-down and bottom-up effects are likely to be important in many systems, and that their relative importance is likely to vary greatly from place to place, especially along gradients of productivity (Elmhagen and Rushton 2007; Oksanen and Oksanen 2000). But we do believe that there has been too little consideration of the role of large predators in Australian ecosystems, as in terrestrial ecosystems elsewhere in the world (Estes *et al.* 2011). Fleming *et al.* illustrate that lack of consideration by stating the confident opinion that in Australia '... bottom-up factors play the most important role in ecosystem dynamics – much more than predator effects in any combination' while supporting that statement only by reference to two wildlife field guides and a popular natural history book.

In any case, the interest in the ecological effects of dingoes on foxes, cats, and their prey species is not primarily motivated by a theoretical belief in the dominant role of top-down processes, in Australia or anywhere else. It has a very different source. The state of mammal conservation in Australia is catastrophic. More than 20 species of mammals have already gone extinct, and many more have declined to small remnant populations (Johnson 2006). These declines are continuing, with the recent collapse of mammal biodiversity across vast areas of northern Australia bringing the possibility of more extinctions in the near future (Woinarski *et al.* 2011). Past declines and extinctions are, for the most part, attributable to the impacts of red foxes and feral cats, and there are strong indications that predation by cats is a major contributor to current declines in the north. We have few options for effective control of these species over large areas, and therefore little capacity for restoration of mammalian biodiversity at landscape scales. Bottom-up processes can be manipulated to allow recovery of small and medium-sized mammals in the presence of foxes and cats, but this entails costly actions such as removal of livestock (Legge *et al.* 2011), and is not always effective if foxes and cats remain abundant. Direct action against foxes can work, but only with huge effort, and reduction of foxes can lead to mesopredator release of cats and thus to further waves of mammal decline (Risbey *et al.* 2000). The most successful small-scale conservation projects on mainland Australia use fencing to exclude all predators (de Tores and Marlow 2012), but it is not feasible to fence off large conservation reserves and maintain exclusion of predators forever, and this

strategy preserves prey populations in a state of naivety to predators and high vulnerability to predator incursions.

Hence the interest in dingoes. If dingoes do have strong effects on foxes and cats, they could provide a low-cost and effectively permanent reduction of the impact of those mesopredators over large areas. Because this form of management would not aim to eliminate predation entirely, but instead hold it at intermediate levels which could be absorbed by prey populations, it provides for the maintenance and, in the long term, the evolution of antipredator behaviour in vulnerable prey. Moderation of predation at large scales could also provide the potential for prey populations to respond to spatial differences in predation pressure caused by habitat variation, and for processes such as source–sink dynamics to confer resilience on the demographics of prey populations. Significant benefits could be secured in this way whether or not top-down control is generally more significant than bottom-up control in Australian ecosystems.

Seven: the loaded dog

Coman and Jones (2007) used the Henry Lawson story of the playful dog in the mining camp who persistently returned a lighted blasting cartridge to his owners as an apt metaphor to describe the effect of the dingo on environmental debates in Australia. The dingo elicits divergent opinions like no other species of Australian wildlife, and discussions on dingo management can quickly turn into explosive disputes. We therefore agree with Fleming *et al.* that arguments for change to management of dingoes should be carefully constructed, and that development of alternative management plans should be collaborative and involve all affected parties. But the management of dingoes in Australia is so fraught with conflict that it is unlikely that we will see significant change in situations where multiple stakeholders are involved. New approaches are most likely to emerge where single stakeholders have control over large areas of land, and are prepared to innovate in pursuit of clearly defined management goals. The best current example of this is provided by the Australian Wildlife Conservancy, who are implementing positive management of dingoes to aid recovery of wildlife across a series of large conservation reserves, and monitoring effects on mesopredators and their prey (Kennedy *et al.* 2012; see also the AWC newsletter Summer 2010/11: <http://www.australianwildlife.org>). Bold and focussed initiatives such as this will test the effectiveness of positive management of dingoes for conservation, and may eventually show the way to more widespread adoption if such management works.

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