

OPINION ARTICLE

Resolving the value of the dingo in ecological restoration

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There is global interest in restoring populations of apex predators, both to conserve them and to harness their ecological services. In Australia, reintroduction of dingoes (*Canis dingo*) has been proposed to help restore degraded rangelands. This proposal is based on theories and the results of studies suggesting that dingoes can suppress populations of prey (especially medium- and large-sized herbivores) and invasive predators such as red foxes (*Vulpes vulpes*) and feral cats (*Felis catus*) that prey on threatened native species. However, the idea of dingo reintroduction has met opposition, especially from scientists who query the dingo's positive effects for some species or in some environments. Here, we ask 'what is a feasible experimental design for assessing the role of dingoes in ecological restoration?' We outline and propose a dingo reintroduction experiment—one that draws upon the existing dingo-proof fence—and identify an area suitable for this (Sturt National Park, western New South Wales). Although challenging, this initiative would test whether dingoes can help restore Australia's rangeland biodiversity, and potentially provide proof-of-concept for apex predator reintroductions globally.

Key words: Australia, *Canis dingo*, extinction, mesopredator release, top predator

Implications for Practice

- To resolve the ongoing debate about whether the dingo can help to restore Australia's degraded rangelands, the best way forward is that a controlled reintroduction experiment be undertaken by realigning the dingo-fence that borders Sturt National Park.
- Sturt National Park provides an optimal reintroduction site because critical interactions between dingoes, mesopredators, overabundant herbivores, native prey, and vegetation can be tested.
- This article sets out the benefits and appropriate design of such a reintroduction study, including the size of the reintroduction area, the nature and number of monitoring sites, its duration, and legislative requirements.

Introduction

The global biodiversity crisis is evidenced by current species losses that are about 1,000 times the background rate (Pimm et al. 2014). Australia provides a case in point, contributing about half the world's mammal extinctions over the last 200 years, with the loss of 29 endemic species (Woinarski et al. 2014). A consistent factor in these extinctions is predation by introduced red foxes (*Vulpes vulpes*) and feral cats (*Felis catus*), which have devastated small- and medium-sized native animal populations (Dickman 1996; Johnson 2006).

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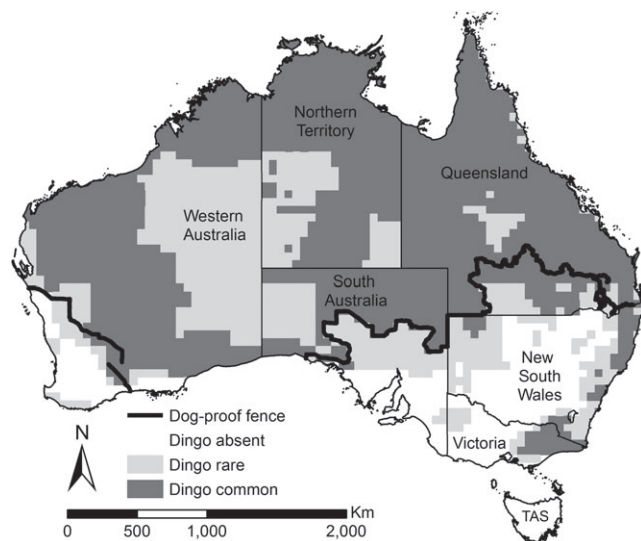


Figure 1. Distribution of the dingo (*Canis dingo*) and hybrids (*Canis dingo* × *Canis familiaris*) in Australia (modified after Letnic et al. 2012). Thick black lines are State Government-maintained dog-proof fences.

Despite perceived intensive efforts, attempts to alleviate the negative impacts of these predators using lethal control are costly and, if not sustained at a high intensity, they often fail (Moseby et al. 2011a; Newsome et al. 2014). One suggested solution is to restore populations of dingoes (*Canis dingo*) in areas where their abundance has declined due to lethal control and exclusion (Dickman et al. 2009; Ritchie et al. 2012) (Fig. 1). This idea is based on theories and the results of studies suggesting that dingoes can suppress mesopredators such as red foxes and feral cats, in turn benefiting small prey by reducing overall predation pressure (Glen et al. 2007; Ritchie & Johnson 2009; Letnic et al. 2012). Additional benefits appear to accrue from suppressive effects of dingoes on other medium-sized introduced species such as feral goats (*Capra hircus*) and native herbivores such as kangaroos (*Macropus* spp.) that together contribute to rangeland degradation when in high densities (Caughley et al. 1980; Newsome et al. 2001; Letnic et al. 2012).

Proposed restoration of dingo populations has, however, met with opposition. Some scientists propose that dingoes themselves could pose a threat to some fauna (Allen & Fleming 2012), which we acknowledge could be a factor for prey species that are more efficiently killed by dingoes than red foxes or feral cats, while others suggest that studies identifying ecological benefits of dingoes have used flawed methodology (Allen et al. 2013b; Hayward & Marlow 2014) or that the evidence for positive effects of dingoes on biodiversity is inconclusive (Fleming et al. 2012), or absent (Allen et al. 2014). These critiques have elicited responses (Letnic et al. 2011; Johnson & Ritchie 2013; Johnson et al. 2014; Nimmo et al. 2014), but the proposal remains controversial (Fleming et al. 2013). In particular, debate surrounds the applicability of the mesopredator release hypothesis (Crooks & Soulé 1999) to Australian ecosystems for two reasons. First, while dingoes appear to have facilitatory effects on small mammals, they did

not prevent red fox-driven extinctions of some medium-sized mammals (Letnic et al. 2012). Second, some scientists argue that dingoes are themselves mesopredators, and thus might have unexpected ecological effects (Fleming et al. 2012). This impasse creates a need to further define the ecological role of the dingo.

As a starting point, we evaluate the merits and feasibility of a dingo reintroduction experiment to help determine whether dingoes can help restore Australia's degraded rangelands and populations of some threatened species. We focus on western New South Wales (NSW) because of ongoing discussions about restoring dingo populations in that region (Newsome 2001; Dickman et al. 2009; Allen & Fleming 2012). There is, however, scope to use our commentary in designing studies of apex predators' roles in other parts of Australia and globally.

Why consider reintroduction experiments?

Manipulative experiments can provide robust platforms for examining the ecological roles of apex predators (Hone 2007; Allen et al. 2013a). Two broad classes of such experiments are possible: (1) experiments that use lethal control to reduce densities of the apex predator ("lethal experiments") and (2) experiments that reintroduce the predator into ecosystems where it once occurred, but has since become locally extinct or rare ("reintroduction experiments"). Of these options, a reintroduction experiment would be preferable with respect to the dingo. This is because most studies assessing the ecological roles of dingoes have already used lethal control (e.g. Allen et al. 2013a, 2014) or exclusion coupled with lethal control (e.g. Newsome et al. 2001; Letnic et al. 2009a; Letnic & Koch 2010) as the treatment. Therefore, although still debated, there is some basis for understanding what happens under conditions of dingo control and exclusion. However, ecosystems commonly change in different ways when perturbations are initiated compared to when they are relaxed, a phenomenon known as "hysteresis" (Beisner et al. 2003). Currently, there is little understanding of what happens when dingo densities increase over ecologically significant spatial and temporal scales (Wallach et al. 2010). If we are to use dingoes as tools for biodiversity restoration, this knowledge gap needs urgently to be filled to avoid "ecological surprises" (*sensu* Lindenmayer et al. 2010) that might undermine restoration goals.

How could a reintroduction experiment be undertaken?

In a dingo reintroduction experiment, a regression design with replicated dingo densities ranging from zero (dingo exclusion) to carrying capacity would provide good inference and information (Hone 2007). However, this approach would be infeasible on the ground. A more viable option is a Before-After-Control-Impact (BACI) design (e.g. Popescu et al. 2012) whereby dingoes are added to a reintroduction area with nested monitoring sites that can be contrasted with

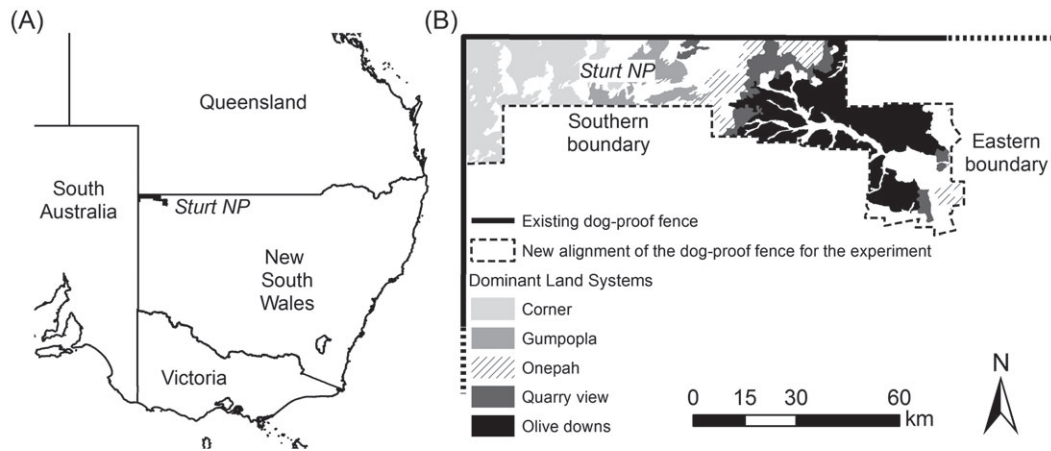


Figure 2. (A) Location of Sturt National Park in the Western Division of New South Wales, Australia and (B) dominant land systems (shaded areas) within the Park. Dingoes may recolonize or build up densities naturally from within the Park should the dingo (dog-proof) fence be realigned on the southern and eastern boundaries.

control sites in surrounding areas, matched where possible to ensure environmental similarity (e.g. broad vegetation types, land-use history, and prey density). This approach accords with Underwood (1992), who proposed that when spatial randomization or treatment replication is infeasible, multiple independent and randomly chosen control sites can be used to obtain average changes for contrast in the BACI framework.

One previous proposal was to realign a section of the 5,500 km dingo-proof fence bordering Sturt National Park in north-western NSW (Newsome 2001) to allow natural recovery of the existing dingo population and recolonization to the National Park from the north and west (Fig. 2). Alternatively, a dingo-proof fence could be added on the southern and eastern boundary to effectively enclose the Park so that densities of dingoes could increase within the Park. The latter approach has the practical advantage of not requiring negotiations with neighboring States (South Australia and Queensland) or changes in legislation associated with realignment of the existing fence. Under this design, however, dingoes cannot recolonize the National Park from surrounding areas, so human-assisted reintroductions may be required. A completely fenced-in population is also difficult to compare with open populations, hindering our ability to fully determine the ecosystem effects of dingo recovery.

Whether the existing dingo fence is realigned or a new fence built, there are several benefits to using Sturt National Park as the reintroduction site. First, dingoes, red foxes, and feral cats are extant (Montague-Drake 2003), and dingo abundance should increase if the dingo-proof fence is realigned and lethal control of dingoes ceases within the Park. Second, there are many red kangaroos (*Macropus rufus*) and emus (*Dromaius novaehollandiae*), moderate numbers of feral goats and European wild rabbits (*Oryctolagus cuniculus*), and some feral pigs (*Sus scrofa*) in the Park (Montague-Drake 2003), providing opportunities to examine dingoes' effects on a variety of prey species. Third, relict populations of native rodents occur in the Park (Montague-Drake 2003) that could benefit from the presence

of dingoes. Finally, Sturt National Park is sufficiently large (3,000 km²) to support several packs of dingoes, and has diverse plant communities to allow assessment of the indirect effects of dingo recovery on vegetation.

The experimental design could, and preferably would include, independent and randomly allocated monitoring sites within the reintroduction area, and these could be compared with multiple control sites outside the reintroduction area. Each monitoring site should be large enough to support at least one, but preferably several, dingo packs. In other arid areas, dingo home-ranges occupy 144–2,000 km² (Newsome et al. 2013a), suggesting that monitoring sites of approximately 700 km² would be appropriate. At this size, at least four monitoring sites could be established within Sturt National Park. Each monitoring site could have 8–10 nested monitoring plots where multiple survey techniques are employed. This would be replicated in each control site. The intent is to survey the key variables relating to a set of *a priori* predictions (e.g. Fig. 3).

Initial monitoring at all sites would need to last at least 1–3 years to establish reliable baseline data, or longer if heavy rains or fires occur to ensure that these events are encompassed. The next step would involve installing a new dingo-proof fence on the southern and eastern sides of the Park, then removing the existing fence on the northern and western sides (Fig. 2). If removing the existing fence is not possible, the alternative option of enclosing the Park could be pursued (but see limitations above). Monitoring of large mammals, other vertebrates (such as native rodents, goannas, and ground nesting birds), and vegetation should then continue for at least five years, or longer if the initial population sizes of key response species in the study system are low and dry conditions preclude swift recovery (Fig. 3). Equally, the ability of dingoes to suppress mesopredators may be stronger in dry years, when competition for resources is heightened (Greenville et al. 2014). Therefore, it would be ideal to run the experiment over multiple wet and dry cycles.

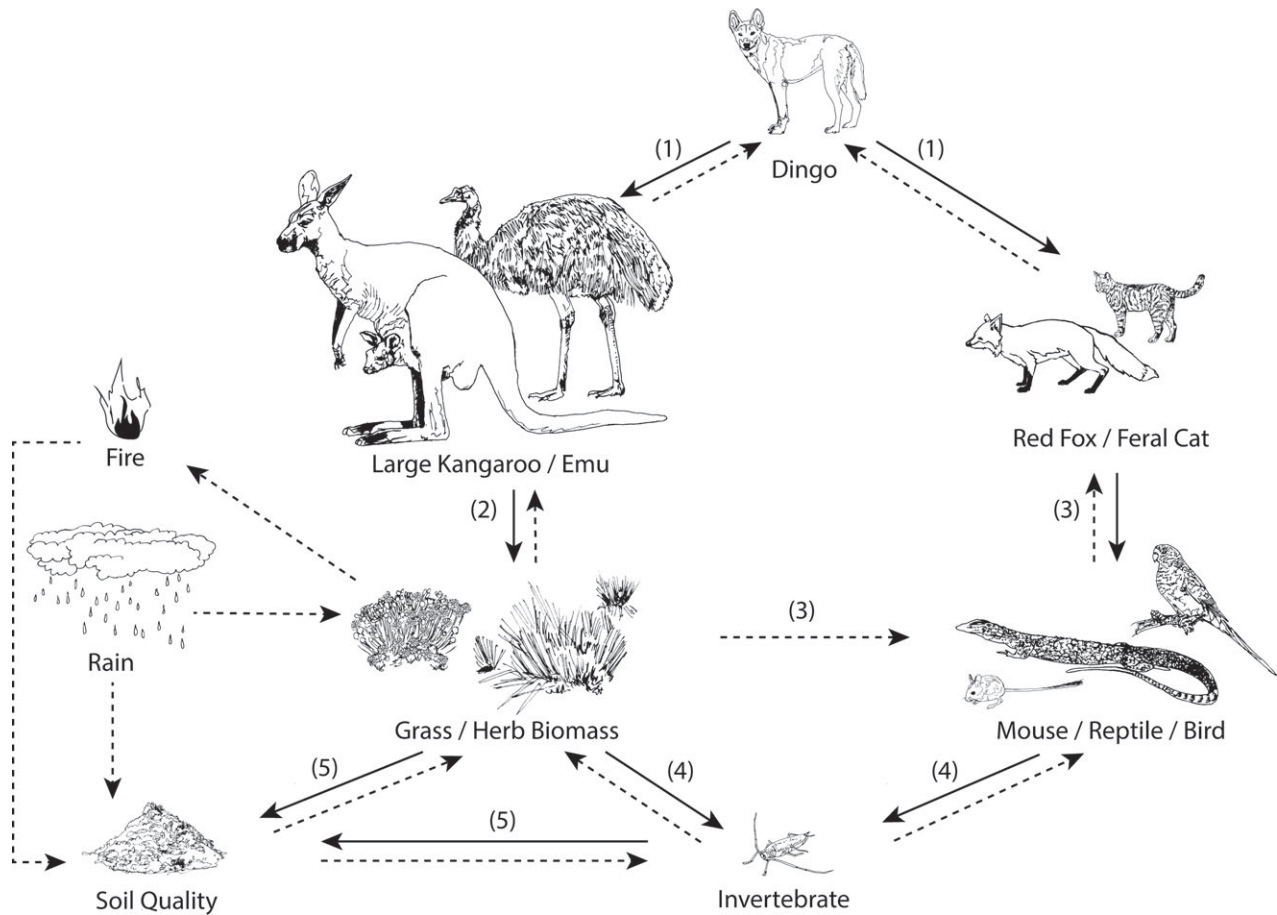


Figure 3. Predicted interactions associated with dingo recovery (solid arrows) in Sturt National Park, New South Wales, Australia (numbers in parentheses represent the predicted sequence of events). For example, if dingoes suppress large herbivores (e.g. kangaroos and emus), then grass and herb biomass is expected to increase. If dingoes also suppress mesopredators (e.g. red foxes and feral cats), then small mammals (e.g. mice), reptiles (e.g. goannas), and birds (e.g. parrots) are expected to increase in numbers, although this response may take longer to manifest than the response of vegetation. Invertebrates may also respond to improved vegetation conditions and contribute to soil quality. However, the strength of all interactions may be influenced by the extent of rainfall and fires (hashed arrows) (e.g. Greenville et al. 2014).

To undertake the experiment, we estimate that 275 km of new (or upgraded) dingo-proof fencing would be required at a cost of approximately AUS\$ 4,000/km (Table 1). However, if there was support for increasing the area of Sturt National Park, then shorter, straight-line fences could be built, with the benefit of also reducing the perimeter-to-area ratio and associated edge effects. Monitoring costs would likely be in the order of \$1 million per year (Table 1). Such expenditure is not unprecedented for large-scale environmental projects in Australia. For example, the Australian Government recently provided \$19 million to reduce the detrimental environmental impacts of feral camels (*Camelus dromedarius*) in central Australia (The Australian Government 2010), while \$35 million has been spent on the red fox eradication program in Tasmania (Tasmanian Government 2014). Satellite collaring of dingoes, red foxes, and feral cats should be undertaken to estimate population densities, rates of mortality and movement, and to provide data for behavioral studies. As hybridization with domestic dogs has been recorded across Australia (Stephens 2011),

Table 1. Estimated costs \$AUS associated with undertaking a dingo reintroduction experiment in Sturt National Park, New South Wales, Australia. Salary costs are not included. Fencing costs are based on Long and Robley (2004) under the assumption that some existing boundary/agricultural fencing could be upgraded to dingo-proof status. Equipment costs are broad estimates that should cover a range of monitoring options from GPS collars, remote sensing cameras, to ground trapping and vegetation surveys.

Item	Parameter	Cost Per Parameter	Replicates	Total Cost
Fencing (removal)	1 km	1,000	135 km	135,000
Fencing upgrades	1 km	4,000	275 km	1,100,000
Equipment/Monitoring	Year	1,000,000	8 years (minimum)	8,000,000

animals captured for collaring should be screened to determine the extent of dog gene introgression, and satisfy questions about their genotypes (Newsome et al. 2013b).

What are the major hurdles, and can they be overcome?

The major hurdle to undertaking a successful dingo reintroduction experiment would be convincing livestock producers and local communities to support the effort. That support (including acceptance from adjacent landholders of the presence of dingoes in the Park) would likely help to sway government policy, and garnering this support would require effective community engagement and extension. Any compelling argument for dingo reintroduction would need to describe potential restorative benefits for the environment (including degraded pastoral lands), but most crucially must also include local and socially acceptable strategies for preventing, and/or alleviating, negative impacts of dingoes on livestock. For economic and animal welfare reasons, any reasonable strategy should not be simply reactive and compensation-based, but rather proactive and focused upon preventing negative impacts from occurring.

The NSW Government would also need to change the current requirement under the *Rural Lands Protection Act 1998* (NSW) that dingoes be destroyed within Sturt National Park. Current NSW legislation requires that all land managers prevent declared “pest animals,” including dingoes, dispersing to neighboring properties. Sturt National Park was removed from Schedule 2 of the Pest Control Order for Wild Dogs, which gives public land managers dual obligations of dingo conservation and control activities, but it could be reinstated. Well-maintained barrier fencing that prevented dingoes from moving between the Park and other areas of NSW would effectively meet both dingo conservation and control obligations.

Cooperation from neighboring States would also help, for example, from adjoining areas in South Australia, where lethal control is not obligatory. There are cattle enterprises north of the Park, in Queensland, creating potential for conflicts in that region. However, there may be ways to align the project with existing management regimes, and in Queensland the dingo is defined as both “wildlife” and “native wildlife” under the *Nature Conservation Act 1992*, and managed as a natural resource within protected areas. Of course, if there was broad support for the reintroduction experiment in western NSW alone, it would negate the need for any new fencing and negotiations with neighboring States. However, this would require ceasing dingo control over a much larger area than Sturt National Park to ensure population recovery.

Could the reintroduction experiment in Sturt National Park be replicated elsewhere?

Our proposed experiment in Sturt National Park addresses the specific question of whether dingo reintroduction can help to restore degraded rangelands in western NSW. In theory, additional reintroduction sites could be added as spatial replicates, but this would be very difficult to achieve in practice, due to legislative, logistical, and cost constraints. However, it is appropriate to consider undertaking reintroduction or recolonization studies elsewhere to assess whether the dingo can exert strong top-down effects in other ecosystems. For example, there

are several sites south of the dingo-proof fence in South Australia (where dingoes are uncommon) that could potentially be explored as possibilities. These include Witchelina, a 4,219 km² former pastoral property that is now owned and managed by the Nature Foundation SA. Witchelina abuts the dingo fence on its western side and, if neighboring properties (to the north) were cooperative, only around 100 km of new fencing would be required to place Witchelina on the northern side of the dingo-proof fence to allow dingoes to naturally recolonize.

Complementary small-scale experiments may also be worth considering, especially if they assist in understanding the mechanisms by which dingoes drive ecological changes. For example, a reduction in red fox and feral cat abundance following dingo reintroduction could be due to intraguild predation, interference competition, or competitive exclusion, but these fine scale interactions can be difficult to discern over large spatial scales. Fenced conservation reserves, such as those managed by Arid Recovery and the Australian Wildlife Conservancy, potentially present a unique opportunity to conduct research under tightly controlled conditions and have been previously used to determine impacts of other predators (e.g. Moseby et al. 2009). Fenced reserves can also be used to closely monitor prey species inside and outside the reserve (with and without dingoes) and determine their long-term population responses while keeping other variables such as rainfall and vegetation constant. Indeed, the notion that dingoes can co-exist with threatened native mammals can be tested inside fenced reserves if low numbers of dingoes are added under controlled conditions.

What can we learn from other studies?

Two studies have reported experimental introductions of dingoes. In the first study, on Townshend Island (70 km²), Queensland (Allen et al. 1998), dingoes rapidly exterminated approximately 3,000 feral goats and facilitated dramatic and positive responses of vegetation as rates of herbivory declined (Allen et al. 2013b). In the second study, six red foxes and five feral cats were released into a fenced paddock already containing three feral cats, a red fox, and a male and female dingo (Moseby et al. 2012). Dingoes killed all seven foxes within 17 days and were implicated in the deaths of at least five of the eight cats (Moseby et al. 2012). Together, these results suggest that some species may respond dramatically to dingo restoration. However, if dingo populations are restored to Sturt National Park, several short-term, direct impacts and other long-term, indirect impacts may occur (Letnic et al. 2012). The former may be noticeable changes in behavior and/or abundance of mesopredators and medium-large sized herbivores, and the latter might be promotion of some small vertebrates and changes in vegetation community structure (Fig. 3).

The reintroduction of grey wolves (*Canis lupus*) into Yellowstone National Park and central Idaho, U.S.A. in 1995/1996 may be the most relevant international example to learn from. There, it took 20 years of debate before wolves were reintroduced, demonstrating the cultural and philosophical conflicts associated with predator-restoration efforts. Similar to the United

States and its wolves, it is likely that the idea of a dingo reintroduction in Australia would spur a strong and emotive debate. However, wolf reintroductions demonstrate the ability of a population of canids to successfully re-establish. By 2007, there were an estimated 1,500 wolves in Montana, Idaho, and Wyoming (Smith & Bangs 2009), due to the reintroduction efforts in Yellowstone and central Idaho and natural recolonization from the Canadian Rocky Mountains. This repopulation suggests that relying on recolonization (as opposed to active reintroduction) could be a feasible means of achieving an experimental treatment in the study proposed for Sturt National Park, or in other areas where dingo control ceases.

Ecological research in Yellowstone has also provided a plethora of studies that document cascading interactions involving wolves (Ripple et al. 2014), although these are not without controversy. For example, there is debate about the exact mechanisms by which wolves affect their main prey, elk (*Cervus elaphus*) (Kauffman et al. 2007; Middleton et al. 2013), and how widespread the cascading effects on vegetation have been (Kauffman et al. 2010, but see Painter et al. 2014 for the most recent analyses). Some of this uncertainty stems from a paucity of detailed studies undertaken prior to wolf reintroduction, making comparisons of pre- and post-wolf reintroduction challenging. There was also no designated control site available for comparison, making it difficult to attribute the ecological changes solely to wolves, but see Christianson and Creel (2014). Therefore, it is crucial that similar studies in Australia incorporate pre-manipulation monitoring, and both treatment and nil-treatment sites, so that any impacts of dingoes, and the likely mechanisms driving them, can be robustly tested and explained.

Are there experimental risks to a dingo reintroduction experiment?

One experimental risk is that our study design may not enable us to say whether any observed changes in vegetation, mesopredators, and prey arise from increased dingo abundance or stochastic or historical anthropogenic and natural factors (Underwood 1992). However, judicious choice of comparable sites from which the controls are selected should minimize this risk. Other risks lie in not being able to obtain sufficient replication, and running the experiment at temporal and spatial scales that are inadequate to clearly demonstrate dingo-effects. However, these risks may only be apparent under a frequentist approach and could be minimized using a Bayesian design that incorporates prior information to help inform expected effect sizes (McCarthy & Masters 2005). Such prior information is available (see Letnic et al. 2009b; Letnic & Koch 2010) and, by incorporating it, models can be assessed relative to one another and dingo-effects reliably and robustly identified.

What other benefits could a dingo reintroduction experiment provide?

Large carnivores are returning in many parts of the world, especially in North America and Europe. Along with these

recoveries, recognition is growing that the future survival of large carnivores will depend heavily upon human acceptance. Communicating the potential benefits of large carnivores, such as increased biodiversity, rangeland productivity, and associated increases in cattle live-weight or local tourism, rather than just their negative impacts, is critical to promoting such acceptance (Bruskotter & Wilson 2014; Prowse et al. 2014). Accordingly, active conservation of dingoes is more difficult without demonstration of the services they may provide and consequent commitment to finding ways for dingoes and livestock to coexist, e.g. via the use of guardian animals (van Bommel & Johnson 2012) or selection of livestock breeds and husbandry practices that minimize negative impacts.

A reintroduction study does lend itself to positive communication, focused more on what could be gained rather than what might be lost (Bruskotter & Wilson 2014). This kind of messaging contrasts with traditional communication about large carnivores, which tends to emphasize the inevitability of negative interactions. Thus, focusing objectively on quantifying potential conservation gains through dingo reintroduction may set a new standard for considering what we can achieve in conservation. For instance, if suppression of mesopredators is achieved with dingo reintroduction, it may allow for experimental reintroductions of locally extinct native mammals such as the greater bilby (*Macrotis lagotis*) and burrowing bettong (*Bettongia lesueur*). Previous attempts in other areas have largely failed because of predation by red foxes and feral cats (Moseby et al. 2011b).

Overall, we conclude that a reintroduction experiment would provide vital information for future management of dingoes and threatened endemic mammals in arid Australia. It would also add considerably to the existing body of knowledge on the ecological role of dingoes and other apex predators more broadly. We acknowledge that the experiment would be costly and involve challenges. However, our approach is purposefully bold. Recent literature on the topic of the value of the dingo in ecological restoration has frequently focused on highlighting perceived limitations of previously published studies, rather than on finding solutions. Our proposal—a controlled dingo reintroduction experiment at a scale large enough to generate meaningful results—would, if implemented as suggested, actually resolve the long-running debate: whether the dingo can help halt Australia's biodiversity collapse and help to restore degraded rangeland environments. It would provide robust data with which stakeholders could properly address an issue of national and international significance.

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