Reintroducing the dingo: the risk of dingo predation to threatened vertebrates of western New South Wales

B. L. Allen[A,C] *and P. J. S. Fleming*[B]

[A]The University of Queensland, School of Animal Studies, Gatton, Qld 4343, Australia.

[B]Vertebrate Pest Research Unit, NSW Department of Primary Industries, Orange Agricultural Institute, Forest Road, Orange, NSW 2800, Australia.

[C]Corresponding author. Present address: Vertebrate Pest Research Unit, NSW Department of Primary Industries, Sulfide Street, Broken Hill, NSW 2880, Australia. Email: benjamin.allen@dpi.nsw.gov.au

Abstract

Context. The reintroduction of dingoes into sheep-grazing areas south-east of the dingo barrier fence has been suggested as a mechanism to suppress fox and feral-cat impacts. Using the Western Division of New South Wales as a case study, Dickman *et al*. (2009) recently assessed the risk of fox and cat predation to extant threatened species and concluded that reintroducing dingoes into the area would have positive effects for most of the threatened vertebrates there, aiding their recovery through trophic cascade effects. However, they did not formally assess the risk of dingo predation to the same threatened species.

Aims. To assess the risk of dingo predation to the extant and locally extinct threatened vertebrates of western New South Wales using methods amenable to comparison with Dickman *et al*. (2009).

Methods. The predation-risk assessment method used in Dickman *et al*. (2009) for foxes and cats was applied here to dingoes, with minor modification to accommodate the dietary differences of dingoes. This method is based on six independent biological attributes, primarily reflective of potential vulnerability characteristics of the prey. Individual-attribute scores were used to derive an overall risk score.

Key results. Up to 75 (94%) of the 80 extant species were predicted to be at risk of dingo predation (71% at high risk) regardless of any effect dingoes might have on foxes or cats. Up to 17 of the 21 (81%) locally extinct species were predicted to be at high risk of dingo predation using this approach. The re-establishment of even low-density dingo populations may have negative effects on at least 22% of extant threatened vertebrates.

Conclusions. The generic risk-assessment method was insensitive, and experienced difficulty in describing the true nature of canid predation risk. Despite this weakness, however, it is clear that several threatened vertebrates are susceptible to dingo predation. Prior to the re-establishment of dingoes, we recommend that dingo predation risks to all vertebrates (threatened or otherwise) be assessed using more sensitive and descriptive techniques, and we strongly caution against the positive management of dingoes under current ecological conditions.

Implications. The results of this study imply that dingoes present similar levels of direct risk to threatened species as foxes and feral cats, and dingo predation of threatened species should be formally considered in any proposal encouraging dingo populations in western New South Wales.

Introduction

The biodiversity benefits of maintaining or restoring populations of apex mammalian predators has been the subject of much discussion (Ray *et al*. 2005; Hayward and Somers 2009). Anthropogenic removal of apex predators is perceived to cause the ascent of mesopredator populations (Prugh *et al*. 2009) and consequent negative changes in faunal and floral biodiversity (Roemer *et al*. 2009; Estes *et al*. 2011) through a

process termed 'mesopredator release' (Crooks and Soulé 1999). This has led to the restoration of grey wolves (*Canis lupus lupus*), brown bears (*Ursus arctos*) and lynx (*Lynx lynx*) into parts of Europe and North America (Merrill and Mech 2000; Trouwborst 2010), and lions (*Panthera leo*), leopards (*P. pardus*) and other predators in southern Africa (Hayward and Somers 2009). Repopulation of large predators has been facilitated by active reintroductions, dispersal following reintroductions, and natural

recolonisation associated with reduced human depredation (Hayward and Somers 2009; Randi 2011).

Recently, the potential consequences of mesopredator release on predator–prey interactions in Australia have also been raised (e.g. Robley *et al.* 2004; Glen and Dickman 2005; Glen *et al.* 2007*a*). These reviews have indicated that dingoes (*Canis lupus dingo* and other free-roaming *Canis* species) are a strongly interactive species, and although formerly exotic mesopredators themselves (Johnson 2006; Fleming *et al.*, in press), they have assumed apex predator status since the mainland extinction of the larger thylacine (*Thylacinus cynocephalus*) which occurred within a few hundred years after dingoes were introduced (Corbett 2001*b*; Johnson and Wroe 2003). Dingoes exert this status over other alien mesopredators, such as foxes (*Vulpes vulpes*) and feral cats (*Felis catus*), which were later introduced to Australia by Europeans (Rolls 1969; Abbott 2002). Applying trophic-cascade and mesopredator-release theory to an Australian context, the general premise is that dingoes will exclude, limit, regulate or suppress foxes and/or cats and thereby protect faunal biodiversity from fox and/or cat predation (e.g. Letnic *et al.* 2009*a*; Wallach *et al.* 2009*a*; Carwardine *et al.* 2011). If this holds true, then the positive management of dingoes might provide indirect benefits to biodiversity. At present, however, multiple methodological shortcomings mean that only weak inferences (*censu* Platt 1964; MacKenzie *et al.* 2006) can be drawn from most studies claiming support for the beneficial roles of dingoes to native fauna (see Allen (2011), Allen *et al.* (2011*b*), but see also Letnic *et al.* (2011*a*), Allen *et al.* (2011*a*)). Because 'weak inference mistaken for strong inference can be ruinously dangerous' (Caughley and Sinclair 1994: p. 230), more careful consideration of the ecological niches of dingoes and other free-roaming dogs in Australia is required before their positive management (Fleming *et al.*, in press).

Predation of Australia's vertebrate fauna by introduced predators has been a commonly identified cause of species declines (e.g. Smith *et al.* 1994; Dickman 1996; Smith and Quin 1996; McKenzie *et al.* 2007). Threat-abatement plans for prevention of extinction by introduced predators have been developed (e.g. DEWHA 2008*a*, 2008*b*). However, threat-mitigation actions also require assessment of *their* relative risks before implementation (IUCNSSC 1995). Given the pervasiveness of threats from introduced predators and the limited resources available to manage threatened species, various rank-scoring models have been developed to identify those species at greatest risk from predation by terrestrial predators (e.g. Mahon 2001; Coutts-Smith *et al.* 2007; Dickman *et al.* 2009). These models rank the likelihood of impact on prey species of concern by considering a set of independent biological attributes common to those taxa and the predator species in question.

In a recent case study for the Western Division of New South Wales, Dickman *et al.* (2009) used a relatively simple set of generic biological attributes to assess the risks of fox and cat predation to extant threatened vertebrates. Approximately 20% of terrestrial vertebrates in the Western Division are listed as threatened (Coutts-Smith *et al.* 2007), and many species were understandably deemed to be at risk of fox and/or cat predation. Assuming dingoes could eliminate these mesopredators,

Dickman *et al.* (2009: p. 258) concluded that reintroducing dingoes into that area would have 'positive effects for most of the threatened vertebrates that currently occur there, and would improve conditions for restoring many others that have become locally extinct'. In total, up to 70 species were predicted to benefit from the suppression of foxes and cats by dingoes, with a further 21 that could be successfully reintroduced along with the dingo (Dickman *et al.* 2009). The direct predation risks of dingoes to these same species were not critically assessed in Dickman *et al.* (2009). However, the authors did indicate that 'four or five large-bodied species' may be eaten by dingoes (e.g. southern hairy-nosed wombats, *Lasiorhinus latifrons*). They postulated that because these four or five species were sympatric with dingoes before European settlement, it was 'unlikely that any impact of dingoes on them would exceed that of foxes and cats under current conditions' (Dickman *et al.* 2009: p. 258).

Although dingoes do prey on large species (e.g. Marsack and Campbell 1990; Thomson 1992), dingoes are also efficient predators of small and medium-sized species, especially in arid areas (e.g. Corbett and Newsome 1987; Pavey *et al.* 2008; Cupples *et al.* 2011; Newsome 2011). Thus, dingoes are known to consume many of the same species they are also assumed to protect. Summarising the prey remains found in 12 802 scats and stomachs from multiple dingo-diet studies conducted in a variety of bioregions across Australia up to 2001, Corbett (2001*b*) reported that 72% of dietary items from all bioregions are mammals, and 80% of those are small–medium-sized (<15 kg). Small–medium-sized mammals could therefore be considered preferred prey in all bioregions except the Fortescue River region of Western Australia, where small–medium-sized prey were largely unavailable (see Thomson 1992). Of the smallest species eaten, insects, reptiles and birds were present in dingo diets from most regions (Corbett 2001*b*). In line with the results of Paltridge (2002, *n* = 75 dingo scats), Newsome (2011) showed that larger blue-tongued skinks (*Tiliqua multifasciata*, ~400 g), desert mice (*Pseudomys desertor*, 25 g), and unidentified small skinks were the three single most frequently occurring live prey in dingo diets from 1907 dingo scats collected over 2 years from the Tanami Desert (23%, 17% and 14% occurrence respectively), with dingoes considered to present significant risks to local populations of these species. All major dingo-diet studies from central Australia therefore report rodents and reptiles (and rabbits, *Oryctolagus cuniculus*) to be the most important dingo prey. Dingoes, foxes and cats therefore exhibit considerable dietary overlap in their shared preference for small prey species (Paltridge 2002; Glen and Dickman 2008; Pavey *et al.* 2008; Letnic *et al.* 2009*b*; Cupples *et al.* 2011). This suggests that the faunal biodiversity outcomes of increased dingo population densities in the Western Division of New South Wales will depend on the strength of interactions between dingoes, mesopredators and the prey they share (Holt and Lawton 1994). If the direct effect of dingoes on prey is greater than their indirect ability to reduce mesopredator predation on the same prey species, then encouraging dingoes might not yield the faunal biodiversity benefits often anticipated.

Dingoes have been implicated in the extinctions of native vertebrates before European settlement (Archer 1974; Baird 1991; Johnson 2006) and the loss of other native vertebrates in the recent past (e.g. Kerle *et al.* 1992; Moseby *et al.* 1998; Corbett

2001*b*; Horsup 2004; Allen 2011). Predation by dingoes and other wild-living dogs is identified as a known or potential threat in no less than 14 national threatened-species recovery plans listed by the Australian government (www.environment.gov.au, verified January 2011) for species weighing as little as 70 g (i.e. marsupial moles, *Notoryctes* spp., Benshemesh 2004). 'Predation and hybridisation by feral dogs (*Canis lupus familiaris*)' is also a listed Key Threatening Process for 'threatened species, populations, and communities' in New South Wales (see Major (2009), for the listing; see Corbett (2001*a*), Stephens (2011), for the distribution of *Canis lupus* subspecies in Australia; and see Allen *et al.* (2011*a*), Corbett (2001*b*, 2008); Claridge and Hunt (2008), Fleming *et al.* (in press), for discussion of taxonomy and functional similarities between wild-living subspecies of *Canis lupus*). Surplus killing by dingoes (and probably other predators as well) was also responsible for the failed reintroduction of burrowing bettongs (*Bettongia lesueur*) in northern South Australia in 2008 (Moseby *et al.* 2011). Dingoes also threaten northern hairy-nosed wombats (*L. krefftii*, Banks *et al.* 2003; Horsup 2004), bridled nailtail wallabies (*Onychogalea fraenata*, Lundie-Jenkins and Lowry 2005; Augusteyn 2010) and a range of other species (Newsome *et al.* 1997; Coutts-Smith *et al.* 2007) in other areas, where it is predicted that some populations (e.g. Lunney *et al.* 2007; Mifsud 2011) will persist only through the control or absence of canid predators, including dingoes.

Maintaining impartial objectivity in discussions of the usefulness of dingoes as biodiversity conservation tools is an important challenge for a debate which is presently highly skewed towards a favourable view of the dingo's effects on faunal biodiversity (Coman and Jones 2007; Allen *et al.* 2011*b*). In consideration of the need for balance, and as a guide to the potential direct impacts of dingoes on threatened species, we therefore provide a risk assessment of dingo predation to the locally extinct and extant threatened vertebrates of the Western Division of New South Wales as a supplement to Dickman *et al.* (2009). To enable comparison, we strictly adhere to the same methods as used by Dickman *et al.* (2009) for foxes, with minor modifications to accommodate dietary differences between dingoes and foxes. We discuss the predicted effects of dingo re-establishment on key threatened species as examples.

Materials and methods

We assessed the risk of dingo predation to the 80 vertebrate species that are listed as *Vulnerable* and *Endangered* in the Western Division of New South Wales under the *Threatened Species Conservation Act 1995* (Dickman *et al.* 2009: table 11.2). There were confusing listings under the *Act* for some reptiles. The whip snake, *Demansia rimicola*, was listed as *D. torquata* in New South Wales, the slider *Lerista aericeps* was listed as *L. xanthura*, the distribution of the long-legged worm-skink, *Anomalopus mackayi*, probably falls outside the western Division, and the murray river turtle, *Emydura macquarii macquarii*, was listed simply as *E. macquarii*, which includes other subspecies that either do not occur in the Western Division or are not threatened there. Every effort was made to identify the exact species concerned; however, readers are encouraged to investigate specific listings further where multiple species and

subspecies names may be in common use. To allow ease of comparison, we retained the lists and taxonomic names used by Dickman *et al.* (2009) in our study. We also assessed the dingo predation risks to an additional 21 locally extinct mammal and bird species that were considered to have reintroduction potential after the restoration of dingoes (referred to by Dickman *et al.* (2009), but sourced from Dickman *et al.* (1993) and Smith *et al.* (1994)). Threatened amphibians were not considered because there is little data on their occurrence in the study region or in dingo diets (Corbett 2001*b*; Dickman *et al.* 2009). As in Dickman *et al.* (2009), risk was limited to the likelihood of predator impacts on prey populations, not individuals. Readers should refer to Dickman *et al.* (2009) for full details of the methods used, and the rationale and justification for this approach.

We used six risk categories associated with the independent biological attributes of predator density, and prey bodyweight, behaviour, mobility, fecundity and habitat requirements (table 11.2 in Dickman *et al.* 2009). We scored each attribute from 0 to 3 (except bodyweight, which was scored from 1 to 3), with 0 indicating no or negligible susceptibility to predation, and 1, 2 and 3 indicating low, medium and high susceptibility respectively. When assessing prey bodyweights, Dickman *et al.* (2009) imposed an upper weight limit (but not a lower one) for species outside the range of fox predation, assuming that predation risk from foxes increases with decreasing body size. Our analyses differed from this, in that we reversed the bodyweight scoring because we assumed that predation risk from dingoes likely decreases with decreasing prey body size (Letnic *et al.* 2009*b*). We also removed the upper weight limit because adults of none of the prey species assessed are too large for dingoes to prey on (Corbett 2001*b*). Hence, scores of 1, 2 and 3 represented prey bodyweight classes of 0–35 g, 35–5500 g and >5500 g, respectively, for mammals (on the basis of Burbidge and McKenzie 1989; Johnson and Isaac 2009), 0–100 g, 100–1000 g and >1000 g, respectively, for birds (on the basis of Owens and Bennett 2000) and 0–100 g, 100–350 g and >350 g, respectively, for reptiles (on the basis of the reptile families identified in dingo diets across Australia; Corbett 2001*b*; Paltridge 2002; Newsome 2011). Dickman *et al.* (2009) assigned slightly different attribute scores for foxes and cats, so we used their fox-attribute scores for prey behaviour, mobility, fecundity and habitat in our risk assessment because dingoes are more similar to foxes than to cats. Thus, the only prey-related attribute scores in our study that differ from table 11.2 in Dickman *et al.* (2009) are those for bodyweights.

By using the identical remaining attribute scores for dingoes, we assumed that if a given habitat restricted access to prey for foxes, it also restricted access to dingoes, which are larger. Similarly, if the threatened species had a particular behaviour that reduced its susceptibility to predation by foxes, that anti-predator behaviour equally reduced its susceptibility to predation by dingoes. Following Dickman *et al.* (2009), we considered predator density and the prey attributes of bodyweight, habitat use and behaviour to be most likely to predispose native fauna to predation risk, and we used these four attributes to assign overall risk values for each of the species assessed. We considered species to be at no risk if they received one or more scores of 0 in any of these four categories, at low risk if they received one score of 3 and non-zero scores in all remaining categories, and

at high risk if they received two or more scores of 3 in any of the four categories. We also scored the mobility and fecundity attributes identically to Dickman *et al*. (2009), which were used to help identify species at low risk only. Absence of scores of 0 or 3 in any category was indicative of no overall risk. Where conflicting scores of both 0 and 3 were found (suggesting that the species was either at no risk or at high risk, depending on which biological attribute we considered to be most important in predisposing the species to dingo predation), we favoured the score of 0 to avoid overstating the risk of dingo predation. Several species had been assigned a range of scores for a given attribute, producing some uncertainty about the assignment of overall risk to a species. In our assessment, we adopted and applied the same precautionary principle that Dickman *et al*. (2009) used to address this when assigning overall risk, and used the higher score. The overall risk of dingo predation to each threatened species was therefore assessed for three possible levels of dingo density (low = 1, medium = 2 or high = 3).

Assessing risk using the methods described did produce some conflicting results, which we discuss later, but these were retained to provide equivalence with Dickman *et al*. (2009). Dickman *et al*. (2009) experienced similar conflicts in their analysis, and we likewise reviewed species' susceptibility to dingo predation and suggest that *post hoc* amendments to the overall risk scores for several species may be warranted. Unless otherwise noted, information on mammal distributions were taken from Wilson *et al*. (1992), Van Dyck and Strahan (2008) and West (2008).

Results

Sixteen extant mammals, 41 extant birds and 23 extant reptiles in the Western Division were assessed for potential susceptibility to dingo predation (Table 1). A further 17 locally extinct mammals and four locally extinct birds were also assessed (Table 2). As expected, the potential risk of dingo predation to Western Division vertebrates varied with dingo density. Low and medium dingo-density scores produced identical corresponding overall risk scores in all cases, so these results were pooled for clarity (Tables 1–3).

For extant species exposed to the presence of low and/or medium dingo densities, the risk-analysis approach identified 7 of the 16 mammals, 18 of the 41 birds and 10 of the 23 reptiles to be at low risk of dingo predation (Table 3). Four mammals, six birds and eight reptiles were at high risk of dingo predation in the presence of low/medium dingo densities, suggesting that the re-establishment of dingoes at any level would have negative effects on at least 22% of the threatened vertebrate species in the Western Division. Only five mammals, 17 birds and five reptiles (34% of all listed species) were identified as having no risk from dingo predation in the presence of low/medium dingo densities (Table 3).

For extant species exposed to the presence of high dingo densities, two of the mammal species, 13 of the birds and three of the reptiles were at low risk of dingo predation (Table 3). In total, 14 mammals, 24 birds and 19 reptiles were at high risk of dingo predation in the presence of high dingo densities. This suggested that if dingo populations reached high densities in the Western Division, they may have serious negative effects on 57 (71%) of the 80 threatened vertebrate species. Only four birds and one

reptile were identified as having no risk of dingo predation in the presence of high dingo densities (Table 3).

For reintroductions of the 21 locally extinct species into areas with low/medium dingo densities, six mammals and one bird would face a low risk of dingo predation (Table 3). Nine mammals and one bird would be at high risk of dingo predation in the presence of low/medium dingo densities, suggesting that reintroduction efforts would be unsuccessful for nearly half of the species considered. Only four locally extinct species considered for reintroduction would be at no risk of dingo predation in the presence of low/medium dingo densities (Table 3). For reintroductions of locally extinct species into areas with high dingo densities, only two mammals (*Antechinus flavipes* and *Phascogale calura*) would be free of risk of dingo predation, two birds would face a low risk, and the remaining 17 species would each be at high risk of dingo predation (Table 3). This suggests that reintroduction efforts could be unsuccessful for 81% of species considered for reintroduction.

Discussion

Risk-assessment approach

Several weaknesses were evident in the risk-assessment approach applied here and in Dickman *et al*. (2009). For example, low and medium overall risk scores were identical in all cases because attribute scores of 0 or 3 largely determined the overall risk scores. This suggests that the scoring system lacks sensitivity because it could not distinguish between low and medium overall risks. We retained this scoring system to provide equivalence with Dickman *et al*. (2009), and therefore present the overall risk scores without amendment for both a 'low/medium dingo density' scenario and a 'high dingo density' scenario (columns A and B, respectively, in Tables 1 and 2).

Conflicting results were obtained for some species. For example, each primary attribute used to assess black-breasted buzzards (*Hamirostra melanosternon*) indicated that they were at high risk of dingo predation, except (of course) their behaviour score, which was 0. Hence, buzzards could be classified as either high risk or no risk, depending on whether or not the attributes bodyweight and habitat were deemed more important than behaviour. Because we followed Dickman *et al*. (2009) in assuming that a score of 0 was more important than any score/s of 3, regardless of the attribute, the model therefore determined buzzards to be at no overall risk of dingo predation. This approach reflected the minor likelihood of a dingo actually preying on a buzzard (let alone threatening their population), and we therefore considered it appropriate. However, this approach is unlikely to reflect reality for all such instances, such as the red-tailed black cockatoo (*Calyptorhynchus banksii*), which was deemed to be at high risk of both dingo (our study) and fox (Dickman *et al.* 2009) predation.

Dickman *et al*. (2009) found similar issues for two owl species, subsequently downgrading them to no risk of fox predation, despite the scores suggesting that they were at low risk. However, we did not attempt *post hoc* amendments to the overall risk scores because of the multiple ways these could have been derived. For example, a particular attribute with a score of 0 may not always be considered more important than

Table 1.   Factors contributing to the susceptibility of threatened mammals, birds and reptiles to dingo predation

A = low and/or medium dingo-density scenario, B = high dingo-density scenario. Species are threatened species as listed in table 11.2 of Dickman *et al.* (2009). See text for explanation of independent biological attributes and an overall risk of predation

Species name	Common name	Independent biological attributes					Overall risk of predation	
		Bodyweight	Habitat	Behaviour	Mobility	Fecundity	A	B
Mammals – vulnerable								
Dasyurus maculatus	Spotted tail quoll	2	2	2	1	1	N	L
Ningaui yvonneae	Southern ningaui	1	2	3	0	1	N	H
Phascogale tapoatafa	Brush-tailed phascogale	2	2	1	1	1	N	L
Sminthopsis macroura	Stripe-faced dunnart	1	3	3	0	0-1	H	H
Phascolarctos cinereus	Koala	3	2	1	1	2	L	H
Leggadina forresti	Central short-tailed mouse	1	3	3	1	1	H	H
Pseudomys hermannsburgensis	Sandy inland mouse	1	2	3	1	0-1	L	H
Rattus villosissimus	Long-haired rat	2	2	3	2	0	N	H
Mammals – endangered								
Antechinomys laniger	Kultarr	1	3	3	0	0-1	H	H
Lasiorhinus latifrons	Southern hairy-nosed wombat	3	2	3	2	3	H	H
Cercartetus concinnus	Western pygmy possum	1	2	3	2	0	N	H
Petrogale penicillata	Brush-tailed rock-wallaby	3	1	2	2	2	L	H
Petrogale xanthopus	Yellow-footed rock-wallaby	3	1	2	2	2	L	H
Notomys fuscus	Dusky hopping-mouse	1	2	3	2	0-1	L	H
Pseudomys bolami	Bolam's mouse	1	2	3	1	0-1	L	H
Pseudomys glaucus	Blue-gray mouse	1	2	3	1	0-1	L	H
Birds – vulnerable								
Anseranus semipalmata	Magpie goose	3	1	1	2	1	L	H
Oxyura australis	Blue-billed duck	2	0	2	1	1	N	N
Stictonetta naevosa	Freckled duck	2	1	2	2	0-1	N	L
Botaurus poiciloptilus	Australiasian bittern	3	1	1-2	1	1	L	H
Lophoictinia isura	Square-tailed kite	2	2	0	0	1-2	N	N
Hamirostra melanosternon	Black-breasted buzzard	3	3	0	0	2	N	N
Falco hypoleucos	Grey falcon	2	3	0	0	1-2	N	N
Grus rubicunda	Brolga	3	1	0-1	1-2	2	L	H
Limosa limosa	Black-tailed godwit	2	1	2	1-2	1	N	L
Calyptorhynchus banksii	Red-tailed black cockatoo	2	2-3	1	1	2-3	H	H
Calyptorhynchus lathami	Glossy black cockatoo	2	1-2	0-1	2	2-3	L	H
Cacatua leadbeateri	Major mitchell's cockatoo	2	2-3	1	1-2	1	L	H
Glossopsitta porphyrocephala	Purple-crowned lorikeet	1	2	1	2	1	N	L
Polytelis swainsonii	Superb parrot	2	2	1-2	1	1	N	L
Neophema pulchella	Turquoise parrot	1	3	2	2	1	L	H
Neophema splendida	Scarlet-chested parrot	1	3	2	2	1	L	H
Ninox connivens	Barking owl	2	1-2	1	1	1-2	N	L
Tyto capensis	Grass owl	2	2-3	2-3	1	1	H	H
Tyto novaehollandiae	Masked owl	3	2	1	1	1-2	L	H
Amytornis striatus	Striated grasswren	1	2	2	1	2	N	L
Hylacola cauta	Shy heathwren	1	2	2	1	1	N	L
Pyrrholaemus brunneus	Redthroat	1	2	2	1	1	N	L
Chthonicola sagittata	Speckled warbler	1	2-3	2	1	1	L	H
Melithreptus gularis	Black-chinned honey eater	1	2	1	1-2	2	N	L
Grantiella picta	Painted honeyeater	1	2	1	1-2	1-2	N	L
Certhionyx variegatus	Pied honeyeater	1	3	1	1-2	1-2	L	H
Drymodes brunneopygia	Southern scrub robin	1	2	2	1-2	2-3	L	H
Pomatostomus halli	Hall's babbler	1	2-3	1	2	1-2	L	H
Cinclosoma castanotus	Chestnut quail-thrush	1	2	2	1	2	N	L
Pachycephala inornata	Gilbert's whistler	1	2	1-2	1	1	N	L
Stagonopleura guttata	Diamond firetail	1	2-3	2	1-2	1	L	H
Birds – endangered								
Leipoa ocellata	Malleefowl	3	2-3	2	1	0	H	H
Ardeotis australis	Bustard	3	3	2-3	1-2	2	H	H
Pedionomus torquatus	Plains wanderer	1	3	2	1	1	L	H
Rostratula benghalensis	Painted snipe	2	1	2-3	2-3	1	H	H
Burhinus grallarius	Bush stone-curlew	2	3	3	1-2	2	H	H

(*continued next page*)

Table 1. (*continued*)

Species name	Common name	Independent biological attributes					Overall risk of predation	
		Bodyweight	Habitat	Behaviour	Mobility	Fecundity	A	B
Phaps histrionica	Flock bronzewing	2	2-3	1-2	2	2	L	H
Polytelis anthopeplus	Regent parrot	2	2-3	1-2	1-2	1	L	H
Amytornis barbatus	Grey grasswren	1	2-3	2	2	2	L	H
Manorina melanotis	Black-eared miner	1	2	1-2	2	1	N	L
Pachycephala rufogularis	Red-lored whistler	1	2-3	1-2	1	1-2	L	H
Reptiles – vulnerable								
Emydura macquarii	Murray river turtle	3	2	1-2	1	0	N	H
Antaresia stimsoni	Stimson's python	2	2	2	1	0	N	N
Aspidites ramsayi	Woma python	3	2-3	2	1	0	H	H
Brachyurophis fasciolata	Narrow-banded shovel-nosed snake	1	2-3	3	1	1-2	H	H
Demansia torquata	Collared whipsnake	1	2-3	1	1	1	L	H
Hoplocephalus bitorquatus	Pale-headed snake	2	2	1	1	1	N	L
Ctenotus brooksi	Striped skink	1	2-3	1	1	1	L	H
Lerista xanthura	Yellow-tailed plain slider	1	3	2-3	1	2	H	H
Tiliqua multifasciata	Centralian blue-tongue	3	2-3	2	1	0-1	H	H
Tiliqua occipitalis	Western blue-tongue	3	2	2	1	0-1	L	H
Diplodactylus stenodactylus	Crowned gecko	1	2-3	2	1	2	L	H
Strophurus elderi	Jewelled gecko	1	2-3	2	1	2	L	H
Reptiles – endangered								
Echiopsis curta	Bardick	2	2	1-2	1	0-1	N	L
Pseudonaja modesta	Ringed brown snake	2	2	1-2	1	0-1	N	L
Ramphotyphlops endoterus	Interior blind snake	1	3	3	1	1	H	H
Aprasia inaurita	Red-tailed worm-lizard	1	2-3	2-3	1	2	H	H
Delma australis	Marble-faced delma	1	2-3	2-3	1	2	H	H
Anomalopus mackayi	Long-legged worm-skink	1	3	2-3	1	1	H	H
Ctenotus pantherinus	Leopard ctenotus	1	2-3	2	1	1	L	H
Cyclodomorphus melanops	Spinifex slender blue-tongue	1	2-3	1-2	1	1-2	L	H
Cyclodomorphus venustus	Saltbush slender blue-tongue	1	3	1-2	1	1-2	L	H
Ctenophorus decresii	Tawny crevice-dragon	1	3	2	1	1	L	H
Diplodactylus conspicillatus	Fat-tailed gecko	1	2-3	2	1	2	L	H

one with a score of 3 for each species or population, and altering the bodyweight classifications also changes the number of species deemed to be at risk of predation. Identifying the most appropriate weight classifications was somewhat arbitrary, but it should be noted (and perhaps obvious) that increasing the lower limits of each body-size class will effectively shift species towards lower overall-risk categories. Although this might easily be performed to overcome some of the problems associated with body-size classes (i.e. very small species determined to be at high risk of dingo predation), it creates additional problems by downplaying the role of other potentially important risk factors, such as habitat and behaviour. Given the multitude of conflicting definitions for small, medium or large prey specific to dingoes (e.g. compare Corbett (2001b) with Cupples *et al*. (2011) and/or Letnic *et al*. (2009b)), we used the well known 'critical weight range' limits reported by Burbidge and McKenzie (1989) for mammals (as did Dickman *et al*. 2009), and those of Owens and Bennett (2000) for birds, although we recognise that these too may not be entirely appropriate in each case. In addition to the issues associated with the scoring system and the bodyweight classes, the overall-risk scores also change significantly if the precautionary principle is not applied, which may not be warranted for each species or population. In any event, further

alteration of the methods to reduce the number of species deemed to be at some level of risk to dingoes cannot escape the conclusion (discussed below) that at least *some* species are indeed at risk of dingo predation should they be reintroduced, and these risks should not be ignored.

Given the weaknesses associated with the approach, we therefore believe that a downgrading of overall risk is warranted for several species, although we encourage readers to discern for themselves which species are reliably described by this method. In passing, however, we believe that many species would receive lower risk scores if risk were determined individually using a more sensitive and descriptive approach, which will similarly apply to the risk of fox and cat predation calculated by Dickman *et al*. (2009). This is not to say that dingoes are unable to capture and kill individuals of these species, but at the population level, the predation risks of many of them may be lower than predicted here. Conversely, populations of some threatened species may be so vulnerable at present that predation by *any* predator may not be sustainable, and dingoes could still present a high risk to populations of some species considered to be at low or no risk using this approach.

The method applied here is also largely based on potential vulnerability characteristics of the prey species. Such an approach

Table 2. Factors contributing to the susceptibility of locally extinct mammals and birds to dingo predation

A = low and/or medium dingo-density scenario, B = high dingo-density scenario. Species referred to in Dickman *et al*. (2009) but sourced from Dickman *et al*. (1993) and Smith *et al*. (1994). See text for explanation of independent biological attributes and overall predation risk scores

Species name	Common name	Independent biological attributes					Overall risk of predation	
		Bodyweight	Habitat	Behaviour	Mobility	Fecundity	A	B
Mammals – reintroduction potential								
Antechinus flavipes	Yellow-footed antechinus	2	1	2	1	0	N	N
Dasyurus geoffroii	Western quoll/chuditch	2	2	3	1	1	L	H
Phascogale calura	Red-tailed phascogale	2	2	2	1	0	N	N
Myrmecobius fasciatus	Numbat	2	3	3	1	2	H	H
Isoodon auratus	Golden bandicoot	2	3	3	2	2	H	H
Perameles bougainville	Western barred bandicoot	2	3	3	2	2	H	H
Macrotis lagotis	Greater bilby	2	3	3	1	1	H	H
Lasiorhinus kreftii	Northern hairy-nosed wombat	3	3	3	2	3	H	H
Bettongia lesueur	Burrowing bettong/boodie	2	3	3	2	2	H	H
Bettongia penicillata	Brush-tailed bettong/woylie	2	2	3	1	2	L	H
Bettongia tropica	Northern bettong	2	1	3	2	2	L	H
Onychogalea fraenata	Bridled nailtail wallaby	3	2	3	2	2	H	H
Leporillus conditor	Greater stick-nest rat	2	3	1	3	2	L	H
Notomys mitchellii	Mitchell's hopping-mouse	2	3	3	2	1	H	H
Pseudomys australis	Plains mouse	2	3	3	3	1	H	H
Pseudomys desertor	Desert mouse	1	2	3	3	1	L	H
Rattus tunneyi	Pale field rat	2	2	3	2	1	L	H
Birds – reintroduction potential								
Xanthomyza phrygia	Regent honeyeater	1	1	1	1	1	N	L
Amytornis textilis	Thick-billed grasswren	1	3	1	1	1	L	H
Pezoporus occidentalis	Night parrot	1	3	3	1	1	H	H
Ixobrychus flavicollis	Black bittern	2	2	2	1	2	N	L

Table 3. Summary of overall risks of dingo predation to extant and locally extinct mammals, birds and reptiles

Category	Low and/or medium dingo density			High dingo density		
	No risk	Low risk	High risk	No risk	Low risk	High risk
Extant mammals (n = 16)						
Vulnerable	4	2	2	0	2	6
Endangered	1	5	2	0	0	8
Total	5	7	4	0	2	14
Extant birds (n = 41)						
Vulnerable	16	13	2	4	12	15
Endangered	1	5	4	0	1	9
Total	17	18	6	4	13	24
Reptiles (n = 23)						
Vulnerable	3	5	4	1	1	10
Endangered	2	5	4	0	2	9
Total	5	10	8	1	3	19
Locally extinct mammals (n = 17)						
Total	2	6	9	2	0	15
Locally extinct birds (n = 4)						
Total	2	1	1	0	2	2

may not be satisfactory for a dingo-reintroduction scenario where the floral and faunal communities in post-European ecosystems have been grossly altered to include abundant artificial water and preferred prey resources (Landsberg *et al*. 1997; West 2008), and where threatened prey species have not been exposed to dingo predation for many decades. Where dingoes have been introduced to systems previously free of their impacts before, novel and unexpected food items (e.g. crustaceans, fruit, and sugar gliders, *Petaurus breviceps*) appear in dingo diets following the dingo-induced decline of preferred prey (Allen *et al*. 1998; Allen and Gonzalez 2000). Hence, we should not be solely concerned with whether or not dingoes *do* pose a risk at present, but also whether or not dingoes *could* pose a risk under future ecological conditions (e.g. Newsome *et al*. 1997). Naive prey species are particularly susceptible to reintroduced predator populations, and extinction risk is heightened where individuals of all age classes are preyed on (Berger *et al*. 2001; Gittleman and Gompper 2001). Consideration of this should be important to dingo-reintroduction proposals because adults of no native terrestrial prey species are too large for a solitary dingo to successfully hunt and kill (Marsack and Campbell 1990; Corbett 2001b; Purcell 2010). Moreover, individual species with apparently similar life histories also respond differently to predation effects (Boutin 1995; Sinclair *et al*. 1998; Moseby *et al*. 2009; Read and Cunningham 2010), which may be better incorporated into a risk-assessment approach that focuses on individual species or populations. In contrast to a prey-based approach (such as that applied here), a risk-assessment technique that evaluated the capabilities of the predator and the probability of predator–prey encounters may produce results that are more applicable to a predator-reintroduction situation. Such an approach may consider knowledge of plasticity in dingo diets, individual and group hunting strategies, foraging patterns, and their propensity

to identify, utilise, exploit and persist on unfamiliar or less-preferred prey species.

Extant species

The overall risks of dingo predation to extant species were inconsistent. This is unsurprising, given the large variation among taxa and the range of processes affecting their decline in the Western Division (Dickman *et al*. 1993; Smith *et al*. 1994; Lunney 2001; McKenzie *et al*. 2007). Reviewing the overall risk scores suggested that mammals and reptiles were at greater risk of dingo predation than were birds, corroborating studies that show mammals occurring most frequently in dingo diets across Australia (Corbett 2001*b*). The elevated risks to reptiles are somewhat puzzling (discussed below), but may reflect dingoes' apparent preference for reptiles in arid areas (Paltridge 2002; Newsome 2011). Regardless, multiple mammal, bird and reptile species were determined to be at high risk of dingo predation, although the methodological constraints highlighted earlier cast doubt on some of the overall risk scores in light of alternative information.

Mammals

Rock-wallabies (*Petrogale* spp.) are naturally confined to rugged refuges used to escape predation (Tuft *et al*. 2011). However, suitable refuge alone has not prevented their decline across Australia because introduced predators are capable of penetrating these refuges (Kinnear *et al*. 2010; Ward *et al*. 2011). Moreover, isolation and fragmentation render populations susceptible to stochastic events (such as disease or wild fire) that may further aid their decline (Fahrig and Merriam 1994; May and Norton 1996; Lindenmayer *et al*. 1999). Yellow-footed rock-wallabies (*P. xanthopus*) predominantly exist in areas where dingoes are in human-induced low densities and where intensive dingo and fox control is routinely conducted (e.g. DEH 2006). Although Wallach *et al*. (2009*a*) recently claimed that dingoes indirectly aid the recovery of *P. Xanthopus* by suppressing mesopredators, such conclusions were beyond the limitations of the data (Allen 2010*a*; Allen *et al*. 2011*b*). Newsome *et al*. (1997) warned of the risk dingoes posed to isolated populations of black-footed rock wallabies (*P. lateralis*) should rabbits become unavailable to dingoes, and Moseby *et al*. (1998) sadly reported the local extinction of an isolated rock-wallaby population when this situation was realised. Read and Ward (2011), therefore, reiterate the risk dingoes (and other predators) currently pose to remaining populations of *P. lateralis*.

Intensive predator-control activities (using 1080 baiting techniques lethal to both dingoes and foxes alike) have sometimes facilitated a rapid and sustained increase of threatened rock-wallaby populations (e.g. Gibson 2000; DEH 2006; Kinnear *et al*. 2010), whereas no studies have shown negative effects to populations of any non-target species as a result of canid control (e.g. Fenner *et al*. (2009), but see Glen *et al*. (2007*b*) and APVMA (2008)). Although the distributions of many other rock-wallaby species occur within the extended range of dingoes (e.g. *P. penicillata inornata*), population growth and dispersal may still be limited by dingoes (Tuft *et al*. 2011; Ward *et al*. 2011), which may confine rock-wallabies to suitable refuge habitat so as to avoid

excessive dingo predation. The risk-assessment technique used here indicates that *P. xanthopus* and *P. penicillata penicillata* are both at risk of dingo predation (Table 1). It seems reasonable to assume that interactions between dingoes and rock-wallabies over the past 5000 years have restricted rock-wallabies to an ecological niche that proffers some protection from dingo predation. The use of rock outcrops as refuge may have been a pre-adaptive behavioural response that evolved to avoid dingo predation, a behaviour still used today to avoid fox predation (Kinnear *et al*. 2010; Ward *et al*. 2011). For these reasons, the reintroduction of dingoes to the Western Division may simply maintain predation pressure (and their current status) on the extant rock-wallaby populations there, which are unlikely to be bolstered without persistent (and costly) predator-control interventions. The intensive control of dingoes and other wild dogs is also viewed as an essential management action to prevent the decline of koalas (*Phascolarctos cinereus*; see Lunney *et al*. (2007), but also Mifsud (2011)), which are particularly susceptible to dingo predation in concert with other threatening processes.

Although some extant mammal populations are undoubtedly at risk of dingo predation, many are not. Dickman *et al*. (2009) determined the stripe-faced dunnart (*Sminthopsis macroura*) to be at high risk of fox and cat predation, suggesting that reintroducing dingoes would benefit them through dingo suppression of mesopredators. Applied here to dingoes, the same risk-assessment approach equally determined *S. macroura* to be at high risk of dingo predation (Table 1), suggesting that the reintroduction of dingoes would unnecessarily threaten Western Division populations of this species further. Although these risk assessments suggest that dingo, fox and cat predation may be a significant limiting factor for *S. macroura*, a closer look at the distribution of this species outside of the Western Division suggests that these conclusions are unjustified.

Sminthopsis macroura is widespread and abundant across most of Australia, including regions where all three predators are common and in high densities (compare Van Dyck and Strahan (2008) with Wilson *et al*. (1992) and West (2008)). Hence, they are not of conservation concern at a national level, but are so listed in the Western Division. Implicit in the argument to remove mesopredators through dingo reintroduction is that mesopredator predation is a limiting factor for a given threatened species. However, *S. macroura* is apparently common in the presence of high mesopredator densities in places largely devoid of dingoes (see references above). This suggests that the conservation concerns raised by Dickman *et al*. (2009) may not be warranted for *S. macroura* or other mammal species that are widespread and common outside of the Western Division, such as the kultarr (*Antechinomys laniger*) or the central short-tailed mouse (*Leggadina forresti*). The Western Division may be an area where dingoes and these small mammals are rare and mesopredators are abundant. However, although speculating about the causal factors for such a correlation is seductive, comparisons of species distribution patterns have weak inferential ability because a large number of competing (and equally plausible) hypotheses are possible (MacKenzie *et al*. 2006); correlations have no power to describe causation. The absence of dingoes may be one explanation for the rarity of these small mammals in the Western Division, as argued in Dickman

et al. (2009) and Letnic *et al.* (2011*b*), but the cumulative impact of historical livestock grazing may likewise explain or contribute to such observations (Dickman *et al.* 1993; Letnic 2000; Lunney 2001; Letnic *et al.* 2009*a*; Allen 2011).

Birds

In our risk assessment, many birds were determined to be at high risk of dingo predation in the presence of low or high dingo populations, whereas the overall risk of dingo predation to many others appeared dependant on dingo density (Table 1). However, the overall-risk scores produced by the model may not be reliable for many bird species for reasons described earlier for some mammals. Nevertheless, some bird species appear particularly susceptible to dingo predation. For example, the majority of primary-attribute scores for grass owls (*Tyto capensis*), bustards (*Ardeotis australis*) and bush stone-curlews (*Burhinus grallarius*) suggest that they are at high risk of dingo predation. These results are confusing, given the widespread distributions of these species nationally (Barrett *et al.* 2003), which suggest that they are not limited by dingoes. Despite this, the behaviour and nesting habits of *A. australis* and *B. grallarius* in particular may provide opportunities for dingo predation to threaten isolated populations of these two species, and their overall-risk scores should not be dismissed simply because they are widespread outside of the Western Division.

Dingoes can be severe predators of terrestrial nests (e.g. Whiting *et al.* 2007; Somaweera *et al.* 2011), and ground-nesting birds such as malleefowl (*Leipoa ocellata*) and the iconic magpie goose (*Anseranus semipalmata*) may be particularly susceptible to dingo predation (Corbett 2001*b*; Benshemesh 2007). Magpie geese primarily occupy shallow flood plains and reach their highest densities in the monsoonal areas of northern Australia, where ~100 000 birds are harvested by humans every year (Delaney *et al.* 2009). They were historically common in the Western Division but are now only vagrants there (Delaney *et al.* 2009). Cats are common in northern Australia and both foxes and cats were not considered to be significant risk factors for the restoration of magpie geese in the Western Division (Dickman *et al.* 2009). Hence, a reduction of mesopredators through any mechanism (including dingo reintroduction) may be an unnecessary step towards the recovery of this species in the Western Division, where reduced availability of suitable habitat (i.e. flood plains and wetlands, Whalley *et al.* 2011) appears to be the limiting factor (Nye *et al.* 2007; Delaney *et al.* 2009). Moreover, dingoes are capable and efficient predators of magpie geese in northern Australia where they form a primary component of dingo diets (Corbett 2001*b*). Sustained dingo predation of magpie geese in northern Australia is hindered by the annual monsoon flooding that takes place there, but in the Western Division, where surface water is heavily regulated by humans and usually scarce, the risk of dingo predation to *A. semipalmata* may be present at all times.

Reptiles

As far as we are aware, no Australian reptile species have become extinct since European settlement, rendering reptile susceptibility to predation difficult to gauge using the methods described by Dickman *et al.* (2009). In our analysis, reptile species such as the woma python (*Aspidites ramsayi*), blue-tongued skinks (*Tiliqua* spp. and *Cyclodomorphus* spp.) and several very small species were at risk of dingo predation (Table 1). Indeed, 19 of the 23 threatened reptiles were predicted to be at high risk in the presence of high dingo densities (Table 3). However, despite the relatively frequent occurrence of some reptiles in arid dingo diets (e.g Paltridge 2002; Newsome 2011), we believe the majority of these predictions to be unsupported.

Interactions between eutherian predators and reptiles are poorly understood, and few studies (e.g. Risbey *et al.* 2000; Olsson *et al.* 2005; Read and Cunningham 2010) have investigated the effects of predator manipulations on reptile populations (Sutherland *et al.* 2010). Dingoes have been predicted to threaten some reptile populations, but such information is typically sparse and the mechanisms of risk inconsistent. For example, Heard *et al.* (2006) identified canid predation as a potentially significant threat to populations of inland carpet pythons (*Morelia spilota metcalfei*) in Victoria after several radio-tracked individuals were found killed by canids, although they could not specifically determine whether or not the killers were foxes or dingoes. Newsome (2011) recently determined populations of arid-zone skinks to be at high risk of dingo predation in the presence of anthropogenically provided food subsidies (i.e. livestock and rubbish). Others have also reported high levels of dingo predation on freshwater crocodile (*Crocodylus johnstoni*) and turtle nests (Whiting *et al.* 2007; Somaweera *et al.* 2011), identifying dingoes as a significant threat to them (e.g. Anon. 2003). Even though reptiles are common in dingo diets across Australia, they usually feature less frequently than mammals – the availability of which may be a significant determinant of the risk of dingo predation to less-preferred species (Courchamp *et al.* 2000; Corbett 2001*b*; Newsome 2011). Although mammals are clearly preferred by dingoes, some skinks, agamids (e.g. central netted dragons, *Ctenophorus nuchalis*) and smaller varanids appear relatively common in arid-zone dingo diets at times, underscoring the capacity of dingoes to exploit some reptile populations when necessary.

In our judgment, most of the snakes (e.g. *A. stimsoni*, *Pseudonaja modesta* and *Ramphototyphlops endoterus*), geckos (e.g. *Diplodactylus* spp. and *Strophurus elderi*) and small skinks (e.g. *Ctenotus* spp. and *Lerista* spp.) listed in Table 1 are unlikely to be threatened by dingo or fox predation under current conditions, irrespective of the individual risk scores shown here and in Dickman *et al.* (2009). Tentatively, we suggest that the addition of dingoes to Western Division ecosystems will not place most reptile populations at increased risk of dingo predation, despite our results to the contrary. However, if dingo-diet studies are indicative of predation risk (e.g. Newsome 2011), then some dragons (such as *Ctenophorus decresii*) and larger skinks (e.g. *Tiliqua* spp.) may be susceptible to dingo predation in the Western Division as predicted. Reptile species may become more vulnerable to dingo predation should mammalian prey populations become unavailable. However, we are not confident in making reliable predictions about the risk of dingo predation to any reptile

population without a greater understanding of dingo–reptile interactions.

Locally extinct species

Only four locally extinct species were not at high risk of dingo predation in the presence of high dingo densities (Table 3). This result is intuitive, given that dingoes may have been at least partly responsible for their post-grazing predation declines in the first place (Corbett 2001*b*; Allen 2011). Common among most locally extinct mammals is their larger size (i.e. they are within the 'critical weight range', Burbidge and McKenzie 1989), with the smallest being the desert mouse at an average weight of 25 g. The risk of dingo predation to *P. desertor* is comparable to that described earlier for *S. macroura*. Our risk assessment determined populations of *P. desertor* to be at risk of dingo predation, in line with Newsome (2011), although the national distribution of this species and its relatively low frequency in dingo diets from other places (Corbett and Newsome 1987; Pavey *et al*. 2008) would suggest that dingoes do not pose a significant threat to this species at the species level. Risks to other *Pseudomys* species were also assessed, and overall risk scores were consistent for each species within this genus, except for plains mice (*P. australis*, also known as the 'plains rat') which were at greater risk (Table 2). Inspection of the attribute scores for this species indicated that this inconsistency is mainly due to the habitat in which it lives.

The plains mouse has not been recorded in Queensland or New South Wales since 1936 (Moseby 2010). Interestingly, however, the species was recently discovered in a dingo scat from atypical *P. australis* habitat in the Strzelecki Desert, ~400 km east of the nearest known population and just <30 km west of the Western Division border (Allen *et al*. 2011*c*). *P. australis* is a much larger species than are other *Pseudomys* species, and is particularly susceptible to predation by eutherian carnivores (e.g. Read and Cunningham 2010; Moseby 2010). Although the newly discovered population just outside the New South Wales border was in the presence of high dingo densities (which could be viewed as support for the concept that dingoes provide protection from mesopredator predation), the risk of dingo predation to *P. australis* had not been assessed until now. The risk-analysis approach applied here determined *P. australis* to be at high risk of dingo predation irrespective of any effect dingoes might have on foxes or cats, suggesting that reintroduced plains mice may face severe predation risks from dingoes. Unlike smaller *Pseudomys* species that frequently occur in areas with high dingo densities, the distribution of *P. australis* has retracted predominately to those areas in northern South Australia where dingo control is most intensive and their abundance is low (Allen 2012). This suggests that dingoes might suppress and exploit *P. australis* similarly to other arid-zone rodents (e.g. Corbett and Newsome 1987; Pavey *et al*. 2008; Newsome 2011), and that dingo control may benefit *P. australis* as it does some larger-bodied mammals (e.g. Glen *et al*. 2007*b*; Augusteyn 2010). This in no way diminishes the importance of alternative processes threatening *P. australis* and other arid-zone rodents (Lee 1995; Moseby 2010), but merely adds dingo predation to the list of potential threats.

Most species that are extinct in the Western Division are larger dasyurids, bandicoots or small macropods. Although the risk of dingo predation to the yellow-footed antechinus (*A. flavipes*) was not identified, this species is found outside the Western Division in places where dingoes have been extensively controlled and fox and cat abundances are high (cf. Wilson *et al*. 1992; Van Dyck and Strahan 2008; West 2008). As for other species discussed above, this suggests that mesopredator predation may not be a limiting factor for *A. flavipes* in the Western Division, and that the reintroduction of dingoes there would be an unnecessary prerequisite to improve conditions for them. Nevertheless, our overall-risk scores do suggest that once the factors ultimately responsible for the local extinction of *A. flavipes* are resolved, the reintroduction of this species into the Western Division could occur with or without dingoes in the landscape.

From a historical range covering most of Western Australia, Northern Territory, South Australia, and extending into the Western Division of New South Wales, the distribution of golden bandicoots (*Isoodon auratus*) has declined to only a small area in north-western Western Australia (Palmer *et al*. 2003; Carwardine *et al*. 2011). They have therefore become extinct from places with and without foxes (Saunders and McLeod 2007), where pastoralism has not occurred (Hamblin 2001; Allen 2011), and from places where dingo populations have been left largely unaltered (Corbett 2001*b*; Allen 2012). This indicates that top-down suppression of mesopredators by dingoes (if it was occurring) was unable to prevent the decline of *I. auratus* and that dingo reintroduction is not a prerequisite to their recovery either. In contrast, their overall risk scores presented here support the information in Allen (2011) and Palmer *et al*. (2003), suggesting that reintroduced populations of both locally extinct bandicoot species (*I. auratus* and *Perameles bougainville*) are unlikely to withstand dingo predation.

Macropods, especially smaller species, are common prey for dingoes in all areas where they are found together (e.g. Robertshaw and Harden 1985; Corbett and Newsome 1987; Thomson 1992). Bridled nailtail wallabies are currently threatened by dingo predation (Lundie-Jenkins and Lowry 2005; Augusteyn 2010), whereas brush-tailed bettongs/woylies (*Bettongia penicillata*) and burrowing bettongs/boodies (*B. lesueur*) are restricted to islands, fenced predator-proof reserves, or areas where regular and persistent predator control occurs (Van Dyck and Strahan 2008). Indeed, reintroducing *B. lesueur* into areas with low-density dingo populations has been met with rapid failure resulting from immediate predation by dingoes and other predators (Moseby *et al*. 2011). A similar situation also exists for the greater stick-nest rat (*Leporillus conditor*) and numbat (*Myrmecobius fasciatus*). Along with the northern bettong (*B. tropica*, which is of questionable suitability to Western Division ecosystems under any future conditions), it seems unlikely that any locally extinct macropods will be able to withstand even minor amounts of predation from low-density dingo populations if their risk scores are indicative of true overall risk (McCallum *et al*. 1995; Moseby *et al*. 2011). Despite their larger body size, northern hairy-nosed wombats (*L. krefftii*) are also unlikely to benefit from the presence of dingoes, which are a recognised threat to their survival in small populations (Banks *et al*. 2003; Horsup 2004).

Final thoughts

The impacts of foxes and cats on fauna are significant (Denny and Dickman 2010; Saunders *et al*. 2010), and any substantial reduction in their abundance is likely to benefit vertebrate biodiversity in the Western Division of New South Wales. However, because Dickman *et al*. (2009) proposed the reintroduction of dingoes there as a mechanism to achieve mesopredator reductions but did not formally assess the direct risk of dingo predation, we applied their method developed for foxes and cats to assess the risk of dingo predation to the same threatened species. We found this method insensitive in describing the true nature of predation risk to many species, and we therefore recommend that our results be used simply as a starting point for assessing the risk of dingo predation to threatened vertebrates.

Dingo diet data seem particularly important to evaluating the risks that dingoes pose to threatened vertebrates; however, caution should be exercised when using diet data in this way because the presence of a prey species in a predator scat/s may not be indicative of population effects of predation or even predation itself. First, dingoes regularly scavenge a wide variety of species, which means that prey remains found in scats cannot provide reliable evidence for predation. The presence or absence of insect larvae (e.g. maggots) in dingo scats has sometimes been used to determine whether or not an animal has been scavenged or killed (e.g. Glen *et al*. 2011). However, this is unlikely to be a reliable indicator of such behaviour because the presence of maggots may occur where dingoes return to scavenge the carcass of an animal they had killed earlier, and the absence of maggots does not necessarily indicate consumption of live prey or preclude consumption of carrion. For example, Allen (2010*b*) observed dingoes scavenging fresh dingo remains and the old and brittle carcasses of cattle (*Bos taurus*), wedge-tailed eagles (*Aquila audax*) and red kangaroos (*Macropus rufus*), none of which was infested with insect larvae. Because insect larvae may not be present or consumed during scavenging, viewing the occurrence of a given species in a dingo scat as evidence of direct killing is tenuous and unsupported. Second, the presence/ absence of a given species in predator diets may simply reflect the predator's selection of one species over the suite of other available species at the time the sample was obtained, and not the true risk to them should species most profitable to dingoes become unavailable at another time; dingoes are capable of switching between very different suites of prey if required (e.g. Robertshaw and Harden 1986; Corbett and Newsome 1987; Thomson 1992; Allen *et al*. 1998; Moseby *et al*. 1998). Third, the abundance of predators and prey present at the time diet samples are obtained is typically unknown, making it difficult to calculate predation rates. Hence, the absence of a particular species in dingo diets might not mean that dingoes do not eat them, but could instead be interpreted to mean that dingoes have already extirpated them. For example, brushtail possums (*Trichosurus vulpecula*) understandably occur infrequently in modern dingo diets from central Australia where dingoes have been implicated in their historical demise (Kerle *et al*. 1992; Foulkes 2001; Allen 2011). Assessments of predation risk, therefore, may be better approached by asking 'could the predator threaten them if they had to?', as opposed to a generic assessment of arbitrary (although sometimes useful) vulnerability characteristics of the prey.

Caution should therefore be exercised when making inferences about predation risk and direct killing on the basis of scat (or diet) data alone. In addition, although knowledge of significant dietary overlap between dingoes and mesopredators is often used to support the idea that dingoes might suppress mesopredators through competition (e.g. Cupples *et al*. 2011), it also points to the direct risks of dingoes to the prey species they each share.

We conclude that the risk of dingo predation is best assessed for each species or population individually using more sensitive and descriptive techniques, perhaps based on the capabilities of the predator and the probability of predator–prey encounters. In addition, predator-manipulation experiments may best provide suitable information for quantifying predation risks (Glen *et al*. 2007*a*). However, in contrast to Glen *et al*. (2007*a*), we would recommend that dingo-removal experiments be conducted in favour of dingo reintroduction experiments because it is easier to protect threatened species from generalist predators than it is to rescue them once the predator is established (Bomford 2008; Allen *et al*. 2011*b*). Dingo-removal experiments may also elucidate the effects of dingo *control* – a management action more immediately applicable to land managers. Dingo-control or dingo-removal experiments may be more ethically and socially justifiable than is intentionally exposing threatened species to predation, and they might also be facilitated more easily by capitalising on current dingo-control practices operating in the Western Division, or other places comparable to it. Alternatively, successful reintroductions of threatened species to arid areas currently occupied by dingoes in similar regions may demonstrate the processes expected by those promoting the reintroduction of dingoes inside the Western Division. Regardless, dingo populations (of unknown genetic integrity) are naturally increasing across the whole of the Western Division at present (B. Allen, unpubl. data, from Livestock Health and Pests Authority records), offering an opportunity to test the predictions made here and in Dickman *et al*. (2009).

Ignoring the shortcomings described throughout, our application of the risk-assessment method of Dickman *et al*. (2009) predicts that up to 71% of extant species will be placed at high risk of predation by dingoes if dingo populations reach high densities, with an additional 23% at low and/or medium risk. By comparison, Dickman *et al*. (2009) predicted up to 66% and 81% of threatened species to be vulnerable to cat and fox predation (respectively) by using this approach. Our study therefore places dingoes within the same 'ball park' as foxes and cats when considering the risk of predation to threatened vertebrates of the Western Division, but the complex interactions between these predators prevent our study being used reliably as a tool to evaluate which predator is 'better' or 'worse' for threatened fauna. Owing to the current overabundance of suitable dingo prey species and the provision of virtually unlimited water supplies in areas south-east of the dingo-barrier fence (Caughley and Grigg 1981; Landsberg *et al*. 1997; West 2008), we see no reason why dingoes could not quickly increase to high densities there in the absence of effective control. It may be argued that dingo populations might limit themselves through social constraints on breeding success (e.g. Wallach *et al*. (2009*b*), but also see Allen (2010*a*)). However, social constraints are unlikely to permanently limit dingo populations to levels below average carrying capacity, with

dingoes reaching extremely high densities when preferred prey are in abundance (Bird 1994; Corbett 2001*b*; Newsome 2011; Allen 2012). Besides, high-density predator populations supported by artificially high prey populations present substantial risks of species extinctions, regardless of whether or not dingo populations (stable or otherwise) have reached maximum carrying capacity (Courchamp *et al*. 2000).

Interest in using dingoes as a biodiversity conservation tool is ultimately driven by the aim of preventing further threatened species' declines (e.g. Carwardine *et al*. 2011). Smith and Quin (1996) and others (e.g. Burbidge and McKenzie 1989; Holmes 1995; McKenzie *et al*. 2007) have asserted that declines in Australian threatened species are not solely dependent on predation by foxes and cats, and that predators *per se* may be problematic. As has been noted (e.g. Dickman *et al*. 1993; Sinclair *et al*. 1998; Norris 2004; Woinarski *et al*. 2010), it is imperative that all the causes of declining populations and extinctions are well understood before imposing radical management actions, such as the reintroduction of an iconic, yet invasive and generalist, predator. This is embodied in the precautionary principle, whereby the *status quo* is maintained until more robust information becomes available (Cooney 2004), or 'until the evidence warrants it' (Soulé *et al*. 2005). We therefore agree with Dickman *et al*. (1993) that a range of causes are responsible for faunal declines in the Western Division and that these causes must be identified and removed before reintroductions of predators or prey occur. Moreover, the available literature discussing the roles of dingoes and their potential efficacy as biodiversity conservation tools provides insufficient supportive evidence for such an approach (Allen 2011; Allen *et al*. 2011*b*). The available literature has not reached consensus on this issue, and our collective knowledge of predator interactions is definitely not well understood (e.g. Robley *et al*. 2004; Glen and Dickman 2005). A range of additional factors also require more careful consideration before dingoes are managed positively (Fleming *et al*., in press).

The Western Division of New South Wales has experienced dramatic and irreparable changes to floral and faunal communities since the arrival of Europeans, livestock and exotic mammals (Dickman *et al*. 1993; Hamblin 2001; Lunney 2001). Thus, pre-European sympatry between dingoes and prey may not be a useful indicator of prey resilience against dingo predation under current or future ecological conditions. We therefore challenge, as did Corbett (2001*b*), the assumption that dingoes will return to fulfil their pre-European ecological roles in socio-ecological systems that have undergone extensive anthropogenic change. We further assert, as did Corbett (2001*b*), Fleming *et al*. (in press), Coutts-Smith *et al*. (2007), Allen (2011) and others (referred to earlier), that dingoes and other wild dogs (regardless of their genetic identity) have the potential to drive modern extinction processes within highly modified ecosystems.

Our application of the method of Dickman *et al*. (2009) therefore suggests that the proposed reintroduction of dingoes risks the further decline of multiple threatened vertebrates, which should not be casually dismissed when considering the net benefits expected to result from positive dingo management. Hence, with Sergio *et al*. (2008) and Fleming *et al*. (in press), we strongly caution against the use of dingoes as a biodiversity conservation tool outside of an adaptive-management framework that addresses all the causes of species declines.

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