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Review

A systematic review of the impacts and management of introduced deer (family Cervidae) in Australia

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Abstract. Deer are among the world's most successful invasive mammals and can have substantial deleterious impacts on natural and agricultural ecosystems. Six species have established wild populations in Australia, and the distributions and abundances of some species are increasing. Approaches to managing wild deer in Australia are diverse and complex, with some populations managed as 'game' and others as 'pests'. Implementation of cost-effective management strategies that account for this complexity is hindered by a lack of knowledge of the nature, extent and severity of deer impacts. To clarify the knowledge base and identify research needs, we conducted a systematic review of the impacts and management of wild deer in Australia. Most wild deer are in south-eastern Australia, but bioclimatic analysis suggested that four species are well suited to the tropical and subtropical climates of northern Australia. Deer could potentially occupy most of the continent, including parts of the arid interior. The most significant impacts are likely to occur through direct effects of herbivory, with potentially cascading indirect effects on fauna and ecosystem processes. However, evidence of impacts in Australia is largely observational, and few studies have experimentally partitioned the impacts of deer from those of sympatric native and other introduced herbivores. Furthermore, there has been little rigorous testing of the efficacy of deer management in Australia, and our understanding of the deer ecology required to guide deer management is limited. We identified the following six priority research areas: (i) identifying long-term changes in plant communities caused by deer; (ii) understanding interactions with other fauna; (iii) measuring impacts on water quality; (iv) assessing economic impacts on agriculture (including as disease vectors); (v) evaluating efficacy of management for mitigating deer impacts; and (vi) quantifying changes in distribution and abundance. Addressing these knowledge gaps will assist the development and prioritisation of cost-effective management strategies and help increase stakeholder support for managing the impacts of deer on Australian ecosystems.

Additional keywords: agriculture, biodiversity conservation, biological invasions, chital, culling, ecosystems, exotic herbivore, fallow deer, fencing, hog deer, red deer, rusa deer, sambar.

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Introduction

Globally, deer (family Cervidae) have been widely introduced outside their native ranges, with populations being established throughout the world (Long 2003; Clout and Russell 2008; Fig. 1). They have adapted to a wide range of habitats (Lowe *et al*. 2000; Long 2003; Forsyth *et al*. 2004), and, in some cases, become overabundant (Côté *et al*. 2004; Nugent *et al*. 2011). Deer have been described as 'keystone species' and 'ecosystem engineers' because of their ability to modify ecosystem function at the landscape scale (Rooney and Waller 2003; Côté *et al*. 2004). Internationally, the detrimental effects of deer on natural and agricultural ecosystems have been extensively documented

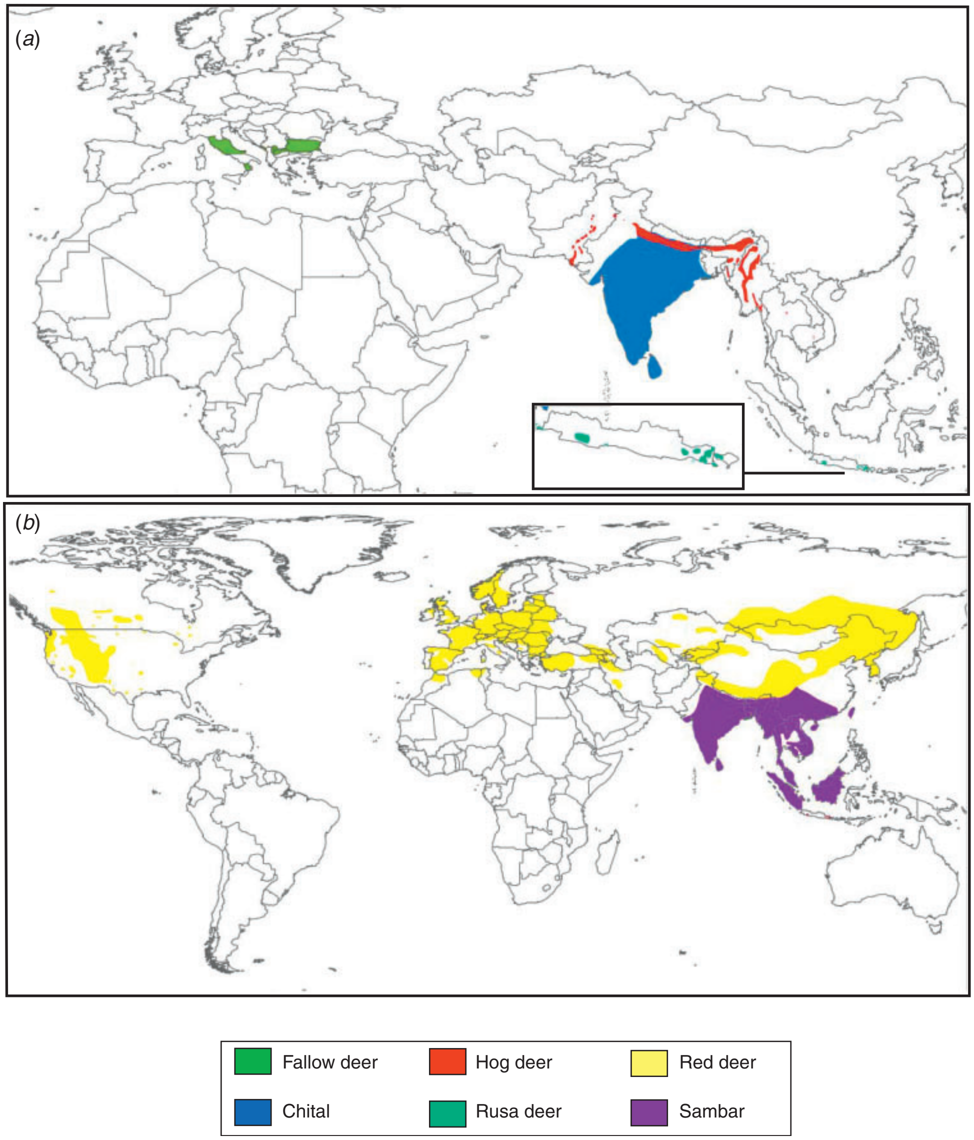

Fig. 1. Global native distributions of the six deer species that have established wild populations in Australia: (*a*) fallow deer (*Dama dama*), chital (*Axis axis*), hog deer (*Axis porcinus*), rusa deer (*Rusa timorensis*); (*b*) red deer (*Cervus elephus*) and sambar (*Rusa unicolor*) (IUCN 2015).

(review in Côté *et al*. 2004). The type and severity of impacts and the mechanisms involved are diverse, ranging from direct physical impacts of foraging on native flora and crops, through to indirect impacts on native fauna and livestock via competition and pathogen transmission, and complex changes to multitrophic interactions (such as nutrient cycling) that influence the functioning of ecosystems (Putman and Moore 1998; Rooney and Waller 2003; Côté *et al*. 2004).

In Australia, establishment of wild deer populations began in the mid-1800s, when Acclimatisation Societies released deer for hunting (Bentley 1998; Hall and Gill 2005). Establishment has continued to the present with accidental farm escapes and deliberate releases (Moriarty 2004*a*). Of 18 species released into the wild, the following six have established wild populations and expanded their ranges beyond the sites of initial introduction: sambar (*Rusa unicolor*), red deer (*Cervus elaphus*), rusa deer (*Rusa timorensis*), fallow deer (*Dama dama*), chital (*Axis axis*) and hog deer (*Axis porcinus*; Bentley 1998; Forsyth *et al*. 2004; nomenclature for all deer species follows Wilson and Mittermeier 2011). Here, we use 'deer' collectively for the six species found in wild populations in Australia, unless otherwise specified.

Historically, deer in Australia were thought to occur at low densities (Strahan 1995). Anecdotal observations and hunting records suggest that the distributions and abundances of some species are increasing (Claridge 2014; Wicks *et al*. 2014; Burgin *et al*. 2015). The few studies that have attempted to quantify trends in abundance and distribution support assertions of recent increases that are likely to continue (Gormley *et al*. 2011; Forsyth *et al*. 2012; Potts *et al*. 2014). Wild deer are present in every state and territory (Moriarty 2004*a*) and occupy habitats ranging from temperate forests to montane and arid woodlands, grasslands, tropical savanna and rainforest (Table S1, available as Supplementary material to this paper).

The management of deer in Australia is an increasingly important, complex (Finch and Baxter 2007; Potts *et al*. 2014; Burgin *et al*. 2015) and costly (McLeod 2004) issue. Although legislation addresses the damaging impacts of deer in some jurisdictions (e.g. NSW Department of Environment and Conservation 2004), it can conflict with game-management objectives (Table 1). The contested policy and legislative positions between viewing deer as a resource (game) or a pest reflects the divergent views within the broader community. Given that it is unlikely that established deer species could be eradicated from Australia, we need to find ways to sustainably coexist with them. Managing deer requires an understanding of the way in which they interact with and affect natural and agricultural ecosystems (English 2007). To assist with the development of cost-effective strategies for managing deer impacts in Australia, we conducted a systematic review (Pullin and Knight 2009) of the evidence regarding impacts of wild deer and the effectiveness of their management in Australia. To establish the potential magnitude of impacts and management, we first mapped the current distribution of wild deer in Australia and estimated the potential for future range expansion. We then (1) synthesised studies investigating modification of natural and agricultural ecosystems by wild deer in Australia, (2) documented the current legal status of deer in Australia and identified evidence of the efficacy of their management and (3) identified current knowledge gaps to inform future research priorities.

Materials and methods

Current and potential distributions of deer in Australia

The distributions of wild deer in Australia were mapped by West (2011). We considered those distributions to be minimum estimates of the current distribution, given that some verified and unverified records of deer in Australia were not included. For example, sambar occur in the Cobourg Peninsula, Northern Territory (http://www.pestsmart.org.au/wild-deer-density-2007-northern-territory/,verified August 2015), and there are unverified records of fallow deer, rusa deer and red deer in south-western Western Australia, red deer in south-eastern South Australia, and fallow and red deer in the mid-north of South Australia.

Potential distributions were inferred using Climatch (Crombie *et al*. 2008; Invasive Animals CRC 2011), an algorithm that predicts the likely range of an exotic species by comparing climates in occupied and potential locations (Crombie *et al*. 2008; S2, available as Supplementary Material to this paper). We used global maps of the native range of each deer species (IUCN 2015; Fig. 1) to define the climate inputs and specified the Australian continent as the target region. For consistency, we used IUCN data for all species. Euclidean distances were used to calculate the 'climate distance' between input sites and each target site across the 16 temperature and rainfall variables used in the analysis (S2).

Reviewing the literature on deer in Australia

We used a systematic search strategy (S3, available as Supplementary Material to this paper) to identify relevant journal articles, books, unpublished reports, conference proceedings and theses. We focussed on evidence of the current detrimental modification of natural and agricultural ecosystems and associated infrastructure, and evidence of the efficacy of approaches being used to manage wild deer in Australia. We did not consider positive impacts. However, we did review studies on the economic benefits of deer hunting, given their relevance to management. We did not consider social impacts, which were reviewed recently by Burgin *et al*. (2015).

We reviewed all literature identified during our searches, including literature that did not contain primary studies, use explicit and reproducible methods, or have a minimum acceptable level of design (Khan *et al*. 2003). This is because deer management is currently being guided by the limited Australian literature available, which includes weak evidence such as anecdotal reports. We included unpublished literature to minimise publication bias (Leimu and Koricheva 2005). We tabulated the objectives, characteristics and outcomes of each study, and used a scoring system to assess the quality of the methods (S3). As the data presented in the literature were unsuitable for meta-analysis, we used a modified systematic quantitative literature review (Pickering and Byrne 2014) and provide a qualitative evaluation of the evidence supporting the conclusions made.

Results

Current and potential distributions of deer in Australia

Successful establishment of deer in Australia is positively related to the number of individuals introduced (Forsyth *et al*. 2004). The mismatch between realised and potential distributions (Fig. 2) and the history of escapes from deer farms and illegal translocations (Moriarty 2004*a*) indicate that current distributions are largely an artefact of historical locations of liberation and escape (Caley *et al*. 2011). Therefore, deer populations are likely to be far from equilibrium (Caley *et al*. 2011). These findings are important because of the irruptive dynamics typical of deer invasions internationally (Forsyth and Caley 2006) and the potential for some species to hybridise (e.g. sambar × rusa deer; Bentley 1998).

Deer have the potential to occupy most of Australia, including parts of the arid interior (Fig. 2). Northern Australia has almost 50% of Australia's cattle (*Bos taurus*) population, and 75% of land is devoted to livestock production (PricewaterhouseCoopers 2011); it could be the next frontier for deer invasion (Fig. 2). Modelling of the factors hypothesised to determine the establishment success of deer introduced to Australia provides evidence that climatic suitability is an important determinant of the spread of existing populations (S3; Forsyth *et al*. 2004). Given the broad climatic zones that deer are able to occupy (Figs 1, 2) and the large areas of Australia that are climatically suitable for some deer species but are currently unoccupied (Fig. 2), it is likely that multiple deer species will occur in sympatry in many areas. We may experience a complex biogeographic re-assortment of the

Table 1. Primary legislation for the management of deer in Australian states and territories
Other legislation such as animal welfare, firearms, workplace health and safety are excluded

State/territory	Status of wild deer	Relevant legislation	Key aspects
Australian Capital Territory	Pest	*Pest Plants and Animals Act* 2005 Pest Plants and Animals (Pest Animals) Declaration 2005	Lists *Cervus* spp. and *Dama* spp. as pests.
New South Wales	Game	*Game and Feral Animal Control Act* 2002	Lists 'Deer (family Cervidae)'.
		Game and Feral Animal Control Regulation 2012	• Deer may be hunted under a licence. Written permission required for hunting on specified public lands (online booking system). • Year-long hunting season for sambar, chital and rusa deer; restricted hunting season for fallow deer and red deer (8 months) and hog deer (1 month). • Different restrictions apply to four Ecological Deer Management (EDM) forests.
	Key threatening process	*Threatened Species Conservation Act* 1995	Feral deer (all species) listed as a Key Threatening Process for herbivory and environmental degradation.
Northern Territory	Pest (feral – prohibited entrant)	*Territory Parks and Wildlife Conservation Act* 2006	
		Northern Territory Government Gazette No. G2, 17 January 2001	Lists *Cervus* spp.
Queensland	Pest	*Land Protection* (*Pest and Stock Route Management*) *Act* 2002	• Class 1 pest – sambar and hog deer. Priority for eradication because not established. • Class 2 pest – chital and rusa deer. Private land managers are required to control. • Class 3 pest – red deer and fallow deer. Private land managers are only required to control where their land adjoins protected environmental assets, e.g. national parks.
South Australia	Pest (declared animal)	*Natural Resources Management Act* 2004	It is an offence to release deer into the wild; sightings must be reported to a Natural Resource Management authority; deer must be controlled on private land.
Tasmania	Wildlife	*Nature Conservation Act* 2002	
	Partly protected wildlife	Wildlife (General) Regulations 2010	• Fallow deer may be hunted under a licence in specified autumn hunting season (1 month antlered males, 2 months antlerless deer). Bag limit of 1 male and 1 antlerless deer or 2 antlerless deer. First-year males are protected. Only rifle hunting permitted. • Crop protection permit (CPP) required for controlling problem deer on private land. May include all sex and age classes. CPP for adult male deer requires a site visit by the Department to assess damage. CPP are generally not issued for antlerless deer November–March when females are pregnant/have dependent young.
Victoria	Wildlife	*Wildlife Act* 1975	• All deer are protected as wildlife. Six species (chital, rusa deer, hog deer, fallow deer, sambar and red deer) are further declared game species for the purpose of the Wildlife (Game) Regulations 2012. • Deer causing damage on public land can be destroyed under an Authority to Control Wildlife Permit. • Deer (excluding hog deer) demonstrably causing damage on private property are subject to an 'unprotection order' and can be destroyed without permit in accordance with specified conditions.
	Game	Wildlife (Game) Regulations 2012	Deer declared to be game can be hunted under a licence where harvest method is specified (e.g. firearms, hounds). Year-long hunting season and unrestricted bag limit for all game deer species, except hog deer (one month season, limit of one male and one female). Other restrictions may apply on public land.
	Pest	*Catchment and Land Protection Act* 1994	All deer except chital, hog deer, red deer, wapiti, sika, sika–red deer hybrids, fallow deer, rusa deer and sambar, are listed as prohibited pest animals.
	Potentially threatening process	*Flora and Fauna Guarantee Act* 1988	Sambar are listed as a Potentially Threatening Process for the reduction in biodiversity of native vegetation.
Western Australia	Declared Pest	*Biosecurity and Agriculture Management Act* 2007	• Fallow, red deer (including wapiti and elk) and rusa deer are declared pests (s22) in Western Australia. • Fallow deer and red deer (including wapiti and elk) may be kept with a permit. All other species are prohibited from being kept in Western Australia.

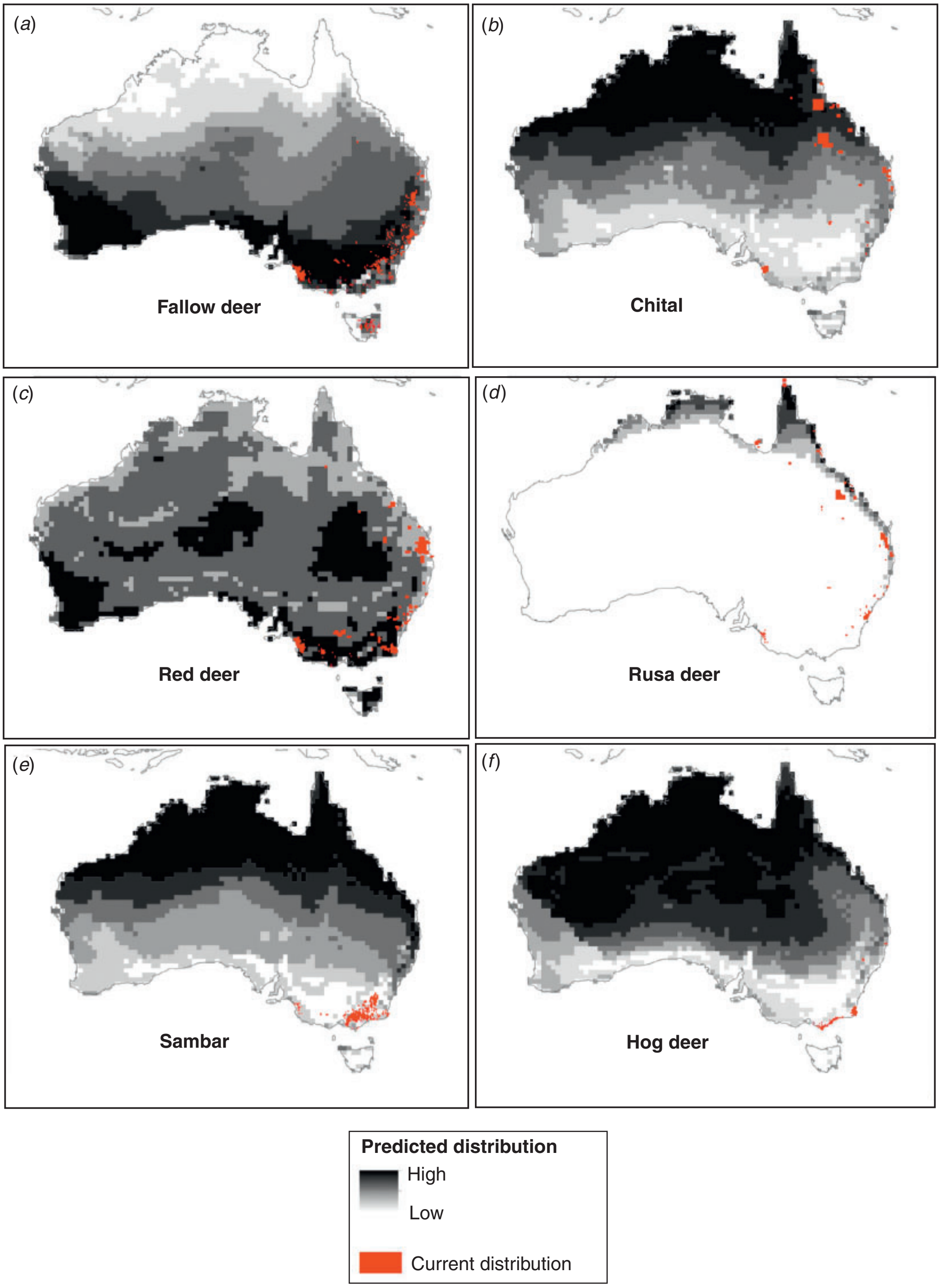

Fig. 2. Current (red; West 2011) and potential distribution (greyscale) of the six deer species established in the wild in Australia. The potential distributions were estimated using the Climatch algorithm (Invasive Animals CRC 2011).

suite of native and introduced herbivores within Australia, and climate change could compound this process (Caley *et al*. 2011).

Colonisation of a new range by deer is likely to be driven by interactions among fire, native vegetation modification, primary production, climate change (Putman and Moore 1998; Rius *et al*. 2014), human population growth and associated human–wildlife interactions (Burgin *et al*. 2015). The influence of fire on deer has been examined in Australia. Anecdotal reports suggest that a direct effect of fire is to cause home-range shifts by deer, which flee fires and must seek food and cover in unburned areas. Fire is speculated to have contributed to the spread of sambar in

Victoria (Bentley 1998). Forsyth *et al*. (2012) assessed the effects of wildfire on sambar abundance, using faecal pellet counts. The large-scale, high-severity fire killed all sambar in the burnt site. However, sambar recolonised within 16–24 months and, subsequently, increased in abundance. In contrast, sambar abundance continued to increase at the unburnt site. At a smaller scale, pre- and post-fire pellet counts revealed that low-intensity fire had little effect on hog deer abundance (Davis 2010*a*). Similarly, vegetation management using mechanical slashing did not alter habitat use by hog deer (Davis *et al*. 2016). The influence of ecological and physiological limits, geographical barriers (Webley *et al*. 2007), drought, climate change, land-use patterns, predation (including hunting) and disease on expansion of Australian deer populations remains unknown.

Summary of literature reviewed

We found 55 peer-reviewed and four non-peer-reviewed journal articles, eight books, seven book chapters, 35 theses, 84 reports and 23 conference proceedings (S3). Of the 216 publications, 21 could not be accessed, 59% were produced in the past decade (2007–16) and 56% of peer-reviewed journal articles were published in the past 7 years (2010–16; S3). Most peer-reviewed research has been conducted in south-eastern Australia, particularly Victoria (45%; S3). Most studies were conducted across multiple habitat types and land tenures, and only 2% were conducted on private agricultural land (S3). Sambar was the most studied species, followed by fallow deer (Table 2, S3).

Impacts

The impacts of deer species can vary according to their body mass, population density and ecology, and habitats and ecosystems vary in their susceptibility to, and ability to recover from, deer impacts (Putman *et al*. 2011). The six wild deer species in Australia evolved in a wide range of environments, from temperate to tropical (Fig. 1; Geist 1998). Their masses overlap those of small (≤30 kg) to large (>180 kg) native and introduced herbivores (van Dyck and Strahan 2008). Although poorly understood in Australian conditions, given that body mass is correlated with foraging niche (Clauss *et al*. 2013), their diets are likely to be broad (S1). The broad ecological niche potentially occupied by deer in Australia has implications for their impacts on natural and agricultural ecosystems.

Changes in plant communities

Studies of deer diet in Australia (S1) have indicated potential negative impacts on a variety of plant species, with qualitative observations of selective foraging by deer and disproportionate effects on plants with an inferred low tolerance to herbivory (Peel *et al*. 2005; Rehwinkel 2008; Claridge 2014). The usefulness of these field observations is limited by the difficulty of distinguishing deer browsing from that by other herbivores (Stockwell 2003), failure to quantify the abundance of deer and other herbivore species, and lack of experimental controls (S3). Most diet analyses have not quantified food availability and, hence, cannot be used to infer diet selection (S3).

Enclosure and exclosure studies provide stronger evidence for effects on vegetation, particularly when differential exclosures partition the effects of deer from those of other herbivores (Table 2, S3). By quantifying changes over time in vegetation exposed to or protected from deer, nine exclosure studies and one enclosure study have provided strong evidence that deer defoliate, strip bark and break stems (Keith and Pellow 2005), leading to reductions in plant biomass in the shrub layer, impeded vertical growth (Bennett 2008) and altered community composition (Hamilton 1981; Moore 1994). Exclosure studies have also provided evidence that deer reduce vegetation cover, tree regeneration (Roberts 2013), plant biomass (Davis 2010*b*, 2014), sapling growth (Davis and Coulson 2010) and plant species diversity (Hamilton 1981). Further, a study comparing vegetation at locations with high (>20 deer km^{-2}) and low (<10 deer km^{-2}) densities of rusa deer suggested that they reduce understorey plant diversity (Moriarty 2004*b*), although the limited scale of that study restricts the extent to which deer can be inferred to be the primary cause. Although these studies have demonstrated that deer herbivory affects vegetation at small scales, evidence of impacts at larger scales is anecdotal (Scientific Advisory Committee 2007; Claridge 2014).

The potential for sambar and hog deer to act as seed dispersers is indicated by their broad diets (S1) and large home ranges (Taylor 1971; Statham and Statham 1996; Mason 2006). Greenhouse trials have demonstrated that deer ingest and excrete viable seeds of exotic and native Australian plant species, including weeds (Eyles 2002; Davis *et al*. 2010). Consumption but not excretion of viable seeds of native and exotic species has also been documented for red deer (Finch 2000) and fallow deer (Philipps 1985; Parker 2009). No Australian study has demonstrated the effects of endozoochory or epizoochory on seed dispersal, or their consequences for plant populations or communities.

The most comprehensive assessments of impacts from antler rubbing have involved targeted surveys for threatened species, which have indicated that rubbing can damage and kill a large proportion of plants (Table 2). However, surveys are correlative, and impacts such as reduced foliage cover and mortality have not been experimentally confirmed as caused by deer (S3; e.g. Bennett and Coulson 2011; Bilney 2013). Selective use of tree species and size classes for rubbing and thrashing by sambar has been demonstrated using systematic vegetation surveys. Shiny nematolepis (*Nematolepis wilsonii*) saplings with a large stem diameter are targeted (Bennett and Coulson 2011), whereas the severity of antler rubbing on yellowwood (*Acronychia oblongifolia*) decreases as diameter at breast height increases (Bilney 2013). However, no study has investigated the consequences of antler activities for plant population viability or community composition.

Rutting and fighting by sambar may create patches of bare ground of up to 30 m in diameter (Bennett 2012). Bowman (2014) attributed reduced understorey vegetation to sambar activity, Moore (1994) documented reduced density of grasses in an enclosure with high sambar densities, and Jesser (2005) reported anecdotal observations that high chital densities expose bare ground. It has been speculated that removal of vegetation by deer causes increased light levels, disrupts moisture dynamics

Table 2. Deer impacts that have been quantified in natural and agricultural ecosystems in Australia

Method: Cd, comparison of high and low deer-density sites; Cs, comparison of deer between sites; D, diet analysis; Dn, DNA sequencing from faecal pellets; Do, direct observation; E, exclosure; En, Enclosure; FP, faecal pellet counts; G, germination trial; S, survey (vegetation, reconnaissance or survey of Authority to Control Wildlife permit holders). *Mechanism*: F, foraging; P, pathogen; R, rubbing; Ro, resource use overlap; T, trampling; Th, thrashing; mechanism demonstrated (●), and mechanism inferred (○). *Deer*: N, not attributable to deer with certainty; Y, attributable to deer with certainty. Research effort ranked by tallying the number of studies that have quantified each impact, where low = 1–3 (■), medium = 4–6 (◆) and high = 7–10 (▲)

Impact category			Vegetation										Fauna				Water	Soil			
Study	Species	Method	Growth	Biomass	Composition	Structure	Diversity	Cover/density	Bare ground	Mortality	Physical damage	Dispersal	Richness/ abundance	Food use overlap	Habitat use overlap	Zoonotic protozoa	Zoonotic protozoa	Erosion	Compaction	Mechanism	Deer
Bartlett 2012	Sambar	Cd											○							F, T	N
Bennett 2002		S									○									F	N
Bennett 2008		E	●	●																F, R, Th	Y
Moore 1994		En			○			○												F, T, Th	Y
Bennett and Coulson 2011		S						○		○										R, Th	N
		E									●									R	Y
Bilney 2013		S								○										R	N
Davies 2014		Cs, Fp, Dn													●	●				Ro, P	Y
Eyles 2002		G										●								F	Y
Lindeman and Forsyth 2008		S									○									F, T, R, Th	N
Lorimer and Lorimer 2005		S						○		○										R	N
Millington 1991		S								○										R, F	N
Nolan *et al*. 2013		Dn															●			P	N
Richardson 2015		S, Do	●										○							F	N
Lindeman and Forsyth 2008	Red deer	S									○									F, T, R, Th	N
Roberts 2013		E						○												F	N
		Fp													●					Ro	Y
Roberts *et al*. 2015		D												○						F	Y
Hamilton 1981	Rusa deer	E	○	○	○			○	○		○									F, T	Y
Keith and Pellow 2005		S				○			○		○								○	F, T	N
		E								○	○									F	Y
		C						○			○							○		F, T	N
Moriarty 2004*b*		Cd, E					○									●				F	N
		D												●						Ro, F	Y
		Fp													●					Ro	Y
Crowther *et al*. 2016		S, Cd			○			○												F	N
Pedersen *et al*. 2014		Cd											○							F, T	N
Davis *et al*. 2008; Davis 2010*a*;	Hog deer	D												●						Ro, F	Y
Davis and Coulson 2010		E	○																	F	N
Davis 2010*a*		E		○																F	N
Davis 2010*a*; Davis *et al*. 2016		Fp													●					Ro	Y
Davis *et al*. 2010		G										●								F	Y
Davis 2013		D												●						Ro, F	Y
Davis 2014		E						○												F	N
Lindeman and Forsyth 2008		S									○									F, T, R, Th	N
Taylor 1971		D												●						Ro, F	Y
		Do													●					Ro	Y
Bailey *et al*. 2015	Fallow deer	S								○	○									R	N
Bird *et al*. 2012		S		○						○										F, R	N
Duncan 1992		D												●						C, F	Y
Nolan *et al*. 2013		Dn															●			P	N
Smith *et al*. 2012; Smith 2012		E	○	○	○				○											F	N
Research effort			◆	◆	◆	■	■	▲	■	▲	▲	■	■	◆	◆	■	■	■	■		

(Department of Sustainability and Environment 2010) and facilitates weed invasion (Jesser 2005); however, this has not been investigated in Australia.

Interspecific competition with native fauna

Potential for competition for food and habitat resources between native herbivores and deer is high in Australia, given the overlap in diet (e.g. Davis *et al*. 2008; Forsyth and Davis 2011). The native species that overlap in body mass (e.g. common wombats, *Vombatus ursinus*, and macropods such as *Macropus* and *Wallabia*; van Dyck and Strahan 2008) and dietary preference are likely to be most at risk from competition for food. Five studies have demonstrated moderate to high dietary overlap, particularly when food limitation is greatest (Table 2), although the strength of evidence in these studies is compromised by limited spatial and temporal replication (S3). Faecal pellet counts and direct observations have demonstrated overlap in habitat use by deer and native herbivores (Table 2; e.g. Davis 2010*a*). No study has experimentally demonstrated resource or interference competition between deer and other fauna in Australia.

Habitat modification

It has been presumed or inferred from the international literature and anecdotal Australian observations that modification of vegetation by deer modifies the habitat of Australian fauna (Peel *et al*. 2005; Claridge 2010; Bilney 2013). Two comparative studies have examined such impacts (Table 2). Bartlett (2012) compared the abundance of small vertebrates in areas of high and low sambar density. Sites with high sambar densities were associated with reductions in small-mammal species richness, abundances of some small mammals and reptile captures (Table 2). These patterns were attributed to reductions in the availability of shelter, food and nesting sites and materials. However, these results were confounded by reduced coarse woody habitat (logs) in areas of high sambar density, which may have affected small-mammal abundance. Pedersen *et al*. (2014) demonstrated negative correlations between the occurrence of two small mammal species and rusa deer at sites burnt within the past 9 years (Table 2). Inferences were limited by the non-experimental nature of the study, the lag time between deer-pellet deposition and small-mammal trapping, and uncertainty in the index of deer abundance. No study has experimentally demonstrated impacts of deer on fauna in Australia.

Interactions with predators

Analysis of scats collected in Victoria has shown that deer are eaten by wild dogs (*Canis familiaris*) and dingoes (*Canis dingo*) and their hybrids, and also by red foxes (*Vulpes vulpes*; Davis *et al*. 2015*b*). Camera traps have shown that wild dogs and foxes scavenge hunter-shot sambar carcasses (Forsyth *et al.* 2014). However, no studies have tested the speculation that increases in deer populations increase food availability for wild dogs, dingoes and foxes (Wicks *et al*. 2014). Similarly, it is unclear whether the presence of deer modifies the functional or numerical responses of predators, with indirect impacts on other prey species, and conversely, whether wild dogs and dingoes reduce deer abundance or contain the spread of some populations. It has been suggested that by opening up the understorey, deer may indirectly facilitate the movements of predators (Claridge and Barry 2000); however, no study has investigated this hypothesis.

Rare and threatened habitats and species

Deer may have direct and indirect impacts on threatened species and communities (NSW Department of Environment and Conservation 2004). Diet analysis (Forsyth and Davis 2011) and field observations (Peel *et al*. 2005; Claridge 2014) have shown that deer ingest rare and endangered plants. However, the only studies that have quantified impacts on threatened species have demonstrated that deer can cause physical damage and mortality of plants (Table 2), but have not ascertained whether deer affect their persistence. The extent and severity of impacts on threatened communities have not been rigorously assessed. For example, in alpine and subalpine peatlands, Tolsma (2009) recorded high use (e.g. 58% of peatlands assessed at Lake Mountain) by deer, but they did not undertake experimental or comparative studies to determine the impacts caused by deer. We do not know whether deer will cause loss of vulnerable species or the degradation and contraction of vulnerable communities.

Pasture, commercial crops, orchards and infrastructure

Evidence of deer impacts on agriculture and associated infrastructure (e.g. fences) in Australia is largely based on surveys of public- and private-land managers (Woolnough and Kirkpatrick 2009; Claridge 2014). This evidence is compromised by reliance on perception and anecdote, lack of quantification of impacts and sampling biases (S3). Therefore, the extent of deer impacts on agriculture in Australia, and the resultant economic loss, is largely unquantified.

The only study to quantitatively examine agricultural impacts of deer included collation of information from Authority to Control Wildlife permits issued during 2002–2007 in Victoria (Lindeman and Forsyth 2008). The most frequently stated reasons for requiring a permit to control deer were eating trees, damaging fences, eating pasture, fruit and vegetable crops, trampling crops and fouling of pasture crops or water (Lindeman and Forsyth 2008).

Deer browse foliage in commercially managed native forests (Di Stefano *et al*. 2009; Hall 2009), and use eucalypt (Masters 2009) and pine plantations (Roff 1960 in Long 2003), but impacts have seldom been quantified. Lindeman and Forsyth (2008) documented browsing and antler rubbing on *Pinus radiata* at one plantation (Table 2).

Competition with livestock

Deer are commonly observed feeding on pastures and crops (Bentley 1998; Mason 2006). Thus, on the basis of evidence from surveys of rural land holders (Finch and Baxter 2007; Peacock 2008) or inference from habitat overlap and international literature (Lindeman and Forsyth 2008; Dryden 2009), deer may compete with livestock for forage. There is evidence that deer consume pasture grasses and forbs in Australia (Finch 2000; Forsyth and Davis 2011; Davis 2013). There are also anecdotal reports that rutting sambar will harass cattle (van Bommel 2013). However, competitive interactions between livestock and deer have not been demonstrated. The only quantitative evidence that has implicated deer in food competition with livestock in Australia used paired exclusion cages to demonstrate spatial and temporal variation in pasture loss to wildlife grazing, as well as impacts on ground cover and species composition (Smith *et al*. 2012; Table 2). However, these impacts were not partitioned among species, and the relative contribution of grazing impacts by fallow deer is unknown.

Vectors of diseases and pathogens

Wild deer are potentially susceptible to endemic and exotic diseases and parasites that may affect other animal species, including humans. The major emergency diseases that could affect wild deer were reviewed by Animal Health Australia (2011; Table 3).

Protozoan parasites (*Cryptosporidium* and *Giardia* species) that could cause zoonotic disease in humans (Hampton *et al*. 2006) have been detected at low levels in deer faecal pellets in Australian drinking-water catchments (Ng *et al*. 2011; Nolan *et al*. 2013), although the risk to human health has not been quantified. Deer are susceptible to lyme disease, which is a common zoonotic viral infection transmitted via ticks from deer to humans, but it has not been documented in Australia.

Deer may transmit endemic diseases and parasites to domestic animals and humans when using improved pastures (Claridge 2014) and livestock water resources (Woolnough and Kirkpatrick 2009), or through contact between hunting dogs and infected animals (Sparkes *et al*. 2016). The potential for disease transmission depends on the susceptibility of individual animals to disease, population distribution and density, and direct contact with domestic animals (Garner and O'Brien 1988). Wild deer could also act as vectors for pathogens that affect wildlife (Phillott *et al*. 2010). However, few diseases have been reported in captive or wild deer in Australia (Table 3), probably reflecting a low survey effort. In the only such survey available, wild sambar, fallow deer and hog deer in south-eastern Australia tested negative for leptospirosis antibodies, whereas rusa deer displayed serological evidence of exposure (Milner *et al*. 1981). Red deer in south-eastern Queensland have displayed serological evidence of exposure to several endemic livestock diseases, including leptospirosis and Akabane virus, and they carry several species of parasitic helminths (McKenzie *et al*. 1985; Table 3). A small sample of rusa deer in Royal National Park have been shown to display serological evidence of exposure to Q fever (*Coxiella burnetii*), leptospirosis, Akabane virus and bovine ephemeral fever virus, in addition to ticks (*Ixodes* spp.) and parasitic helminths (Moriarty 2004*b*).

The potential introduction of exotic animal diseases such as foot-and-mouth disease (FMD) or surra (*Trypanosoma evansi*) is of concern, with the cost of an outbreak of FMD in Australia being estimated at AU$0.6–5.2 $\times 10^{10}$ (Buetre *et al*. 2013). Australian wild deer could play a significant role in the introduction of surra, given that rusa deer have been implicated in the transmission of this disease from Indonesia to Papua New Guinea, posing a high biosecurity risk to Australia (Reid *et al*. 1999). There is a protocol to implement if such a disease establishes in Australia (Animal Health Australia 2011), but our ability to effectively contain an emergency disease in wild deer populations is unknown. Factors likely to contribute to the risk of a disease becoming established, transmitted and dispersed within deer populations are gregarious grouping and high mobility, cryptic behaviour and the use of inaccessible terrain, which would make control difficult (Animal Health Australia 2011). Climate change and range expansion of insect vectors (e.g. midges (*Culicoides* spp.), which carry bluetongue disease), coupled with range expansion of wild deer, could increase the risk of livestock disease outbreaks (Simpson and Srinivasan 2014). Only one study has investigated such aspects of disease in wild deer. Statham and Statham (1996) conducted a short, small-scale simulated disease-control program. They showed that following ground-based shooting, the location of 92% ($n = 12$) of radio-tracked fallow deer remained within their pre-shooting home range. They concluded that shooting to sample deer during a disease outbreak would not cause surviving deer to disperse.

Water quality, soil properties and nutrient cycling

Deer activity may be concentrated around water sources (Forsyth *et al*. 2009), and observational evidence has generated concerns

Table 3. Emergency diseases of concern and endemic parasites and diseases that could be carried by wild deer in Australia
Other endemic disease lists only diseases previously recorded in deer in Australia

Type of disease	Disease
Emergency disease[A]	Anthrax[B], Aujeszky's disease, bluetongue[B], brucellosis[B], foot-and-mouth disease, Japanese encephalitis, peste des petits ruminants, rabies, screw-worm fly, surra
Other endemic disease	
Parasite	Protozoan parasites (*Cryptosporidium* spp. and *Giardia* spp.)[C,D], cattle tick (*Rhipicephalus microplus*)[E,F], other tick species (e.g. *Ixodes* spp.)[E,G], gastrointestinal helminths[C,G,H]
Bacterial disease hosted by ticks	Anaplasmosis (caused by *Anaplasma phagocytophilum*)[I], dermatophilosis (caused by *Dermatophilus congolensis*)[I]
Other bacterial disease	Leptospirosis (caused by sprirochaete *Leptospira* spp.)[G,J], Johne's disease (caused by bacterium *Mycobacterium paratuberculosis*)[K], Q fever (caused by bacterium *Coxiella burnetii*)[G]
Virus	Akabane virus[E,G], bovine ephemeral fever virus[G], malignant catarrhal fever[L,M]

[A]Animal Health Australia 2011; [B]Organism, or strains of organism, that cause(s) disease endemic to Australia; [C]Ng *et al*. 2011; [D]Nolan *et al*. 2013; [E]McKenzie *et al*. 1985; [F]Cutullé *et al*. 2009; [G]Moriarty 2004*b*; [H]Davies 2014; [I]Garner and O'Brien 1988; [J]Milner *et al*. 1981; [K]Animal Health Australia 2015; [L]Fyffe 2008; [M]Tomkins *et al*. 1997.

about degradation of water quality in Australian creek and river systems (Department of Conservation and Environment 1992; NSW Department of Environment and Conservation 2004). However, no study has determined whether erosion and loss of vegetation in riparian areas is caused by the activities of deer, or whether deer contribute to water degradation (e.g. changes in turbidity or increased nutrient load) in natural ecosystems.

It has been speculated, on the basis of anecdotal observations and international literature, that deer may cause compaction and erosion of soils, particularly in areas of heavy use, such as water points (Parks Victoria 2005) and tracks (Forsyth 2006), and in low-lying areas vulnerable to gully erosion (Peel *et al.* 2005). Even if localised, the consequences may be severe in environments with erodable soils (Department of Conservation and Environment 1992) and in creek and river systems (NSW Department of Environment and Conservation 2004). However, only one study has examined such impacts (Table 2). Keith and Pellow (2005) used visual assessments to compare the effects of rusa deer on soils at areas of concentrated deer activity and at sites where activity was not concentrated. They recorded localised soil erosion associated with high densities of deer tracks and pellets; however, owing to the qualitative nature of their observations, the presence of other mammalian herbivores, and lack of experimental manipulation, evidence for these impacts is weak. We do not know whether deer activity affects soil physical properties in Australia, nor whether there are flow-on effects for plant communities. Further, we do not know whether deer in Australia affect nutrient cycling, decomposition and primary productivity by mediating feedback cycles between plant communities and the decomposer subsystem, or by redistributing nutrients in the environment, although the potential for such impacts is evident on the basis of knowledge regarding habitat use (Table S1).

Management of impacts

Deer are managed for different outcomes in each Australian state and territory (Hall and Gill 2005) and are, therefore, variously classified as 'pest', 'game' and 'protected wildlife species' (Table 1). Nonetheless, all states and territories have legislative provisions for managing deer impacts (Table 1).

Ground-based shooting is being used to manage deer in peri-urban and regional areas (e.g. Department of Primary Industries Parks Water and Environment 2011; Moriarty and Brown 2012) and in national parks (e.g. NSW Department of Environment and Conservation 2005). However, there is little information on how, or if, success is being measured in most programs. Only one study has examined the efficacy of ground-based shooting for managing deer in Australia. A shooting program in Victoria reduced the quantity of sambar faecal pellets adjacent to a reservoir, thereby reducing the risk of faecal contamination of water resources, primarily through deterrence rather than a density reduction (Bennett *et al.* 2015). Ground-based shooting was used in parts of Kangaroo Island (440 500 ha; Invasive Animals CRC 2013) to eradicate fallow deer. These examples support the assertion that targeted, ground-based shooting may be effective for reducing densities at small spatial scales (e.g. <1000 ha) and could be used to eradicate isolated populations at locations where immigration is unlikely.

Aerial surveys of deer in Queensland have suggested that helicopter-based shooting may be a promising technique for controlling deer in open habitat where visibility is high (Baillie 2014). Aerial shooting is being used to control deer in New South Wales and South Australia. In the former, 1795 deer were shot during 2013–2015 in habitats ranging from open forest and woodland to grassland and wetlands (G. Eccles, NSW National Parks and Wildlife Service 2016, pers. comm.). In the latter, 182 deer (mostly fallow deer but also red deer and sambar) were killed in 4 h (Peacock 2008). No study has quantified the efficacy of recreational hunting as a management strategy, but a trial is underway in Victoria to assess whether this approach can reduce sambar abundance and impacts on natural ecosystems (Davis *et al.* 2015*a*).

Few studies have examined other options for management of deer impacts. A bait station has been designed and trialled to selectively deliver bait to wild ungulates, including deer, while largely excluding native wildlife (Hunt *et al.* 2014). However, there are no registered poisons for the management of deer in Australia (Australian Pesticides and Veterinary Medicines Authority 2015), although the use of cyanide is currently being investigated (Natural Resources Commission 2016). Two studies have demonstrated that small-scale fencing can protect vegetation from deer (Davis and Coulson 2010; Bennett and Coulson 2011). One study found contraceptive implants prevented reproduction in captive female—but not male—rusa deer (Webley 2009).

Many local- and state-level deer management plans have been developed (e.g. NSW Department of Environment and Conservation 2005). However, these largely rely on anecdotal observations and perceptions of deer abundance and impacts, and assume that the proposed management actions will fulfil management objectives (Pople *et al.* 2009) without monitoring to evaluate the requirement for, or efficacy of, management.

Although few studies have experimentally evaluated management options, monitoring techniques have been developed to evaluate the effectiveness of management interventions. For example, monitoring protocols have been developed that use faecal pellet counts (Parks Victoria 2005; Forsyth 2006) and camera trapping to index the relative abundances of deer (Davis 2014; Davis *et al.* 2015*a*), and studies have compared the efficacy of spotlight counts, distance sampling and aerial surveys (Amos *et al.* 2014*a*; Baillie 2014) to index or estimate deer abundances (S3). Bennett and Coulson (2008) developed a method to measure the impacts of sambar on vegetation using differential exclosures, although the application of this design to smaller deer species has not been tested. Importantly, the relationship between deer density and impacts has not been reported for Australia; rather, it has been assumed that a reduction in deer density will reduce impacts.

Efforts of deer management can be compromised when there are competing value systems (Finch and Baxter 2007). Targeted surveys of stakeholder groups have been used to gauge community support for deer-control options in Australia (Finch and Baxter 2007; Claridge 2014), highlighting the wide-ranging views of stakeholder groups and how attitudes change with time. For example, Finch and Baxter (2007) surveyed landowners and managers in Queensland in 2005, and found that >50% of respondents wanted deer populations to stay at current levels or increase, whereas in recent surveys of ranger-level staff across the Australian Alps, Claridge (2014) reported almost unanimous agreement that control of deer is necessary.

Knowledge gaps

The case of wild deer in Australia is similar to that described by Bengsen *et al*. (2014) for feral pigs (*Sus scrofa*), namely, many potential impacts have been inferred from anecdotal observations, untested retroductive hypotheses or international studies, rather than from systematic and quantitative studies. Most research relating to deer impacts and management has consisted of small-scale, short-term, single-species case studies, with limited generality (S3).

Coordinated research, using experimental and comparative approaches, is needed to place the management of Australia's deer on a sound footing. We draw on international experience to outline key knowledge gaps and research priorities for Australia. However, we emphasise that managers should not solely rely on international knowledge, for three main reasons. First, impacts vary between ecosystems with different evolutionary histories (Dolman and Wäber 2008). Second, Australia's unique environment (i.e. infertile soils, variable climate and associated characteristic biota; Braithwaite 1990; Orians and Milewski 2007) may mean that ecology of deer in Australia differs from that of the same species in other environments (Amos *et al*. 2014*b*). Third, evaluation of the impacts of deer in Australia is complicated by difficulties in partitioning their effects from those of sympatric native and introduced mammalian herbivores, and dealing with this requires novel approaches.

Impacts

There is evidence that wild deer in Australia have impacts on individual plants at small spatial scales (Table 2). However, we do not understand how deer affect vegetation communities. Moreover, few studies have investigated community- and ecosystem-level implications of these impacts on fauna, water quality and soil processes. Although there is overlap in resource use between deer and native fauna (Table 2), no study has demonstrated competition. Given the diversity of deer species and the environments that they occupy, even those research areas that have received most attention (Table 2) have substantial knowledge gaps. The international literature has clearly shown the potential for far-reaching ecosystem-level consequences of deer at high population densities and in sensitive habitats (Rooney and Waller 2003; Côté *et al*. 2004). In Australia, evidence of increasing distributions and abundances of deer (Fig. 2), coupled with increasing evidence of impacts (Table 2), suggests that in the future deer will have serious and widespread effects on natural and agricultural systems. Robust demonstration of the type and extent of impacts in Australia is essential to justify investment in deer management and to gain social acceptability and stakeholder support (English 2007; Nugent *et al*. 2011).

Management

There has been little evaluation of the efficacy of deer-management techniques in Australia, and our understanding of deer ecology (required to guide deer management) is limited (Table S1). International experience provides insight into management tools that should be investigated for the management of deer in a range of Australian environments.

Professional helicopter- and ground-based shooting has reduced deer densities in New Zealand (where some of the deer species that occur in Australia are managed as pests; Crouchley *et al*. 2011; Forsyth *et al*. 2013). Government-funded ground-based shooting has had little effect on deer distributions and abundances (Caughley 1983), whereas widespread commercial ground- and helicopter-based shooting has substantially reduced deer abundances at large spatial scales (Forsyth *et al*. 2011). Commercialisation of wild deer in Australia may reduce deer distributions and abundances, although there is no evidence that commercial harvesting of feral pigs and goats (*Capra hircus*) in Australia has reduced their densities (Ramsay 1994). Indeed, commercialisation carries the risk of greater *de facto* protection of harvested species, leading to higher average densities (Pickles 1992). Recreational ground-based shooting has been proposed as a tool for controlling overabundant deer (e.g. in North America; Urbanek *et al*. 2011), but a recent review concluded that recreational ground-based shooting in Australia contributes little to the control of pest species such as deer (Bengsen and Sparkes 2016). Overall, the evidence from international control programs indicates that ground- and aerial-based shooting by professional shooters is likely to be the most widely applicable approach for controlling deer across large-scale (e.g. 4000 ha; Forsyth *et al*. 2013) management units in Australia. In New Zealand, helicopter-based shooting is the most cost-effective method for reducing deer densities in non-forested and montane forest habitats (Nugent *et al*. 1987; Forsyth *et al*. 2013). However, the best way to implement shooting in Australia, for any deer species or habitat, is untested.

Fertility control (surgical sterilisation, hormone implants or vaccination; Warren and Warnell 2000) has been used to reduce the reproductive success of female deer in North America (Garrott 1995; Kirkpatrick *et al*. 2011). This can reduce the density of small isolated populations (Rutberg and Naugle 2008) and is best suited to situations where the aim is to maintain deer at reduced densities, rather than eradication (Massei and Cowan 2014). Fertility control is considered infeasible for large populations (Raiho *et al*. 2015) and is expensive to implement even for small populations (Garrott 1995). Fertility control would, therefore, be unsuitable for most Australian situations.

Strategic exclusion of deer with fencing can be used for protection of natural or agricultural assets or to enable restoration (Dvorak and Catalano 2016). Fencing is most commonly used at small scales (e.g. <100 ha) to prevent deer impacts in Australia (e.g. Lorimer and Lorimer 2005), and is not being used for large-scale asset protection as occurs internationally (e.g. fencing in Maungatautari, New Zealand, which protects 3400 ha of ground-bird habitat from introduced mammals, including deer; Innes *et al*. 2012). The optimal design and scale for implementation of fencing to protect Australian natural and agricultural assets from deer are unknown.

In New Zealand, aerially sown baits intended for control of brushtail possum (*Trichosurus vulpecula*), and hand-laid foliage gel, both containing sodium monofluoroacetate (1080) poison, kill deer (Nugent 1990; Forsyth *et al*. 2013). However, the efficacy of poisoning varies with the mass of deer (Nugent and Yockney 2004) and food availability (Crouchley *et al*. 2011). Using poison would pose significant risks to non-target species in Australia (McIlroy 1982).

Reintroduction of the wolf (*Canis lupus*), a top-order predator, to parts of North America has reduced impacts of deer (Beschta and Ripple 2009). The presence of high predator densities can induce behavioural responses in deer (e.g. increased vigilance and altered habitat use) that alter their impacts on vegetation (Manning *et al*. 2009). Hence, reintroducing dingoes to their former range, or relaxing control regimes so that populations increase, might reduce the impacts of deer through reduced abundances (by direct predation) and/or altered habitat use.

Guardian dogs are widely used to protect livestock from predators, and studies have shown that they may be able to separate deer from livestock, either because the dogs harass deer or because deer avoid areas used by the dogs (Gehring *et al*. 2010; VerCauteren *et al*. 2012). This may reduce disease transmission (VerCauteren *et al*. 2012) and food competition between deer and livestock (Gehring *et al*. 2010).

Other approaches that have been considered internationally include limiting the use of key habitats for deer range expansion, for example, through fencing ecotone areas and manipulation of forest and agricultual landscape matrixes (Saito *et al*. 2016). Scare devices and repellents (topical application of distasteful chemicals or predator scent) may be useful at high-value sites, but are generally effective for short periods only (weeks–months) (Walter *et al*. 2010). Finally, biological control of deer does not appear to be a feasible option for managing deer because of the threat this would pose to farmed animals (Nugent and Fraser 1993).

Research priorities

We have identified the ecological information required for understanding and managing the risks posed by wild deer (Table 4). There are many knowledge gaps, but below we list the six most important areas for further research.

(1) Long-term changes in plant populations and communities

Identification of plant populations and communities most vulnerable to deer impacts is important for prioritising management efforts. The long-term consequences of selective herbivory by deer on Australian plant populations and communities can be separated from the effects of native species by using differential exclosures. Long-term (decadal) studies using networks of exclosures (Wardle *et al*. 2001) in a range of vegetation types and faunal assemblages are required.

(2) Interactions with native fauna

Many Australian animals are threatened or endangered, and increasing deer abundances could affect some of these species directly or indirectly. Understanding direct and indirect interactions with native fauna, such as food-resource competition,

Table 4. Research on risks posed by deer that would directly (●) and indirectly (○) improve understanding of deer impacts and management in Australia

Research priorities are ranked by tallying the number of risks posed by wild deer that each research area can directly (1 point) or indirectly (0.5 point) contribute to improved understanding and management, where low = 1–5 (■), medium = 6–10 (◆) and high = 11–14 (▲)

Research area	Risk posed: Natural ecosystems						Agricultural ecosystems			Natural and agricultural ecosystems		Management			Priority
	Plant community composition, structure, regeneration	Plant dispersal and establishment	Interspecific competition: native fauna	Habitat and food resource modification	Interactions with predators	Habitats and species at high risk	Plantation, crop and infrastructure damage	Interspecific competition: livestock	Disease and pathogen transmission	Water quality	Soil properties and nutrient cycling	Techniques or strategies	Identifying priorities	Social acceptability	
Abundance	●	●	●	●	●	●	●	●	●	●	●	●	●	○	▲
Distribution	●	●	●	●	●	●	●	●	●	●	●	○	●	●	▲
Habitat use	●	●	●	●	●	●	●	●	●	●	●	○	●	○	▲
Diet	●	●	●	●		●	●	●			○		●	○	◆
Social organisation	○	○	○	○	○	○	○	○	○	○	○	●			◆
Movement	●	●	●	●	●	●	●	●	●	●	○	●	●	○	▲
Behaviour	●			●		●	●								■
Genetics and hybridisation												●			■

habitat alteration and predator facilitation, requires large-scale manipulative experiments that examine how the presence of deer alters the health and population dynamics of sympatric fauna.

(3) Impacts on water quality

Water is a limiting resource in many parts of Australia, and deer commonly inhabit drinking-water catchments (Ng *et al*. 2011; Amos *et al*. 2014*b*; Bennett *et al*. 2015). The impact of deer on water quality through contamination with protozoan parasites is difficult to measure (Nolan *et al*. 2013). However, the effects of soil compaction (Kumbasli *et al*. 2010), reduction of vegetation cover and associated erosion, sedimentation and nutrient contamination (Clarke *et al*. 2000; Keith and Pellow 2005; McDowell 2007) can be assessed by comparing sites with differing densities of deer.

(4) Economic impacts on agriculture

To justify management, the economic benefits of management should outweigh the sum of the impacts and the cost of management. Quantifying the susceptibility of pasture, crops and livestock (Gehring *et al*. 2010; Bleier *et al*. 2012) to deer impacts, such as the risks of disease transmission between livestock and deer (VerCauteren *et al*. 2012), and determining how these effects can be mitigated by management, would enable primary producers to minimise the economic costs of deer (Putman and Moore 1998).

(5) Cost-effective management of deer impacts

Despite recognition of the risks posed by wild deer, the costs and benefits of management options have not been reported. The applicability and efficacy of management options will vary with scale and biophysical factors, including deer species, terrain, canopy cover and proximity to roads and residential areas. For example, the most effective control techniques for solitary sambar in dense forest may differ from those for group-living chital in open grassland. The management effort required to improve the condition of resources affected by deer can be established through monitoring both deer abundance and resource condition. Wherever possible, robust study designs (i.e. replication, randomisation, and comparison of treatment and non-treatment areas) should be implemented.

(6) Changes in the distribution and abundance of deer

Knowledge of deer abundance and distribution is critical for understanding the current and future impacts of deer, and for evaluating and reporting on the success of management actions (Table 4). Our simple approach to predicting future deer distributions in Australia (Fig. 2) demonstrates the potential for substantial range expansion. More sophisticated approaches to modelling deer distributions (e.g. Gormley *et al*. 2011) would improve our understanding of the factors that limit and enable range expansion. This information could be used to anticipate and prevent range expansion (West and Saunders 2003). Monitoring deer occupancy and abundance at a sample of locations would enable robust statements about trends in distribution and abundance. This information would help identify new deer populations that could be eradicated, and invasion fronts where investment in surveillance and control has the greatest potential for containing expanding deer populations.

Conclusions

This systematic review illustrates the potential for deer to have an impact on Australia's natural and agricultural ecosystems. Impacts are likely to occur primarily through the direct effects of herbivory on vegetation, but there may be cascading indirect effects on fauna and ecosystem processes. It is likely that some deer species in Australia will further increase in distribution and abundance (Gormley *et al*. 2011; Potts *et al*. 2014), resulting in increased impacts on natural and agricultural systems and social amenity.

Internationally, it has been demonstrated that intensive management is required to reduce the impacts of high-density deer populations on ecosystems (Frost *et al*. 1997; Husheer and Robertson 2005). Over 35 years ago, Frith (1979) observed that in Australia:

'One of the most pressing needs in wildlife conservation is to discover what is the place of the sambar in the ecology of the south-east highlands. If the presence of the deer is adverse to native flora and fauna, then how can they be controlled? If they are not adverse, then how could they be managed for long-term productivity as game animals?' (p. 163).

However, progress towards understanding and managing the impacts of deer in Australia has been slow (Claridge 2014). Evidence of deer impacts is largely observational, and most studies have not experimentally partitioned the impacts of deer from those of sympatric native and introduced herbivores. A lack of knowledge regarding the efficacy of methods for mitigating deer impacts, and disagreement regarding the most cost-effective techniques, is hindering the management of wild deer in Australia (West and Saunders 2003; Claridge 2014). Research at appropriate management scales is needed to determine the current extent, severity and nature of deer impacts in Australia, and to better predict future impacts. Addressing these knowledge gaps will assist with the development and prioritisation of cost-effective management strategies, and would likely increase stakeholder support for managing the impacts of deer on Australian ecosystems.

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