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Minimizing species extinctions through strategic planning for conservation fencing

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Abstract

Conservation fences are an increasingly common management action, particularly for species threatened by invasive predators. However, unlike many conservation actions, fence networks are expanding in an unsystematic manner, generally as a reaction to local funding opportunities or threats. In a gap analysis of Australia's substantial predator exclusion fence network, we found highly uneven protection, with 67% of predator-sensitive species remaining unrepresented. Predator exclusion fences all contain small populations of threatened species, therefore a novel systematic prioritization method for expanding fence networks that explicitly incorporates population viability analysis and minimises expected species' extinctions was developed. The approach was applied to New South Wales, Australia, where the state government intends to expand the existing conservation fence network. A systematic prioritisation yields substantial efficiencies, reducing the expected number of species extinctions as much as 17 times more effectively than *ad hoc* approaches. This dramatically superior outcome emphasises the importance of governance when management action is applied in multiple instances with similar objectives and using systematic methods rather than expanding networks opportunistically.

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Introduction

Invasive predators are a leading driver of global biodiversity decline and loss (Mack et al. 2000, Clavero and García-Berthou 2005), particularly in ecosystems where prey species are evolutionarily naïve. Introduced predators have been implicated in 60% of mammal extinctions (46 species) and 55% of bird extinctions (77 species; (IUCN 2015), particularly in southern hemisphere ecosystems. On islands, populations of invasive predators are frequently targeted for eradication, but this becomes infeasible over large areas (Clout and Veitch 2002, Rejmánek and Pitcairn 2002). Australia (Dickman 2012), New Zealand (Burns et al. 2012) and many island ecosystems (McCreless et al. 2016) are increasingly turning to conservation fences to exclude introduced mammalian predators where eradication is impossible, and when prey species are vulnerable to any nonzero density of an introduced predator.

Conservation fencing is a rapidly expanding management action (Hayward and Somers 2012). Fencing creates a physical barrier between conservation assets and threatening agents, providing a level of protection that is much higher than alternative management actions (Hayward and Kerley 2009). Reintroductions of prey species into areas with ongoing predator control are typically less successful than predator free areas (Short et al. 1992, Short 2009). For an upfront investment in the construction of the fence (Bode et al. 2012, Norbury et al. 2014), conservation organisations can reintroduce species with a rate of success comparable to translocations on predator free islands (Short 2009). Fences are consequently popular across the conservation sector (Hayward and Kerley 2009), even for small organisations.

Economic theory suggests that sectors made up of diverse, independent organisations will be better able to adapt to local environmental and socio-political conditions; to access diverse funding sources and local volunteers; to have lower operating and transaction costs; and to experiment and innovate (Bilodeau and Slivinski 1997, Albers and Ando 2003, Armsworth et al. 2012). A network of independently operated fences is therefore a positive reflection of a diverse conservation community. However, this broad accessibility has helped to create an organisationally decentralised fence network. In many instances, fencing projects are rapidly arising independently of each other. In Australia, for example, the majority (58%; see Table S1) of fences are operated by nongovernmental organisations or local councils. The same decentralisation can be observed in New Zealand, where 78% of fences are nongovernmental initiatives (Saunders & Norton 2001; Burns et al. 2012). This situation is unusual for conservation in these two countries, whose political systems and history of land tenure has seen the majority of protected area designations undertaken by state or federal governments (Saunders and Norton 2001, Burns et al. 2012).

Decentralisation in conservation can result in unsystematic, uncoordinated actions and costly inefficiencies (Pressey et al. 1993), incomplete protection (Margules and Pressey 2000) and enormous legacy costs (Stewart et al. 2007, Fuller et al. 2010). Such inefficiencies are also likely to be

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a feature of existing fence networks. By considering fences built for similar purposes as a network, systematic approaches (such as those used for designing protected area networks; Margules and Pressey 2000) will improve the effectiveness of conservation fencing. However, unlike protected areas, fences need to be sited and constructed, the animals often translocated into the area, and populations actively maintained. Moreover, compared to reserve systems (see Barr et al. 2016), the decentralised organisation and funding structure of fence networks mean that any inefficiencies will be difficult to correct using top-down control. Nevertheless, coordination has the potential to substantially increase the performance of fence networks. New methods are therefore required to identify and prioritise new fencing projects.

Australia is a global epicentre of mammal extinctions (Woinarski et al. 2015), driven primarily by invasive foxes (*Vulpes vulpes*) and cats (*Felis catus*; (Abbott 2011, Woinarski et al. 2011). Currently 58 mammal species are recognised in the Environment Protection and Biodiversity Conservation Act 1999 (EPBC) as threatened by invasive predators, many of which might benefit from predator exclusion fences (Woinarski et al. 2014). However, because fenced populations are small and constrained, methods must explicitly calculate and minimise species extinction probability. This focus on viability is therefore an essential element of fence network planning. To prioritise further fencing projects in this context requires systematic methods which better cater to this context.

In this paper, we design and implement a systematic method that evaluates the current performance of a network of fences, and quantifies the relative benefits of alternative future fence projects using the theories of population viability and systematic conservation planning. This method has application for any number of fence networks such as those in Hawaii (Cole et. al. 2012), Africa (Hayward and Kerley 2009) and New Zealand (Burns et. al. 2012). Similarly, its application need not be restricted to fencing, being equally applicable to the prioritisation of threatened species management actions generally. To illustrate the approach, we consider Australia's network of predator exclusion fences built for the conservation of threatened mammals within the state of New South Wales. We first review the state and performance of Australia's existing network of predator exclusion fences and assess whether the network exhibits the inefficiencies expected from such a decentralised structure. Second, we outline and explain a flexible systematic framework for optimally expanding existing fence networks, and apply it to a New South Wales (NSW) case study, where the state government is planning two new fence projects. As with systematic conservation planning, the method seeks to construct an efficient and complementary network of fences.

METHODS

Goals and objective function

A number of methods have been developed for the spatial prioritisation of protected areas and reserve design. Tools such as Marxan (Ball et. al 2009) are used to identify the collectively cheapest set of planning units required to attain a minimum acceptable level of representation (i.e. populations of species contained within reserves). The logic behind these prioritisation methods

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indeed have great overlap with the spatial prioritisation of conservation fencing, but the formulation of the optimisation problem differs in three important ways. Firstly, fencing is typically used as a crisis management action, where the goal of action is recover species on the brink of extinction. Prioritising fence projects over a suite of species therefore requires a quantitative and comparable method to assess a species extinction risk, not goals such as area coverage or percentage representation. Second, locally extirpated species are almost always translocated into fences (Dickman 2012) rather than populations that remain *in-situ*. As a result, the locations of new projects must be based on the suitability of a site for key species, rather than areas of current occupancy. Finally, since fenced populations are often small and spatially constrained, one cannot assume that representation guarantees persistence (Lindsey et.al. 2011), instead fence networks must focus explicitly on population viability. We therefore choose fences with the goal of minimising expected extinctions across a suite of species. In the following methods, we describe a new systematic approach for choosing priority locations for fencing projects based on this objective.

Mathematical definition of the benefit function and search algorithm

When viewing fences as a network, we need to choose fence locations that provide the greatest aggregate benefit to conservation. Clearly fences should be sited in areas that would provide suitable habitat for a large number of species that are threatened by invasive predators (Fig. S1). However, the optimal choice is not as simple as overlaying suitability maps to identify biodiversity hotspots. Problems such as overrepresentation in the fence portfolio can only be rectified if the presence of each species is modified by a series of filters. First, we need to modify the value of each species by taking into account their current conservation status. Second, we need to correct the species richness of each site by the existing representation of those species in conservation management projects elsewhere – that is, we need include complementarity. Finally, we need to consider the risk of full or partial project failure, a serious and acknowledged problem for threatened species translocations (IUCN/SSC, 2013; Short 2009). In the section following, we integrate each of these factors into a single benefit function for a proposed fence.

Current conservation status

The benefits provided by a candidate fencing project can be measured in different ways. In general, we assume that the primary purpose of the fence is to minimise the extinction risk of species that are threatened by invasive predators. This goal is explicitly stated in the relevant state (the NSW National Parks & Wildlife Act 1974), federal (The Environment Protection and Biodiversity Conservation Act 1999) threatened species policy, and international protocols (the IUCN Red List of threatened species, Criterion E). Fences, however, can have other goals, such as the provision of ecosystem services (Miller et al. 2010), the reconstruction of extirpated communities (Shorthouse et al. 2012), or as ecotourism attractions (Daily and Ellison 2012).

We therefore define an extinction probability function $P_e(\boldsymbol{N}_s, T)$ that translates the current distribution of each threatened species to its probability of extinction over a given time period of T years. The vector $\boldsymbol{N}_s$ indicates the current population and distribution of each species:

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$$\boldsymbol{N}_s = \{K_s^1, K_s^2, K_s^3, \dots, K_s^{M_s}\}.$$

(Eq. 1)

Each of the K_s^m values describes the carrying capacity of species s is the m^{th} population, where M_s is the number of existing populations of species s. In highly variable and stochastic environments such as Australia, the fluctuating nature of resources makes carrying capacity difficult to calculate (McLeod 1997). With this in mind we assumed all existing populations to be at their current carrying capacity, recognising that population sizes might change in the future, and this might alter the viability of populations. Ideally the function $P_e(\boldsymbol{N}_s, T)$ would be defined by species- and site-specific population viability analyses, but these are rarely available for even the best-researched threatened species (Reed et al. 2002). In their absence, we choose a general model of species extinction that includes both environmental and demographic stochasticity (Lande 1993, McCarthy et al. 2005). The constant annual probability of extinction of a single population with carrying capacity K is:

$$P_e(\boldsymbol{N}_s = \{K\}, T = 1) = \frac{\sigma^2 b^2}{2K^b}$$

(Eq. 2)

where σ^2 is the variance in the population growth rate (which has a mean of r) and $b = (2r/\sigma^2) - 1$. By assuming that each population is independent, and that the populations are exposed to uncorrelated catastrophic failure (e.g., fence breach, large fire or flood) with annual probability p_c, we can calculate that the probability P_s of a set of populations $\boldsymbol{N}_s$ going extinct in T years is:

$$P_s(\boldsymbol{N}_s, T) = \prod_{m=1}^{M_s} 1 - \left(1 - \frac{\sigma^2 b^2}{2(K_s^m)^b}\right)^T (1 - p_c)^T$$

(Eq. 3)

$$P_s(\boldsymbol{N}_s, T) = \prod_{m=1}^{M_s} 1 - \exp\left(-T\frac{\sigma^2 b^2}{2(K_s^m)^b}\right)(1 - p_c)^T$$

(Eq. 3)

We note that, in extending Eq. 2 to Eq. 3 we have assumed that all the populations are independent, and that the species' extinction will occur when each local population has gone independently extinct. This assumption will be invalid if translocations are commonly used to re-colonise locally extirpated populations. If this assumption does not hold, our estimates of extinction probability will likely be over-estimates. We show this function in Fig. 1 for a range of population sizes, project times and catastrophic extinction probabilities p_c. Throughout the analyses that follow we have assumed a catastrophe probability of $p_c = 0.05$, an environmental variance of $\sigma^2 = 1$, a project period of 20 years, and a maximum per-capita population growth rate based on the estimates of Hone et al.

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(2010). For those NSW species where data did not exist, we substituted values from similar taxa provided by Hone et al. (2010).

Existing representation

Although we perform our analysis at the scale of NSW, we consider the distribution and abundance of each species across Australia in our assessment of complementarity. However, we acknowledge that the NSW government may have different values for species representation within NSW and outside. For example, the distribution of the greater bilby (*Macrotis lagotis*) historically extended into western NSW. Although the species is well represented in Australia's fence portfolio and persists in portions of its historical habitat and therefore considered a low priority, it does not currently persist in land managed by the NSW government. A decision-maker who was only interested in NSW representation could therefore legitimately consider greater bilbies a high priority for a new fence.

We include the existing distribution of each species in other locations by including extant populations of each species in the vector $\boldsymbol{N}_s$. For example, western barred bandicoot (*Perameles bougainville*) are currently extant in four populations across Australia, with populations of 350, 900, 1500 and 500. For this species, this means $M_s = 4$ and $\boldsymbol{N}_s = \{350, 900, 1500, 500\}$.

Probability of translocation failure

We note that the additional fenced population will only contribute to the population viability if the translocation there is successful, and that this is not guaranteed. We therefore estimate for each candidate species, a probability of translocation success q_s. We will assume that this value does not vary between fence sites, but does vary between species. We calculate each species' probability of success based on the observed outcomes of recorded translocations within islands and predator exclusion fences from managers following best practice guidelines, using the mean value of the beta distribution $B(1+\theta, 1+\phi)$ where ϕ is the number of successful translocations and θ is the number of failed translocations (Rout et al. 2009). For those species that have never been translocated, we use the mean probability of the remaining species ($q = 0.73$). The probability of successful translocation for each species is shown in Fig. S2.

Integrating the elements of the benefit function

The expected number of extinctions in T years, across the set of threatened mammal species is calculated as:

$$\langle X \rangle = \sum_{s=1}^{S} P_e(\boldsymbol{N}_s, T)$$

(Eq. 4)

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where S is the total number of listed species. Each candidate fence will create new populations for species suitable for the chosen location, of particular sizes (depending on the suitability of the fenced habitat for those species). This will effectively add a new element to the $\boldsymbol{N}_s$ vectors that correspond to those species for which the fence contains suitable habitat. These new elements, K_s^f, are based on a function of likely population density within a fenced area multiplied by the modelled habitat suitability of each candidate fence location (See Supporting information).

Substituting the new abundance vector into Eq. 1, conditional on successful translocation, we can calculate the expected number of extinctions in the presence of the new fence:

$$\langle X_f' \rangle = \sum_{s=1}^{S} q_s \cdot P_e\left(\{\boldsymbol{N}_s, K_s^f\}, T\right) + \sum_{s=1}^{S} (1 - q_s) \cdot P_e(\boldsymbol{N}_s, T)$$

(Eq. 5)

We therefore recommend that a new fence be placed at the location f which maximises the cost-efficiency of the improvement:

$$\max_f \left[\frac{\langle X \rangle - \langle X_f' \rangle}{C_f} \right]$$

(Eq. 6)

where C_f is the cost of constructing a fence at location f. We note that this particular definition of the objective function should be used in conjunction with a minimum effective action constraint (e.g., that each fence be larger than 2,500 ha, as we do below), to avoid the selection of very small actions with even smaller costs.

Because there are a reasonable and finite number of fence locations, it is possible to identify a single optimal fence location for Eq. 6 by exhaustive search. However, if managers plan to build multiple fences, finding the true optimal solution becomes difficult because the number of options increases combinatorially. When siting multiple fences, we use a greedy search heuristic, re-calculating each of the problem parameters each time. Specifically, after we identify the single best fence, we update the list of each species' populations $\boldsymbol{N}_s$ by adding the new fenced population. We then recalculate the predicted probability of extinction for each species, with and without all possible new fences. However, we no longer consider the site of the first chosen fence, on the assumption that managers will not want to site multiple fences close together, in case a single large-scale stochastic disturbance damages a large part of the network (Helmstedt et al. 2014). In our NSW example we exclude any locations within 25 km of a fence from the analyses.

Current Australian fence network

Australian conservation fence efforts have been previously summarized, but rapid expansion has made these assessments out of date. We focus specifically on the 58 Australian mammal species

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listed as threatened by invasive predators under EPBC and IUCN red list criteria, 22 of which have suitable habitat in NSW (Table S2). Starting with a baseline literature specifically Short (2009), Dickman (2012) and Woinarski et al. (2014), we reviewed the formal scientific literature using Google Scholar and Web of Science searching both the scientific and common names of all listed predator-threatened Australian mammals known to have occurred in NSW (Table S2), and once identified, the names of fence and locations. For small nongovernmental fencing organisations, much of the relevant information occurs outside the peer-reviewed literature. We therefore used internet search engines to search for the scientific name, common name and fence location terms. Once a translocation site was identified, online search and direct contact were used to determine which species had been translocated into each fence, the outcome of the translocation, and current estimates of the fenced abundance.

To assess these data, we first construct frequency histograms summarising fenced protection for all Australian mammals. Then, we use our benefit function to compare extinction risk of each species to the number of known translocation attempts. If the current fencing network were designed to minimise extinctions, we would expect a positive relationship between extinction probability and attempted fence translocations, since an efficient network would prioritise species with wild populations at greater extinction risk. Finally, we contrast IUCN status with number of translocation attempts, expecting that species with higher threat status should attract a greater number of translocation attempts.

Systematic planning of fence network expansion: NSW case study

The state government in NSW is expanding their existing fence network. With this in mind, we applied our benefit function as a search algorithm for identifying locations for new fences that will produce the greatest expected reduction in the number of threatened species extinctions, based on maximising the marginal benefit of each new fence. The state was divided into 30,640 5x5km planning units, each of which a potential new fence project of 2500ha, approximating the NSW criteria for large fences.

We apply two different land tenure constraints. In the first, all tenure types are considered, but only if the cells contain sufficient intact habitat (specifically, no more than 10% of vegetation cleared, as assessed by NVIS version 4.1. In the second, we limit new fences to intact habitat within the current protected area system (CAPAD, 2014). We also consider two different spatial scopes for the project. The first is focused on NSW, and aims to minimise each species' probability of extinction from NSW. That is, based only on current populations within the state. The second considers the probability of global extinction, calculated from all known populations of each species.

For each combination of land tenure constraint and spatial scope, we compare our systematic approach to two reasonable alternative strategies. (1) An uncoordinated, uncooperative approach in which new locations are chosen opportunistically based for example on local funding opportunities or by focusing on individual species. We model this scenario using random selection of the new fence locations. (2) A species-richness approach, where a spatially-flexible organisation chooses new fence locations that maximise the number of species that can persist within the new fence. This

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method ignores complementarity, does not account for the state of the existing fence network, and does not consider the species threat status. All combinations of scenarios and prioritization approaches are summarized in Table S3.

Results

State of the current Australian fence network

Currently there are 30 predator exclusion fences (See Table S1) above 40 hectares in size operating in Australia, managed by 17 different organisations (6 government; 11 non-government/council) containing 31 species. Of these 31 species, only the koala (*Phascolartos cinereus*) is not primarily threatened by invasive predators. The number of fenced translocations is highly skewed in favour of certain species and only half the species threatened by introduced predators are represented (Fig. 1).

Conservation status is not a strong predictor of the species that have been favoured for translocation into fences (Fig. 1). Total population size is not related to the number of fenced translocation attempts (linear regression, $F_{1,57} = 0.11$, $P = 0.74$), and the estimated probability of extinction for NSW species is unrelated to the number of attempted translocations (linear regression, $F_{1,22} = 0.35$, $P = 0.56$; Fig. 1B). The IUCN red list status is essentially independent of the number of translocation attempts (ordinal regression, $P = 0.75$), with the 5 species that received the most translocations ranging from the Critically Endangered woylie (*Bettongia pencillata*), to the Least Concern southern brown bandicoot (*Isoodon obesulus*).

Systematic planning for fence networks

We considered the expansion of the existing fenced network using two different constraints on land tenure (all intact habitat; all protected intact habitat), and with two different objective functions (minimise global extinctions; minimise NSW extinctions). For each of the four scenarios (Fig. 2, Table S3), a systematic approach consistently reduces overall extinction probability more than both random and richness-based (Fig. 3) fence expansions (Fig. 4 & S3).

The systematic approach prioritises fenced sites that support combinations of species with few viable populations elsewhere. Individual species with high returns are characterised by an ability to attain viable populations within the confines of a fence, have a history of successful translocations, and a high risk of extinction. Consequently, a fence site containing only a single, high-risk species can be prioritised over an alternative location containing more species. The construction of a new fence reduces the extinction probability of each translocated species, and this changes the relative value of each potential fence location (Fig. 5).

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Both spatial scope and land tenure strongly influence new fence locations, and which species will benefit. For example, under the Australia-wide objective (Fig. 2c-d; Table S3c-d), the method frequently selects fences that can support the northern hairy-nosed wombat (*Lasiorhinus krefftii*), but never for the NSW objective (Fig. 2a-b; Table S3a-b). This is for three reasons: (1) species extirpated from NSW (but found elsewhere Australia) yield large reductions in extinction probability if a fence creates its first NSW population; (2) the northern hairy-nosed wombat distribution barely overlaps with other threatened species; (3) new wombat populations yield only a low marginal reduction in extinction risk, due to their low population density.

Discussion

Our method describes a systematic method for improving conservation fenced networks. However, our results also underscore and quantify an important conservation policy issue. Fences, like protected areas, will be inefficient if they are not established in a systematic manner (Stewart et al. 2003, Fuller et al. 2010, Radeloff et al. 2013). The decentralised nature of conservation fencing projects makes inefficient outcomes more likely, and our synoptic analyses show the familiar pattern of *ad hoc* conservation actions: when viewed collectively as a network, Australian fences show an over-protection of particular species, no representation for others, and substantial inefficiency. These results do not negate the enormous conservation benefits of fence networks, but rather highlights the potential benefits of coordination and planning. Our tools, which merge systematic conservation planning with population viability analysis, can help reduce these inefficiencies in the future.

Compared with two reasonable alternative strategies, an explicit consideration of both species viability and complementarity can more effectively reduce expected extinctions. For an equivalent investment, systematic choices can improve network performance by as much as factor of 1.8 over random choices and by a factor of 17 over decisions based on species richness (Fig 4, scenario C, Fig S3). Returns asymptote rapidly, suggesting that only a small number of systematically allocated fences are needed to achieve most of the potential gain. This highlights the degree of benefit that can arise from choosing new projects under complementarily frameworks.

The benefits of systematic assessments extend beyond superior performance. A quantitative approach to fence network expansion provides stakeholders with a clear explanation of why a particular choice was made. In an open tender process, an explicit benefit function provides funding organisations with defensibility and rigor, and provides the applicant organisations with a transparent description of the funder's objectives. State or nation-wide priorities may not be wholly applicable to many funding sources for conservation fences, which are locally constrained. Nevertheless, even in these contexts a systematic approach can provide benefits, by quantifying how local actions contribute to broader-scale objectives. This can highlight regional priorities, motivate local fundraising, and help attract regionally-flexible resources. However, while systematic optimisation approaches can support efficient decisions in a decentralised management context,

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they assume that conservation agencies are pursuing a shared goal, and that the actions of independent agencies contribute to this overall goal. In reality, conservation organisations may have diverging goals, or see each other as competitors (Bilodeau and Slivinski 1997). Systematic, efficient decision-making in this context will depend on an explicit understanding of organisations' strategy behaviour (Iacona et.al. 2016).

Our method focused particularly on two essential features of conservation prioritisation (Margules & Pressey 2000): it seeks to represent a range of biodiversity features, and in so doing, offer adequate protection to each. However, the current formulation does not include variation in project cost between sites. The cost of building and maintaining fences varies at fine spatial resolutions, responding to land prices, accessibility, topography, soil type, flood risk, and predator species and densities (Bode et al. 2012, Hayward and Kerley 2009). Variation in cost is therefore an important consideration for fencing projects, and decision-makers may choose to prioritise projects that return the greatest reduction in extinctions given a number of financial constraints. All else being equal, the inclusion of cost will emphasise cheaper species – those that can reach high densities (smaller-bodied), and whose habitat is in low-cost, agriculturally unproductive landscapes. In the absence of data at suitable resolution for the scale of our study, we did not include variation in cost, but acknowledge that it will affect priorities. In an open-tender process, bidding organisations would propose both a location and size for their fence, and would also indicate the cost. Across a large number of bids, this information would allow a calculation of each project's return-on-investment. This could be easily incorporated into the approach.

Our benefit function calculates extinction probability based on both the number of independent populations, and the species' abundance in each. The probability of each fenced population becoming extinct reflects its abundance, its maximum growth rate, and stochasticity (Lande 1993). This general model of population extinction risk can be readily applied to a fenced population of any species, but it clearly makes very simple assumptions about the species' population dynamics, and the processes of local extirpation. Other general models could be used to offer alternative estimates of demographic population viability (Hilbers et. al. 2016, Traill et. al.2007, McCarthy et. al. 2014), however, the assumptions that the Lande (1993) and other models makes about species' population ecology are likely less important than the assumptions they make about management. In fenced populations, active population management (i.e., managed dispersal and recolonisation) can decouple extinction risk from demographics. In fact, species generally go extinct from fenced populations as a result of catastrophic events (e.g., floods, fire, predator incursions), rather than demographic stochasticity. From this perspective, the most appropriate model for extinction risk may be more simple: a benefit function that applies equal weight to all extant populations, regardless of their size. The result is a different set of priority sites (Fig. 2), but a similar improvement in efficiency resulting associated with systematic approaches. These differences stress the importance of correctly formulating the network objectives, and the dynamics of the ecological and economic systems.

Conservation actions are expensive and the available resources are severely constrained. As a result conservation decisions are consistently moving in the direction of systematic and transparent

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prioritisation (Margules and Pressey 2000, Joseph et al. 2009, Januchowski-Hartley et al. 2011, Pannell et al. 2012). Conservation fencing, an increasingly common threatened species management approach, is a rare exception to the trend of systematic prioritisation. Our method adds to the existing toolkit, with potential application to any spatially-constrained management action that aims to provide population viability benefits to a limited suite of species such as poison baiting programs, weed control, population monitoring or island prioritisations. Our Australian case-study highlights the value of applying systematic approaches to networks of conservation fences, with similar benefits likely to be observed across the increasing set of conservation fencing networks across the globe (Hayward and Somers 2012).

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Figure 1

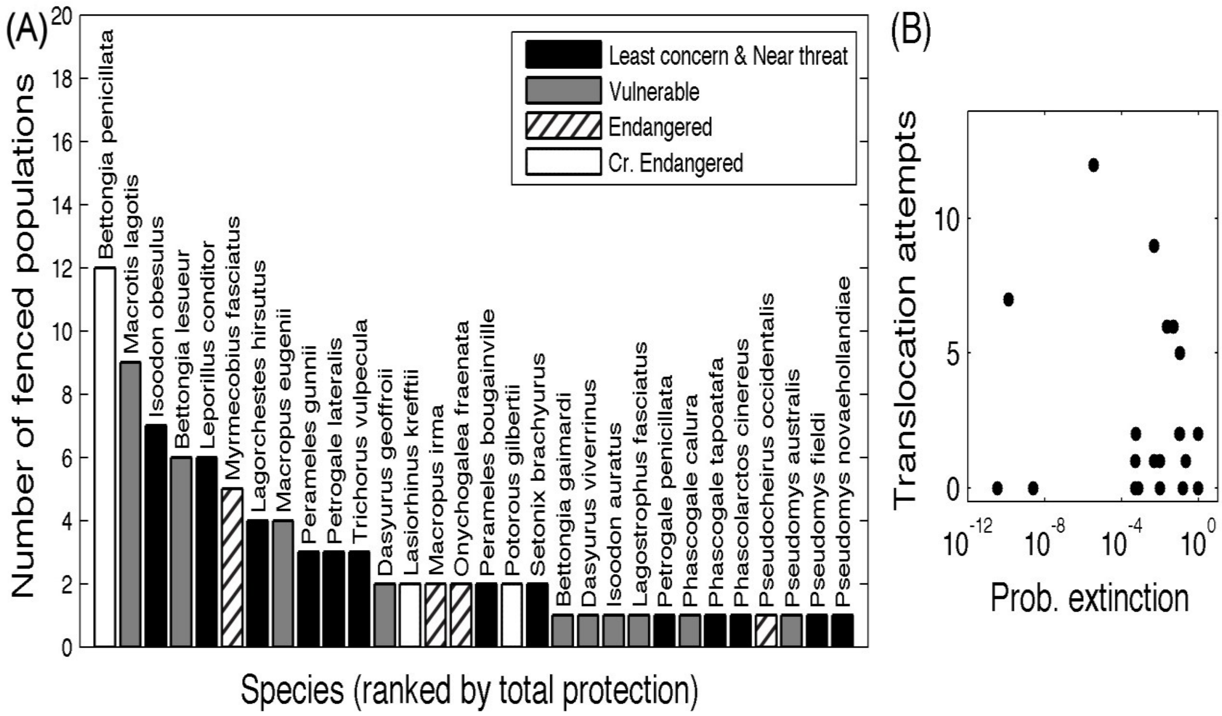

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Figure 2

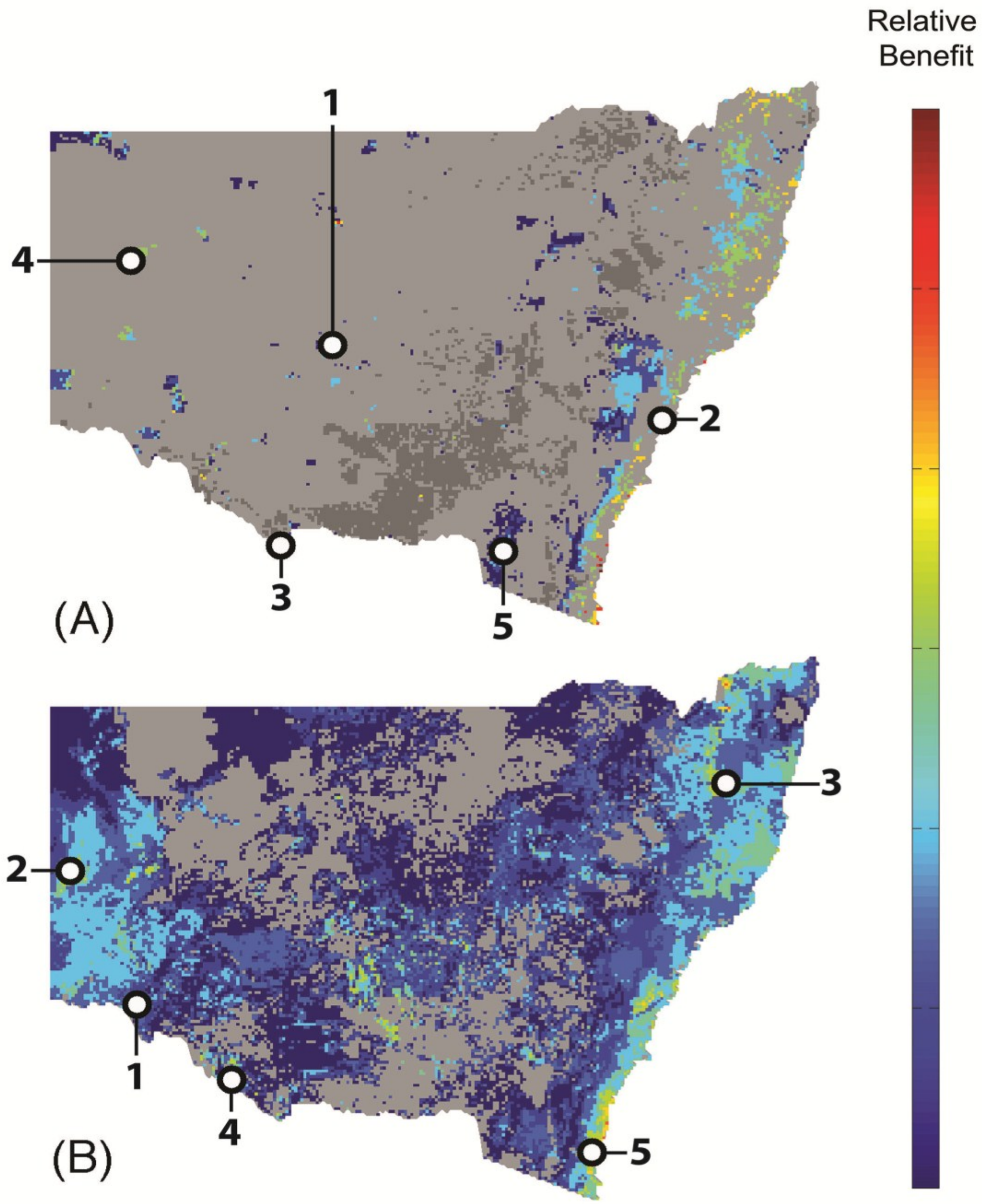

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Figure 3

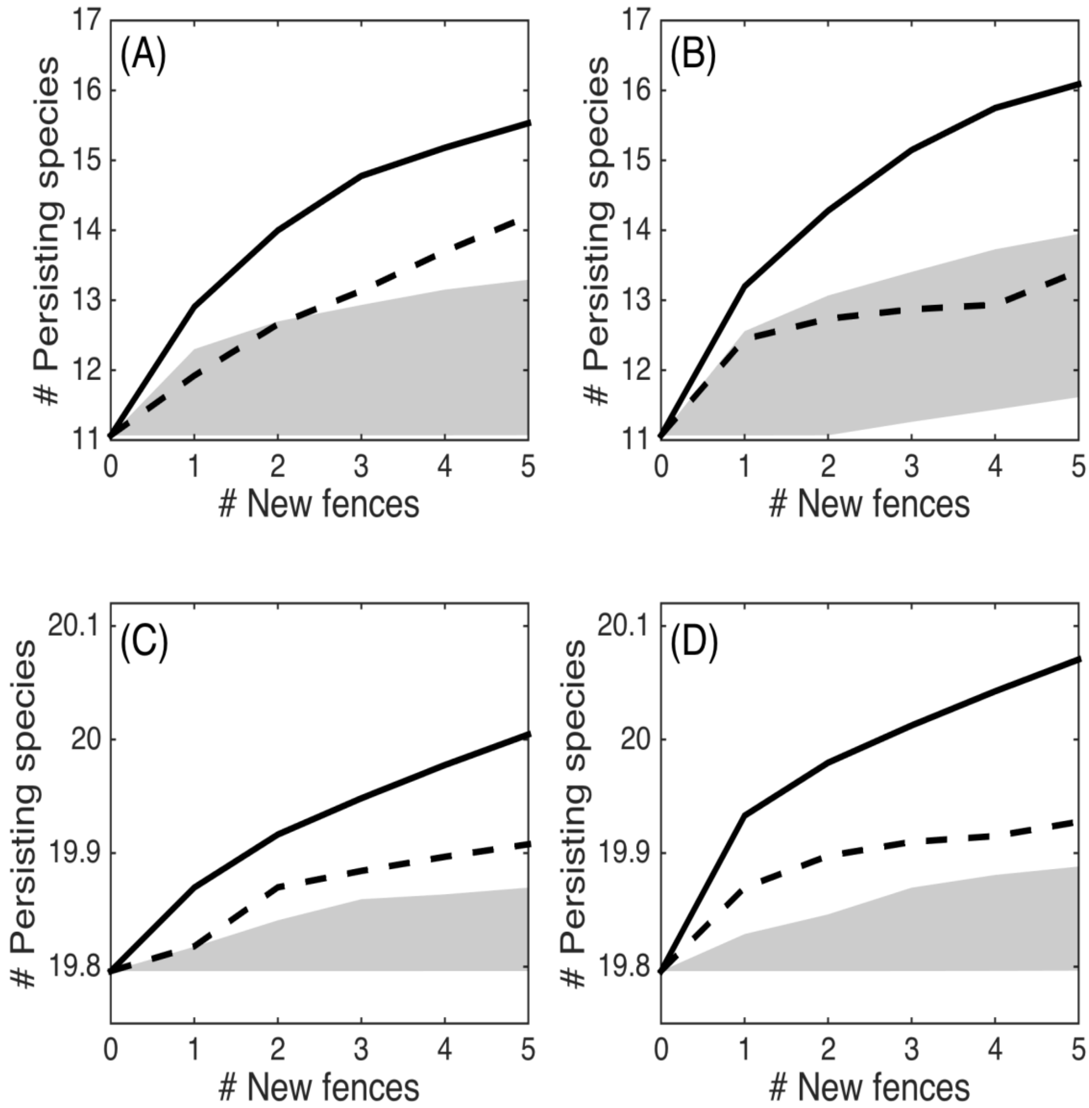

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Figure 4

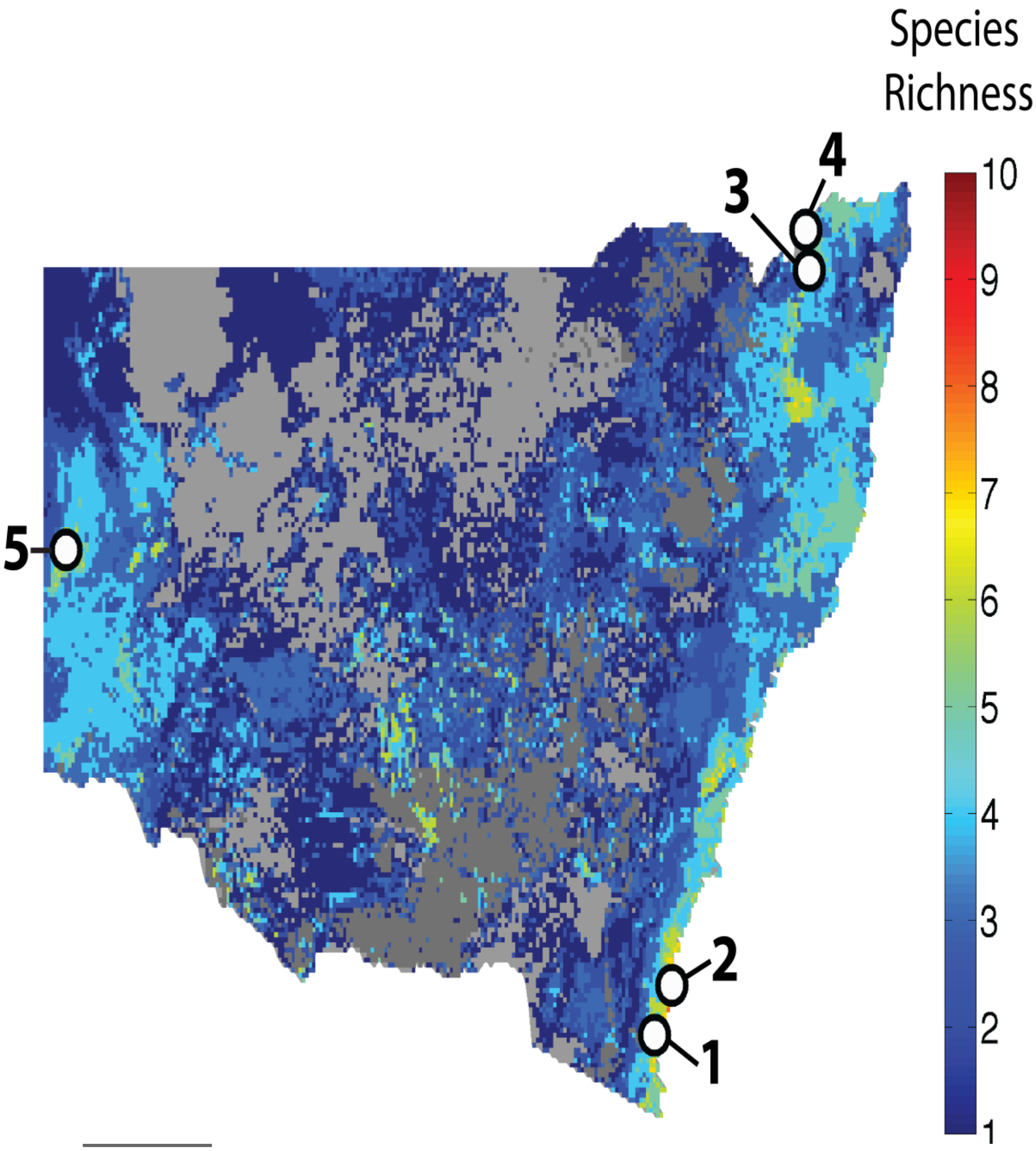

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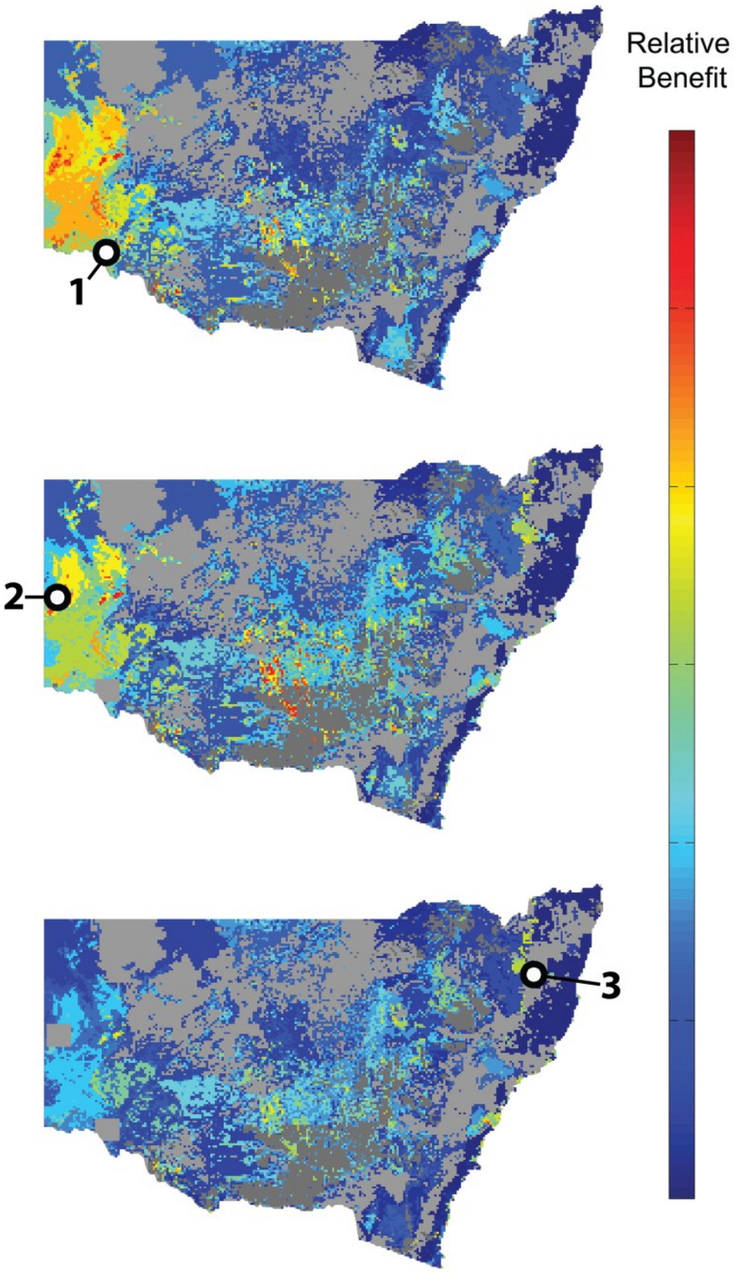
Figure 5



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