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. We conducted a gap analysis of Australia’s large predator-exclusion fence network by examining translocation of Australian mammals relative to their extinction risk. To address gaps identified in species representation, we devised a systematic prioritization method for expanding the conservation fence network that explicitly incorporated population viability analysis and minimized expected species’ extinctions. The approach was applied to New South Wales, Australia, where the state government intends to expand the existing conservation fence network. Existing protection of species in fenced areas was highly uneven; 67% of predator-sensitive species were unrepresented in the fence network. Our systematic prioritization yielded substantial efficiencies in that it reduced expected number of species extinctions up to 17 times more effectively than ad hoc approaches. The outcome illustrates the importance of governance in coordinating management action when multiple projects have similar objectives and rely on systematic methods rather than expanding networks opportunistically.

Keywords: invasive alien species, population viability, predator exclusion fence, spatial optimization, translocation

Minimización de las Extinciones de Especies por medio de la Planeación Estratégica del Enrejado de Conservación

Resumen: Los cercos de conservación son una acción de manejo cada vez más común, particularmente para las especies amenazadas por los depredadores invasores. Sin embargo, a diferencia de muchas acciones de conservación, las redes de cercos se están expandiendo de manera unisistémica, generalmente como reacción a las oportunidades de financiamiento local o a las amenazas. Realizamos un análisis de vacío de la red de cercos de exclusión de depredadores en Australia al examinar la traslocación de los mamíferos australianos en relación con su riesgo de extinción. Para abordar los vacíos identificados en la representación de las especies diseñamos un método de priorización sistemática para expandir la red de cercos de conservación que incorporaban explícitamente el análisis de viabilidad poblacional y minimizaban las extinciones esperadas de las especies. La estrategia se aplicó en Nueva Gales del Sur, Australia, donde el gobierno del estado pretende expandir la red existente de cercos de conservación. La protección existente de las especies en las áreas cercadas fue muy desigual; el 67% de las especies sensibles a los depredadores estuvo mal representado en la red de cercos. Nuestra priorización sistemática produjo eficiencias sustanciales ya que redujo el número esperado de extinciones de especies hasta 17 veces más efectivas que las estrategias ad hoc. El resultado ilustra la importancia de la gobernanza en la coordinación de la acción de manejo cuando múltiples proyectos tienen objetivos similares y dependen de los métodos sistemáticos en lugar de expandir las redes de manera oportunista.

Palabras Clave: cerca de exclusión de depredadores, especies no-nativas invasoras, optimización espacial, traslocación, viabilidad poblacional

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Introduction

Invasive predators are a leading driver of global biodiversity decline and loss (Mack et al. 2000; Clavero & García-Berthou 2005), particularly in ecosystems where prey species are evolutionarily naive. 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 & Veitch 2002; Rejmánek & 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 density of an introduced predator.

Conservation fencing is a rapidly expanding management action (Hayward & 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 & 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 organizations can reintroduce species with a rate of success comparable to translocations on predator-free islands (Short 2009). Conservation fences are consequently popular (Hayward & Kerley 2009), even with small organizations.

Relative to single organizations, economic theory suggests that sectors made up of more diverse, independent organizations are better able to adapt to local environmental and sociopolitical conditions; to access diverse funding sources and local volunteers; to lower operating and transaction costs; and to experiment and innovate (Bilodeau & Slivinski 1997; Albers & 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 create an organizationally decentralized fence network. In many instances, fencing projects are rapidly arising independent of each other. In Australia, for example, the majority (58%) (Supporting Information) of fences are operated by nongovernmental organizations or local councils. The same decentralization occurs in New Zealand, where 78% of fences are nongovernmental (Saunders & Norton 2001; Burns et al. 2012). This situation is unusual for conservation in these 2 countries, whose political systems and history of land tenure has seen the majority of protected-area designations under-taken by state or federal governments (Saunders & Norton 2001; Burns et al. 2012).

Decentralization in conservation can result in unsystematic, uncoordinated actions, and costly inefficiencies (Pressey et al. 1993), incomplete protection (Margules & Pressey 2000), and enormous legacy costs (Stewart et al. 2007; Fuller et al. 2010). Such inefficiencies are also likely to be 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 & 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, relative to reserve systems (see Barr et al. 2016), the decentralized organization and funding structure of fence networks mean that inefficiencies will be difficult to correct through top-down control. Nevertheless, coordination has the potential to substantially increase the performance of fence networks. New methods are therefore required to identify and prioritize new fencing projects.

Australia is a global epicenter 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 recognized 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 minimize species extinction probability. This focus on viability is therefore an essential element of fence-network planning. To prioritize additional fencing projects in this context requires systematic methods.

Based on theories of population viability and systematic conservation planning, we designed and implemented a systematic method to evaluate the current performance of a network of fences and quantified the relative benefits of alternative future fence projects. This method may be applicable to various fence networks, such as those in Hawaii (Cole et al. 2012), Africa (Hayward & Kerley 2009), and New Zealand (Burns et al. 2012). Similarly, it may be equally applicable to the prioritization of threatened species management actions generally. To illustrate our approach, we considered Australia’s network of predator-exclusion fences built for the conservation of threatened mammals in New South Wales. We reviewed the state and performance of Australia’s existing network of predator exclusion fences and assessed whether the network exhibited the inefficiencies expected from such a decentralized structure. We devised a flexible,
systematic method to optimally expand existing fence networks and applied it to a New South Wales case study, where the state government is planning 2 new conservation-fence projects. As in systematic conservation planning, we sought a method that would delineate an efficient and complementary network of fences.

Methods

Goals and Objective Function

A number of methods have been developed for the spatial prioritization 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 prioritization methods have great overlap with the spatial prioritization of conservation fencing, but the formulation of the optimization problem differs in 3 important ways. First, fencing is typically used as a crisis management action, where the goal of action is to recover species on the brink of extinction. Prioritizing 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 percent representation. Second, locally extirpated species are almost always translocated into fenced areas (Dickman 2012), rather than populations remaining 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, because 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 areas for fencing with the goal of minimizing expected extinctions across a suite of species. Our new systematic approach for choosing priority locations for fencing projects is based on this objective.

To consider fences in a network context, we needed to choose fence locations that would provide the greatest aggregate benefit to conservation. Thus, fences had to be sited in areas that would provide suitable habitat for a large number of species that are threatened by invasive predators (Supporting Information). 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 modified the value of each species by taking into account their current conservation status. Second, we corrected the species richness of each site based on the existing representation of those species in conservation projects elsewhere (i.e., we included complementarity). Finally, we considered the risk of full or partial project failure, a serious and acknowledged problem for threatened species translocations (Short 2009; IUCN/SCC 2015). We integrated 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 assumed the primary purpose of the fence was to minimize the extinction risk of species threatened by invasive predators. This goal is explicitly stated in the relevant state (the NSW National Parks & Wildlife Act 1974) and federal (The Environment Protection and Biodiversity Conservation Act 1999) threatened species policies and in international protocols (International Union for Conservation of Nature criterion E [IUCN 2015]). Fences, however, can have other goals, such as the provision of ecosystem services (Miller et al. 2010) and ecotourism attractions (Daily & Ellison 2012) and reconstruction of extirpated communities (Shorthouse et al. 2012).

We therefore defined an extinction probability function \( P_e(N_s, T) \) that translates the current distribution of each threatened species to its probability of extinction over a given period of \( T \) years. The vector \( N_s \) indicates the current population and distribution of each species:

\[
N_s = \{ K_1^s, K_2^s, K_3^s, \ldots, K_M^s \},
\]

where each of the \( K_i^s \) values describes the carrying capacity of species \( s \) is the \( m \)th population and \( M_s \) 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). Thus, we assumed all existing populations were at their current carrying capacity but recognized that population sizes may change in the future and that this could alter the viability of populations. Ideally, the function \( P_e(N_s, T) \) 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:

\[
P_e(N_s = \{ K \}, T = 1) = \frac{\sigma^2 b^2}{2K^b},
\]

where \( \sigma^2 \) 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 $p_s$, we calculated that the probability $P_s$ of a set of populations $N_s$ going extinct in $T$ years as

$$P_s(N_s, T) = \prod_{m=1}^{M_s} \left( 1 - \left[ 1 - \frac{\sigma^2 b^2}{2(K_s^m)^b} \right] (1 - p_s)^T \right)$$

and

$$P_s(N_s, T) = \prod_{m=1}^{M_s} \left( 1 - \exp \left( -T \frac{\sigma^2 b^2}{2(K_s^m)^b} \right) (1 - p_s)^T \right).$$

In extending Eqs. (2) and (3), we assumed that all the populations are independent and that extinction will occur when each local population has been independently extirpated. This assumption will be invalid if translocations are commonly used to recolonize locally extirpated populations. If this assumption does not hold, our estimates of extinction probability are likely overestimates. We show this function in Fig. 1 for a range of population sizes, project times, and catastrophic extinction probabilities ($p_s$). Throughout our analyses we have assumed a catastrophe probability of $p_s = 0.05$, 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. (2010). For those New South Wales species for which data did not exist, we substituted values for similar taxa provided by Hone et al. (2010).

**Existing Representation**

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

We included the existing distribution of each species in other locations by including extant populations of each species in the vector $N_s$. For example, western barred bandicoot ($Perameles bougainville$) are currently extant in 4 populations across Australia (350, 900, 1500, and 500 individuals). For this species, this means $M_s = 4$ and $N_s = (350, 900, 1500, 500)$.

**Probability of Translocation Failure**

The additional fenced population will contribute to population viability only if the translocation there is successful, and success is not guaranteed. We therefore estimated for each candidate species a probability of translocation success ($q_s$). We assumed this value does not vary between fence sites but does vary between species. We calculated 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. We used the mean value of the beta distribution $B(1 + \theta, 1 + \phi)$, where $\phi$ is the number of successful translocations and $\theta$ is the number of failed translocations (Rout et al. 2009). For those species that have never been translocated, we used the mean probability of the remaining species ($q = 0.73$). The probability of successful translocation for each species is provided in Supporting Information.

**Integrating the Elements of the Benefit Function**

We calculated the expected number of extinctions in $T$ years, across the set of threatened mammal species as

$$\langle X \rangle = \sum_{s=1}^{S} P_s(N_s, T),$$

where $S$ is the total number of listed species. For species suitable for the chosen location, each candidate fence creates new populations of particular sizes (depending
on the suitability of the fenced habitat for those species. This effectively adds a new element to the $N_t$ 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 modeled suitability of each candidate fence location (see Supporting Information). Substituting the new abundance vector into Eq. (1), conditional on successful translocation, we calculated the expected number of extinctions in the presence of the new fence:

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

(5)

We therefore recommended a new fence be placed at location $f$ because it maximises the cost-efficiency of the improvement:

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

(6)

where $C_f$ is the cost of constructing a fence at location $f$. This particular definition of the objective function should be used in conjunction with a minimum effective action constraint (e.g., that each fence encompass over 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 used a greedy search heuristic that recalculated each of the problem parameters each time. Specifically, after we identified the single best fence, we updated the list of each species’ populations ($N_s$) by adding the new fenced population. We then recalculated the predicted probability of extinction for each species with and without all possible new fences. However, we no longer considered the location of the fence chosen first because we assumed managers do 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 New South Wales example, we excluded locations within 25 km of a fence from the analyses.

Current Australian Fence Network

Australian conservation fence efforts have been summarized, but rapid expansion has dated these assessments. We focused specifically on the 58 Australian mammal species listed as threatened by invasive predators under EPBC and IUCN Red List criteria, 22 of which have suitable habitat in New South Wales (Supporting Information). Starting with Short (2009), Dickman (2012), and Woinarski et al. (2014) as a baseline, we reviewed the peer-reviewed scientific literature through searches in Google Scholar and Web of Science. We searched for scientific and common names of all listed predator-threatened Australian mammals known to have occurred in New South Wales (Supporting Information). Once identified, fence-location names were searched for to identify additional species they may have contained. For small nongovernmental organizations that manage fenced areas, much of the relevant information was not in the peer-reviewed literature. Therefore, we 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 abundance of fenced species.

To assess these data, we constructed frequency histograms that summarized fencing 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 minimize extinctions, we expected a positive relationship between extinction probability and attempted fence translocations because an efficient network prioritizes species with wild populations at greater extinction risk. Finally, we contrasted 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 in New South Wales

The New South Wales government is expanding their existing fence network. With this in mind, we applied our benefit function as a search algorithm to identify locations for new fences that would produce the greatest expected reduction in the number of threatened species extinctions, based on maximizing the marginal benefit of each new fence. The state was divided into 30,640 5 × 5 km planning units, each of which contained a potential new fence project of 2500 ha, approximating the state’s criteria for large fences.

We applied 2 different land-tenure constraints: all tenure types were considered but only if the cells contained sufficient intact habitat (specifically, no more than 10% of vegetation cleared, as assessed by Australian Government Department of the Environment. Nation Vegetation Information System version 4.1)
and new fences were limited to intact habitat within the current protected area system (CAPAD, Commonwealth of Australia 2014). We also considered 2 different spatial scopes for the project: minimize each species’ probability of extirpation from New South Wales, based only on current populations within the state, and minimize species’ probability of global extinction, calculated from all known populations of each species.

For each combination of land-tenure constraint and spatial scope, we compared our systematic approach with 2 reasonable alternative strategies. First, an uncoordinated, uncooperative approach in which new locations were chosen opportunistically based for example on local funding opportunities or by focusing on individual species. We model this scenario with random selection of the new fence locations. Second, a species-richness approach, in which a spatially flexible organization chooses new fence locations that maximize the number of species that can persist within the new fence. This method ignores complementarity, does not account for the state of the existing fence network, and does not consider the species threat status. These combinations of scenarios and prioritization approaches are summarized in Supporting Information.

Results

State of the Current Australian Fence Network

In Australia at the time of our study, there were 30 predator-exclusion fences (Supporting Information) that enclosed over 40 ha. These fenced areas were managed by 17 different organizations (6 government; 11 non-governmental or council) and contained 31 species. Of these 31 species, only the koala (Phascolartos cinereus) was not primarily threatened by invasive predators. The number of fenced translocations was highly skewed in favor of certain species, and only half the species threatened by introduced predators were represented (Fig. 1).

Conservation status was not a strong predictor of the species that have been favored for translocation into fenced areas (Fig. 1). Total population size was 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 New South Wales species was unrelated to the number of attempted translocations (linear regression, $F_{1,22} = 0.35, p = 0.56$) (Fig. 1b). The IUCN Red List status was essentially independent of the number of translocation attempts (ordinal regression, $p = 0.75$); the 5 species that received the most translocations ranged from the critically endangered woylie (Betongia penicillata) to the least concern southern brown bandicoot (Isoodon obesulus).

Systematic Planning for Fence Networks

Under all scenarios of land tenure and regional priorities (Fig. 2 & Supporting Information), our systematic method of fence expansion more effectively reduced extinction risk than either random or richness-based (Fig. 3) fence expansions (Fig. 4). The systematic approach prioritized fenced sites that supported combinations of species with few viable populations elsewhere. Individual species with high returns were characterized by an ability to attain viable populations within the confines of a fence and had a history of successful translocations and a high risk of extinction. Consequently, a fence site containing only a single, high-risk species was prioritized over an alternative location containing more species. The construction of a new fence reduced the extinction probability of each translocated species, and this changed the relative value of each potential fence location (Fig. 5).

Figure 2. In New South Wales, the locations of the 5 fenced areas most beneficial to conservation of native mammals (a) as defined by the summed expected reduction in species extinction rate (fences are in protected areas only and are based on species’ population status) (dark gray, protected areas where translocations would not occur) and (b) in all areas with sufficient remaining vegetation based on Australia-wide population status (benefit: red, highest; blue, lowest).
Figure 3. Relative performance over 20 years of different strategies to expand conservation fencing based on 3 methods: systematic minimization of expected extinctions (black line), greatest number of unique species that can persist in each fenced area (dashed lines), and random selection of new fenced areas (grey [extremes are 95% bounds]). Benefit was measured in terms of (a, b) the expected number of species persisting within New South Wales and (c, d) globally relative to the number of species expected to exist in the absence of fencing. Fenced areas in (a) and (c) are in New South Wales protected areas only, and fenced areas in (b) and (d) are in any intact habitat in New South Wales.

Figure 4. Locations of the 5 conservation fencing projects that most improve species richness (based on the summed probability-of-occurrence maps of New South Wales threatened species) when as many species as possible are added to each fenced area, ignoring complementarity with existing network and subsequent fences. This figure contrasts with results of the systematic but noncomplementary approach depicted in Fig. 5, where fence locations are based on predicted reduction in extinction risk.

Both spatial scope and land tenure strongly influenced new fence locations and which species benefited. For example, under the Australia-wide objective (Figs. 2c & 2d; Supporting Information), the method frequently selected fence locations that supported the northern hairy-nosed wombat (Lasiorhinus krefftii), but under the New South Wales objective the method never selected locations that supported this species (Figs. 2a & 2b; Supporting Information). This occurred because species extirpated from New South Wales (but found elsewhere Australia) yield large reductions in extinction probability if a fence creates its first state population; the northern hairy-nosed wombat distribution barely overlaps with other threatened species; and new wombat populations yielded only a low marginal reduction in extinction risk because of their low population density.

Discussion

We devised a systematic method to improve conservation fencing 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 decentralized 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 overprotection of particular species, no representation for others, and substantial inefficiency. These results do not negate the enormous conservation benefits of conservation fence networks. Rather, they highlight
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Figure 5. Locations of the 3 new fenced areas that most reduce species extirpations in New South Wales (red, most reduction; blue, least reduction). All land with intact vegetation was considered for potential fence locations. The sequence of panels shows how the relative benefit changes as new fenced areas are added to the existing portfolio. For example, locations with the biggest initial benefit decrease in value after the addition of the new fence decreases the expected extirpation probability of the species that it contains.

The potential benefits of coordination and planning. Our tools, a merger of systematic conservation planning with population viability analysis, can help reduce these inefficiencies in the future.

Compared with 2 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. 3 & Supporting Information). 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 complementarity frameworks.

The benefits of systematic assessments extend beyond superior performance. A quantitative approach to expansion of the fence network provides stakeholders with a clear explanation of why a particular choice was made. In an open-tender process (where organizations provide proposals to funders), an explicit benefit function provides funding organizations with defensibility and rigor and the applicant organizations with a transparent description of the funder’s objectives. State- or nationwide fencing priorities may not be shared by funders or locally constrained conservation organizations. Nevertheless, a systematic approach can still provide benefits by quantifying how local actions contribute to broader-scale objectives. Systematic approaches can be used to highlight regional priorities, motivate local fundraising, and help attract regionally flexible resources. However, although systematic optimization approaches can support efficient decisions in a decentralized management context, 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 organizations may have diverging goals or see each other as competitors (Bilodeau & Slivinski 1997). Systematic, efficient decision making in this context will depend on an explicit understanding of organizations’ strategy behavior (Iacona et al. 2016).

Our method focused particularly on two essential features of conservation prioritization (Margules & Pressey 2000): representation and thus protection of a range of biodiversity features. However, this formulation does not include variation in project cost between sites. The cost of building and maintaining fences varies at fine spatial resolutions in response to land prices, accessibility, topography, soil type, flood risk, and predator species and densities (Hayward & Kerley 2009; Bode et al. 2012). Differences in cost are therefore an important consideration for fencing projects, and decision makers may prioritize projects that return the greatest reduction in extinctions given a number of financial constraints. All else being equal, the inclusion of cost will emphasize cheaper species – those that can reach high densities (smaller bodied) and whose habitat is not of high value for other purposes such as agriculture. In the absence of data at suitable resolution for the scale of our study, we did not include variation in cost, but we acknowledge it will affect priorities. In an open-tender process, bidding organizations 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 can be used to calculate 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 (Traill et al. 2007; McCarthy et al. 2014; Hilbers et al. 2016); however, the assumptions that Lande (1993) and others make in their models 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 recolonization) 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, and 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 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 prioritization (Margules & 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 prioritization. Our method adds to the existing toolkit and is potentially applicable to any spatially constrained management action that aims to provide population viability benefits to a limited suite of species, such as poisoning programs, weed control, population monitoring, and island prioritizations. Our Australian case study highlights the value of applying systematic approaches to networks of conservation fences; similar benefits are likely to be observed across the increasing global set of conservation fencing networks (Hayward & Somers 2012).

Supporting Information

A list of fences and management organizations, a list of species, a summary of scenarios, a map of translocation suitability, probability of translocation success, and number of new fences required to achieve the same benefit as systematically chosen fences (Appendix S1) and a description of the species-distribution modeling process (Appendix S2) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

Literature Cited


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