Methods in Ecology and Evolution

Methods in Ecology and Evolution 2013, **4**, 819–827 doi: 10.1111/2041-210X.12072

Interior fences can reduce cost and uncertainty when eradicating invasive species from large islands

Michael Bode[1]*, Karl E. C. Brennan[2], Kate Helmstedt[3], Anthony Desmond[4], Raphael Smia[1,5] and Dave Algar[6]

[1]*Australian Research Council Centre of Excellence for Environmental Decisions, School of Botany, University of Melbourne, Parkville, Victoria 3010, Australia;* [2]*Nature Conservation Division, Western Australian Department of Environment & Conservation, Locked Bag 104, Bentley Delivery Centre, Geraldton, WA, 6983, Australia;* [3]*Australian Research Council Centre of Excellence for Environmental Decisions, School of Biological Sciences, University of Queensland, St Lucia, Queensland 4072, Australia;* [4]*Regional Services Division, Western Australian Department of Environment & Conservation, PO Box 72, Geraldton, WA, 6531, Australia;* [5]*Ecole Polytechnique ParisTech, 91761 Palaiseau, France; and* [6]*Science Division, Western Australian Department of Environment & Conservation, PO Box 51, Wanneroo, WA 6946, Australia*

Summary

1. The conservation of many threatened species can be advanced by the eradication of alien invasive animals from islands. However, island eradications are an expensive, difficult and uncertain undertaking. An increasingly common eradication strategy is the construction of 'interior fences' to partition islands into smaller, independent eradication regions that can be treated sequentially or concurrently. Proponents argue that, while interior fences incur substantial up front construction costs, they reduce overall eradication costs. However, this hypothesis lacks an explicit theoretical or empirical justification.

2. We formulate a general theory that relates the number of interior fences to the magnitude and variation of the economic cost of island eradication. We use this theory to explore the conditions under which interior fences represent a defensible management strategy, under cost and risk minimisation objectives. We then specifically consider the forthcoming eradication of cats *Felis catus* from Dirk Hartog Island, Western Australia, by parameterising our general theory using published data on the cost and success of previous projects.

3. Our results predict that under a wide range of reasonable conditions, interior fences can reduce the expected cost of a successful invasive alien animal eradication from large islands. On Dirk Hartog Island, interior fences will marginally reduce eradication costs, with two fences reducing expected costs by 3%. Interior fences have a much more substantial effect on the variability of eradication costs: two fences reduce the width of the 95% confidence bounds by more than one-third and halve the size of the average project cost overrun/underrun.

4. Our results reveal that the construction of interior fences is a defensible management strategy for eradicating alien invasive species from islands. However, the primary benefit of interior fences will be risk management, rather than a reduction in expected project costs.

Key-words: feral animals, conservation fencing, cost effectiveness analysis, restoration, return on investment, invasive species, cats, *Felis catus*

Introduction

Islands are critical to global biodiversity conservation (Courchamp, Chapuis & Pascal 2003; Mittermeier *et al.* 2005; Kier *et al.* 2009). Islands contain a high proportion of the globe's narrow range endemics (Carlquist 1974; Mittermeier *et al.* 2005), support remnant populations of many species that were once widely distributed (Burbidge *et al.* 2008) and will be the translocation destination of species that are threatened with extinction in mainland habitats (Abbott 2000; Algar, Johnson & Hilmer 2011). The high conservation value of these islands depends on their threat-free status; most frequently this means the absence of invasive alien species (IAS), and particularly invasive alien predators (Carlquist 1974; Blackburn *et al.* 2004; Sax & Gaines 2008). The removal of IAS from islands has therefore become a key focus of scientific research and substantial international conservation investment (Burbidge & Morris 2002; Courchamp, Chapuis & Pascal 2003; Towns & Broome 2003; Campbell *et al.* 2011). Alien vertebrate species have been successfully eradicated from more than 1160 islands, at an average rate of 20 eradications per year since 1980 (Keitt *et al.* 2011; Island Conservation 2012).

In recent years, IAS have been eradicated from islands of increasing area (Brooke, Hilton & Martins 2007a; Courchamp *et al.* 2011), including islands that were once explicitly considered 'too large' by several orders of magnitude (Courchamp, Chapuis & Pascal 2003). This progress is the result of advances in technology, and also improvements in eradication tactics;

*Correspondence author. E-mail: mbode@unimelb.edu.au

one such tactical innovation is the construction of 'interior fences'. By constructing fences that partition an island, managers reduce a large eradication project into a series of smaller, independent eradications that can be undertaken concurrently or sequentially (Parkes & Panetta 2009). An 'interior fence' strategy proved effective during the successful eradication of feral pigs *Sus scrofa* from California's Santa Catalina and Santa Cruz Islands, which were divided into four and five regions, respectively, before eradication was undertaken (Schuyler, Garcelon & Escover 2002; Morrison 2007). Interior fences were also constructed during the eradication of cattle *Bos taurus* from Amsterdam Island (Micol & Jouventin 2002), and sheep *Ovis aries* from Campbell Island (Rudge 1986). They are currently being planned as part of a multispecies eradication on Stewart Island, New Zealand (Beaven 2008), and as part of the world's largest feral cat *Felis catus* eradication programme on Dirk Hartog Island, Western Australia (Algar, Johnson & Hilmer 2011), hereafter referred to as DHI.

The optimality of an interior fencing strategy hinges on the assumption that it is more expensive to eradicate IAS from a single large region, than from two regions of half the size (Usher 1989). While plausible, this proposition has not been theoretically justified nor empirically demonstrated. An island's size is known to affect both the probability of eradication and the project cost, regardless of the species being eradicated or the island's features and location (Martins *et al.* 2006; Pluess *et al.* 2012b). Larger eradication projects have a higher probability of failure because they offer greater opportunities for errors (e.g. areas accidentally left unbaited), because the last individuals will be at lower density, and because larger spatial scales encompass greater environmental heterogeneity (Choquenot, Hone & Saunders 1999; Pluess *et al.* 2012a,b). Eradication costs will increase with island area primarily because treatment is area dependent (e.g. baiting density), although the rate at which costs increase with area may decrease due to economies of scale (Towns & Broome 2003; Martins *et al.* 2006). However, it is unclear whether the combination of these scale effects supports the construction of interior fences. The analyses we present here have three primary aims. First, by integrating the effects of spatial scale on both eradication costs and probability of success, we will formulate a general theory for island eradication using interior fences. We will use this theory to explore and describe the conditions, if any, under which interior fences are a cost-effective strategy. Secondly, for a specific project – cat eradication from Dirk Hartog Island – we will identify the optimal number of interior fences for two plausible management objectives: minimising the expected cost of successful eradication, and reducing the probability that project costs are either dramatically above or below the estimated cost. Finally, we will quantify the magnitude of the improvement in both performance metrics that would be expected under the optimal interior fencing strategy.

General model of interior fencing

We construct a general model for optimal interior fencing as follows. Consider a rectangular island with width w and total area A that hosts an IAS population (the model is very similar for other regular island geometries, such as circular). Managers intend to eradicate this species using a given technique (e.g. a particular baiting strategy, or a combination of baiting and trapping), that is defined by two factors. The first is the per unit area cost of applying the technique, c_T, which we denote the 'treatment cost'. Note that this is not the same as the cost of eradicating species from the island, c_E, which we call the 'eradication cost', since not all attempts will be successful. The second factor is the probability that an eradication attempt will succeed, p_E. Both $c_T(A)$ and $p_E(A)$ are functions of the area being treated. The managers are also planning a series of F interior fences that will partition the island into $(F + 1)$ separate regions.

If the managers choose not to construct any internal fences ($F = 0$), their eradication treatment will be applied to the entire island at once. If the first treatment is successful, the eradication cost will be $c_E = c_T(A)A$. However, the first treatment will be unsuccessful with probability $(1-p_E(A))$, as will any subsequent reapplication. If managers persist with successive treatments until the IAS is eradicated, then the expected cost of successful eradication will be:

$$\langle c_E(F=0)\rangle = c_T(A)p_E(A)A\sum_{k=0}^{\infty}(k+1)(1-p_E(A))^k = \frac{c_T(A)A}{p_E(A)}. \qquad \text{eqn 1}$$

Equation 1 makes the conservative assumption that each eradication attempt is independent of previous attempts, since the large majority of effort is generally expended removing the last individuals (Myers, Savoie & van Randen 1998). Nevertheless, follow-up eradication treatments could also cost less than initial treatments or have a higher probability of success, particularly if failures are detected very early.

Alternatively, the managers could construct a single internal fence ($F = 1$) to bisect the island into two regions of equal area that are treated separately. If the fence effectively separates the two regions, the expected eradication cost will be:

$$\langle c_E(F=1)\rangle = 2\frac{c_T\left(\frac{A}{2}\right)\frac{A}{2}}{p_E\left(\frac{A}{2}\right)} + c_F w + 2c_B = \frac{c_T\left(\frac{A}{2}\right)A}{p_E\left(\frac{A}{2}\right)} + c_F w + 2c_B, \qquad \text{eqn 2}$$

where c_F is the per kilometre cost of constructing a fence (materials and labour), w is the width of the island and c_B is the additional cost of building fencing structures where the fence meets the island coastline. The first term on the right-hand side of eqn 2 indicates that a single interior fence essentially creates two islands of half the size of the original, which must be eradicated independently. Finally, if managers fence the island into more than two regions, the expected eradication cost using a F fence strategy is:

$$\langle c_E(F)\rangle = \frac{c_T\left(\frac{A}{F+1}\right)A}{p_E\left(\frac{A}{F+1}\right)} + F(c_F w + 2c_B), \qquad \text{eqn 3}$$

where each fence runs the width of the island. The number of fences that minimise the expected cost of eradication, F^*, can be found by choosing the value of F that minimises eqn 3. Given that eradication is uncertain, the eradication cost will be negative binomially distributed about this expected cost.

Parameterisation the interior fence model for DHI

At least 12 species of mammal (many threatened nationally) are locally extinct on DHI (62 790 ha), a land bridge island in the Shark Bay World Heritage Area (WHA) (Fig. 2). The primary driver of these extinctions is likely to be IAS: both alien predators (especially cats) and grazing degradation of habitat (sheep and goats *Capra aegagrus*). All have been implicated in the demise of mainland mammals (McKenzie *et al.* 2007). Furthermore, there is a strong relationship between the presence of mammalian IAS on islands and the local extinction of Australian mammals (Abbott & Burbidge 1995). As with many islands (Courchamp, Chapuis & Pascal 2003), pastoralists likely brought cats to DHI in the mid-19th century (Algar, Johnson & Hilmer 2011). The eradication of cats from DHI will allow reintroductions to proceed and will reduce predation pressure on threatened species still extant on DHI (a skink and three endemic bird subspecies) (Algar, Johnson & Hilmer 2011).

While cats have been successfully eradicated from more than 80 islands worldwide, DHI is much larger than previous eradication campaigns, with more than twice the area of the largest eradication to date (Marion Island in the Indian Ocean, at 29 000 ha). Eradication of cats from DHI will therefore be challenging (Burbidge & Morris 2002; Algar, Johnson & Hilmer 2011), but the project satisfies four key criteria for successful eradication (Parkes 1990). First, managers consider the eradication achievable based on the demonstrated success of the planned treatment in similar environmental contexts (Algar *et al.* 2010), and pilot studies on DHI itself (Algar, Johnson & Hilmer 2011). Secondly, all cats on the island will be exposed to treatment. Baits will be deployed aerially across the island on a pattern and timing that optimises the likelihood of all individual cats encountering a bait, in a season when they are hungry. A monitoring programme will detect cats that survive aerial baiting, so that these individuals are targeted for trapping and/or hand baiting (Johnston *et al.* 2010). Thirdly, the density of baits is expected to kill cats at a greater rate than their reproduction. Efficacy trials on DHI show that aerial baiting kills 82–90% of cats (Johnston *et al.* 2010), which is high enough to eradicate a cat population with an intrinsic increase rate of 166% per year (an estimate from within the Shark Bay WHA; Short & Turner 2005). Finally, there is a very low chance of recolonisation. At 1·6 km from the Australian mainland, the risk of cats swimming or drifting to DHI will be low, since there are many islands further than 1 km from the Australian mainland that are without cats (Abbott 2000). To facilitate this eradication, managers intend to partition the island into more manageable sections of smaller area. We consider the optimal strategy and potential benefits of interior fences on DHI using the general methods described earlier.

To parameterise the optimal interior fencing formula described in eqn 3, we need to estimate the treatment cost, c_T, and the probability of that attempt successfully eradicating cats from a given area, p_E. Direct estimates of these parameters are not available, and we must therefore extrapolate their values from data on past eradications. These data are from two sources: the first is a data set on the outcomes of 87 cat eradication projects over the past few decades (Campbell *et al.* 2011). The data set reports the success or failure of each project, and a small amount of geographical information on the island, such as its total area. In the absence of more specific information, we assume that these outcomes are the result of a single eradication attempt. The second data set is a nine project subset of the first data set, for which the cost of successful projects had been recorded. We use these data to estimate c_T as a function of island area.

These two data sets contain a relatively small and possibly biased sample of eradications (particularly for the smaller cost data set) and describe projects on islands that are much smaller than DHI. As a result, the predictions and conclusions we base on our model will also be uncertain. Despite this fact, a modelling approach provides useful insights for three reasons. First, a quantitative model of eradication can clarify and sharpen our understanding of how interior fences will alter management outcomes. By demanding explicit statements about the expected costs and benefits, a process-based model of eradication reduces ambiguity by testing the logical consistency of the assumptions behind interior fences. Secondly, we can still make statistically robust management recommendations if we propagate this uncertainty through to our management conclusions. We use nonparametric bootstrapping to generate confidence intervals around our estimates of treatment cost and the probability of success and then use combinations of the resultant bounds to assess how uncertainty influences the optimal interior fencing decision. Finally, the transparent treatment of uncertainty required by a modelling approach emphasises research and data gaps, identifying which are most important from a management perspective.

TREATMENT COST

We estimate treatment costs for DHI without a fence, based on previous costed eradications (Campbell *et al.* 2011). Tactically, DHI eradication will combine aerial delivery of monofluoroacetate ('1080') poison baits and ground-based trapping (see Algar, Johnson & Hilmer 2011), a strategy similar to many examples from this data set. Figure 2a shows project cost data for nine cat eradications that were successful, and that reported project costs (none used interior fences), which we assume are the result of a single treatment. Superimposed on these data are a previously published cost model for vertebrate eradication (Martins *et al.* 2006) and a linear regression to these cost data (a higher-order polynomial was not supported by AIC).

Extrapolating these fits to an island area of 62 790 ha, these methods estimate that a single eradication treatment on DHI will cost USD 4·9 M according to the Martins *et al.* (2006)

model. Our linear model estimates treatment costs at USD 11·5 M, with 95% confidence intervals of USD $3·7 M and USD $24·2 M. Given that the Martins *et al.* (2006) estimates fall within the confidence bounds of the linear model, we henceforth consider only the best-fit linear model and its confidence bounds. We note, however, that the previous eradication programme most similar to DHI was an eradication of cats from Faure Island (Fig. 1), which has an area of 5800 hectares (Algar *et al.* 2010). This eradication, the result of a single treatment, cost USD $33 700 (corrected for inflation), which lies outside the 99% confidence intervals of the best-fit model.

FENCE CONSTRUCTION COSTS

Managers intend to construct an east-west fence that will partition the island into two management regions. The cost for fence materials and construction is estimated at USD $28 000 per km. The two ends of any such fence – where it meets the ocean – are particularly vulnerable to incursions (Long & Robley 2004). We therefore add an additional USD $10 000 at each boundary of each fence to account for the construction costs of additional structures.

PROBABILITY OF SUCCESS

To complete the parameterisation of eqn 3, we need to estimate the probability that a treatment will result in successful eradication given that it is attempted over a particular area. Campbell *et al.* (2011) documented the outcome of cat eradications from more than 100 different islands, and we have fitted a logistic function to the relationship between total island area and outcome (Fig. 2b). Although these data collate several decades of eradication projects, recent analyses suggest that the probability of successful eradication is not changing through time (Parkes & Panetta 2009). The logistic fit to these data indicates that, as expected, the probability of eradication decreases with increasing area. This relationship provides a strong motivation for constructing interior fences on DHI, since it indicates that cat eradication will be easier in smaller, fenced subregions than over the entire island. However, the 95% confidence bounds indicate how our uncertainty about eradication success also increases drastically as the model moves outside the range of historical eradications. For an island of the size of DHI, these confidence bounds span almost the entire range of probabilities. At the model's upper confidence bound, island size has almost no effect on eradication success and an eradication treatment on DHI is 92% likely to succeed. In contrast, at the lower confidence bound, each treatment on DHI is projected to eradicate cats with a probability of only 29%.

OPTIMAL STRATEGY WITH RECURRENT FENCE BREACHES

The model encapsulated in eqn 3 is predicated on the assumption that interior fences are impregnable. In reality, fence breaches are not unusual events, both because animals continually test the integrity of the new structures (Connolly, Day & King 2009), and because environmental events can cause unavoidable damage (Long & Robley 2004). The reinvasion of previously eradicated areas will likely reduce the cost savings and risk mitigation benefits offered by interior fences and is therefore a factor that must be considered when assessing interior fences on DHI (Bode & Wintle 2010).

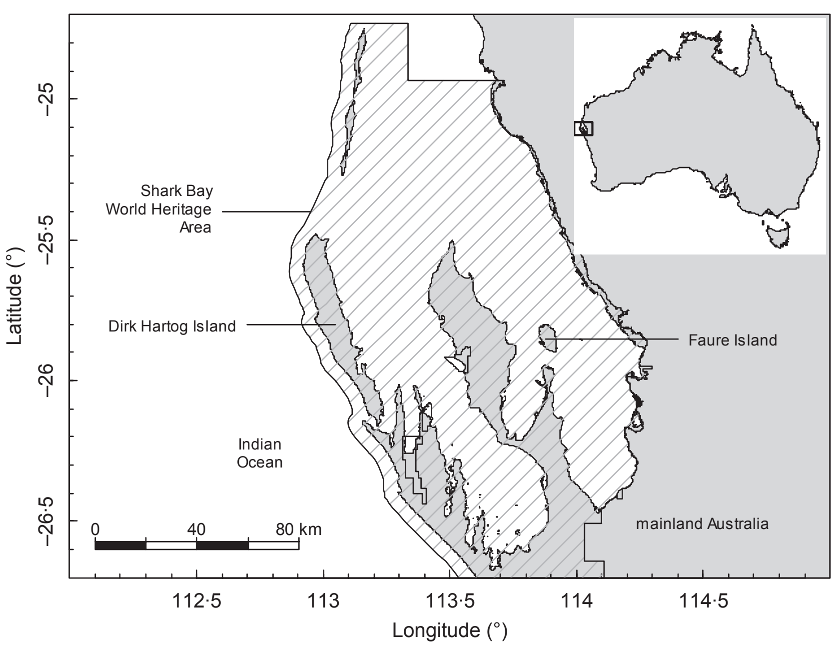

Fig. 1. Location of Dirk Hartog Island off the West Australian coast, the site of the proposed interior fence eradication. Map also indicates the location of Faure Island.

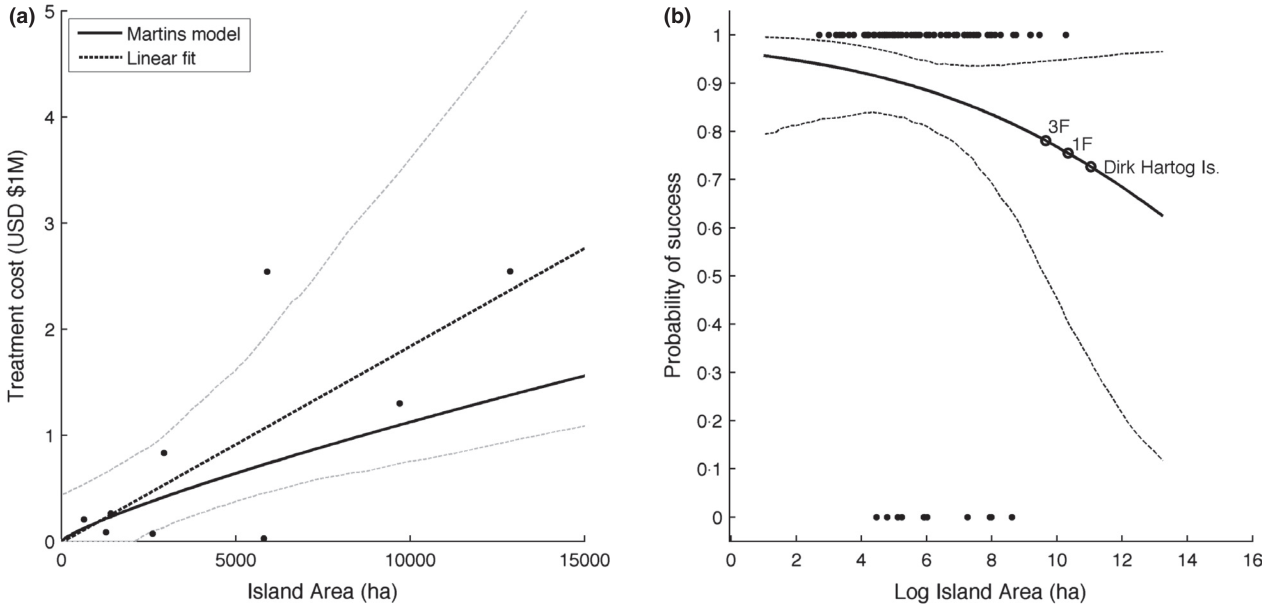

Fig. 2. (a) Estimates of treatment costs for cats as a function of island area. Circles indicate data on costed projects. Dashed line indicates best-fit linear model, with 95% bootstrap confidence intervals shown with dotted lines. Solid line shows the model parameterised by Martins *et al.* (2006). (b) Relationship between probability of successful eradication and island area. Closed circles show data on the success of known cat eradication projects. Solid line shows the best-fit logistic function; dashed lines show 95% bootstrap confidence intervals. Open circles on the line of best-fit show the probability of successful eradication in a DHI region with no fences, one fence (1 *F*) and three fences (3 *F*).

We therefore reformulate our model to allow for the probability that a fence is breached while IAS are being eradicated from one region. We assume for simplicity that the probability of fence breach is related to the length of fence separating the two areas, rather than the size of each fenced area. However, it is possible that because smaller eradications are likely to be completed more rapidly, they permit fewer breach opportunities. We again assume that the fences are used to partition the island into a sequence of regions, which are eradicated in a sequence that begins at one end of the island. During the treatment of each region (starting with the second region), animals can breach the fence into the previous region with probability p_B. If this occurs, we assume that the managers immediately become aware of the fact and return to the reinvaded region to reapply treatment (they are therefore always treating the edge of the invaded region).

When interior fences can be breached, we can derive an equation for the expected number of treatments needed to successfully eradicate IAS by treating the project as a 'random walk'. Managers begin the random walk in state $x = F + 1$, corresponding to IAS being present in all fenced regions. Eradicating from one fenced region, without any breaches, moves the state one step to the left (with probability $\lambda = (1 - p_B)p_E\left(\frac{A}{F+1}\right)$); a failed eradication attempt and a fence breach will move the state one step to the right (with probability $\rho = p_B\left(1 - p_E\left(\frac{A}{F+1}\right)\right)$). Complete island eradication is analogous to the random walk reaching the point $x = 0$, which is an absorbing barrier because the invasive species has been completely eradicated from the island. The state $x = F + 1$ is a reflecting barrier, because no additional breaches can take place (the whole island is already invaded). We assume that $p_E \neq 1$ (i.e. there is some chance the treatment will be unsuccessful). When there are F fences, the expected number of steps required to reach the absorbing boundary is (Weesakul 1961) as follows:

$$\langle N \rangle = \begin{cases} \frac{\rho}{(\lambda-\rho)^2}\left(\left(\frac{\rho}{\lambda}\right)^{F+1} - 1\right) + \frac{(F+1)}{\lambda-\rho} & \text{if } p_E \neq p_B, \\ \frac{(F+1)(F+2)}{2\lambda} & \text{if } p_E = p_B. \end{cases} \qquad \text{eqn 4}$$

The expected cost of successful eradication will then be the following:

$$\langle c_E(F) \rangle = \langle N \rangle c_T\left(\frac{A}{F+1}\right)\frac{A}{F+1} + F(c_F w + 2c_B). \qquad \text{eqn 5}$$

We can use this expression to calculate the optimal number of interior fences in a similar manner.

ANALYSIS OF THE DHI MODEL

Equation 3 allows us to calculate the expected cost of eradicating feral cats from DHI for any number of unbreachable interior fences. We apply this approach to the DHI parameterisation, with the number of fences varying between 0 and 40, and identify the integer value of F that minimises the expected eradication costs. The confidence intervals around our best estimates of treatment costs and success also allow us to consider the influence of parameter uncertainty on our management strategies. Combining these two sets of confidence bounds, we construct an 'optimistic' eradication scenario (the upper 95% confidence interval for eradication success with the lower 95% confidence interval for treatment costs), and a 'pessimistic' scenario (the lower and upper bounds of success and treatment cost, respectively).

We consider the decision about whether to build an interior fence from the perspective of two managers with different objectives: (i) a 'cost minimising' manager, who wants to eradicate cats at the lowest expected cost, and (ii) an 'accurate' manager, who wants to minimise the magnitude of a cost overrun or underrun. The first objective would reflect an individual who is risk neutral and therefore is not adverse to cost overruns as long as they are balanced by underruns. This would be the likely position of a conservation manager with control over a wide range of projects, whose performance is therefore assessed by the average outcome across all of their projects. The second objective might belong to a manager who will be negatively assessed if their projections of the project cost are inaccurate – even if they are underestimates. This is an attitude towards risk known as 'uncertainty aversion'. A manager with this objective function would be willing to undertake a strategy that will cost more on average, if they could make more accurate estimates of project cost by doing so. The manager of a single eradication project is more likely to be uncertainty averse, since the variance of their projections will be naturally very large. Finally, for DHI we consider the outcome of eradication efforts with nonzero probabilities for each fence being breached, ranging between 0% and 7%.

Results of the general model

Our analyses of the general interior fencing model support the use of interior fencing as a cost-effective tactic in island eradications. Figure 3 shows that as the total island area increases, and as the relative cost of fence construction to eradication treatment declines, the optimal number of interior fences increases. For reasonable estimates of island area (1 km^2–1000 km^2), and reasonable ratios of fencing costs to treatment costs (between 1 : 100 and 1 : 20), the construction of up to five interior fences is required to minimise the expected cost of successful IAS eradication.

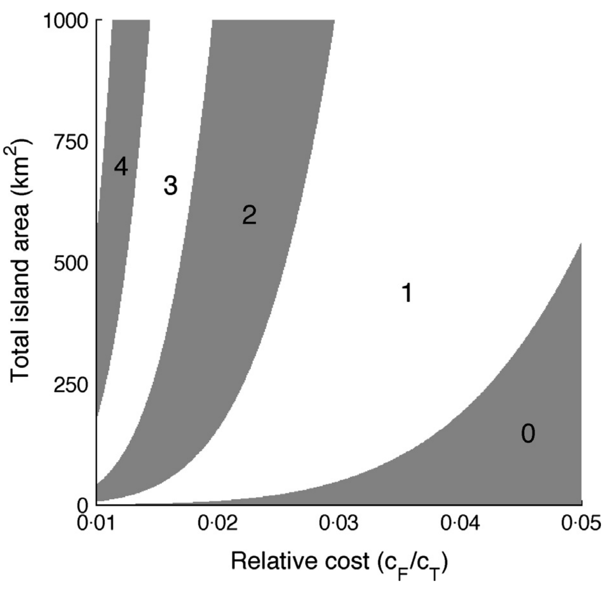

Fig. 3. Optimal number of interior fences for a range of costs and island areas. Within each shaded region, a given number of interior fences minimises the expected cost of successful eradication. Costs are measured by the relative cost of constructing an interior fence (c_F) and the cost of a single eradication treatment (c_T).

Results for the DHI application

By substituting our DHI parameter estimates into eqn 3, and using the best estimates of the relationships for p_E and c_E, we find that managers can best reduce the expected costs of feral cat eradication on DHI by constructing either one or two interior fences (Fig. 4a). Moreover, managers should expect an overall reduction in eradication costs from interior fencing strategies that involve up to five interior fences (Table 1). Specifically, a two fence strategy marginally reduces the expected cost of cat eradication by USD $500 000, which is approximately 3% of the expected costs that would be incurred without an interior fence. The construction of interior fences also reduces the width of the 95% confidence intervals, with most of the reduction affecting the upper confidence bound. That is, interior fences dramatically reduce the magnitude of cost overruns. The uncertainty surrounding our best estimates of treatment cost and success probability greatly affect the optimal number of interior fences. Under our 'optimistic' scenario, the eradication on DHI will not only be achievable at a significantly lower cost, it will be most cost effectively implemented without any interior fences (i.e. the cost is minimised when $F^* = 0$). In contrast, under our 'pessimistic' scenario of high treatment costs and low success probabilities, managers should construct as many as 18 interior fences (Fig. 4a).

In comparison with a 'cost minimising' manager, an 'accurate' manager would be predisposed towards constructing a large number of conservation fences. Figure 4b shows the average size of the difference between a manager's estimate of the project cost and its realised cost. In this analysis, the variation reflects both the uncertainty in our estimates of treatment cost and success, and also the possibility that treatments will fail. It is clear from Fig. 4b that, as well as reducing the expected cost of eradication, the construction of interior fences reduces the variation in eradication costs around their expected value. The realised cost of an eradication project on DHI that constructs two interior fences will cost within USD $5·6 M of the expected cost, whereas an eradication project without any

Table 1. Predicted costs of successfully eradicating cats from DHI, for zero to five interior fences

Number of interior fences (F)	Expected eradication cost (USD $M)	Lower 95% cost confidence bound (USD $M)	Upper 95% confidence bound (USD $M)
0	15·9	4·3	81·0
1	15·5	4·6	62·3
2	15·4	5·0	54·8
3	15·6	5·3	51·1
4	15·7	5·6	48·7
5	15·9	5·9	46·6

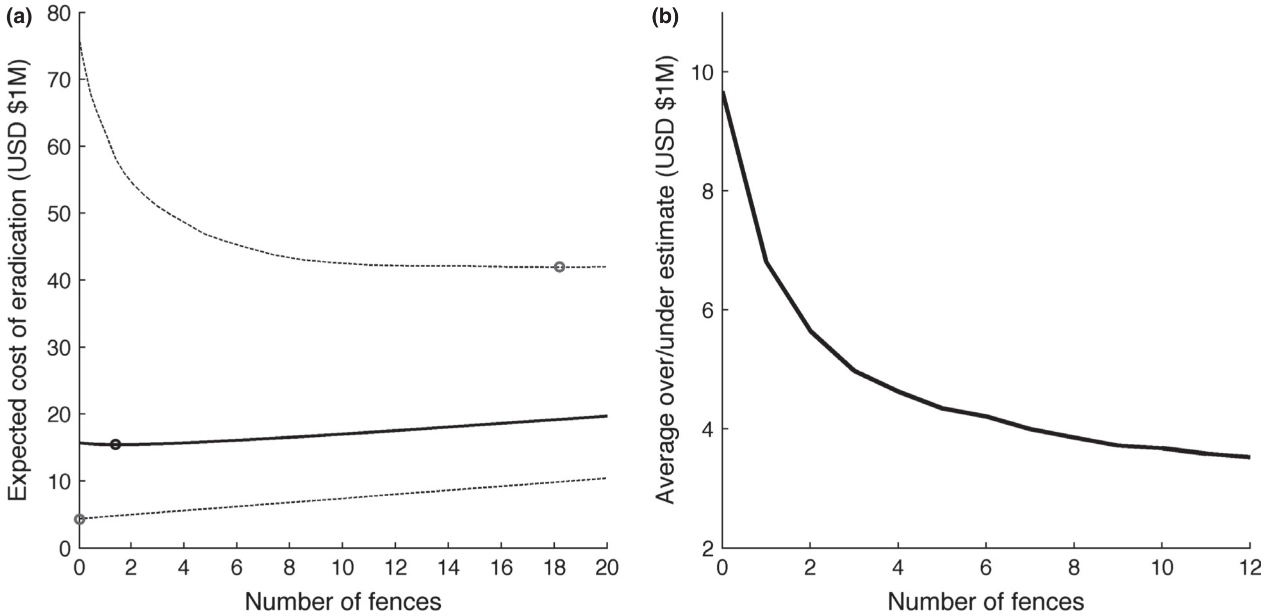

Fig. 4. (a) Expected cost of successful eradication under the three scenarios for a given number of interior fences. Solid line shows the best-fit parameterisation; lower and upper dashed lines show the optimistic and pessimistic eradication scenarios, respectively. Circles denote the number of interior fences that minimise the expected eradication costs under each scenario. (b) The average magnitude of a project cost overrun or underrun, for a given number of interior fences. A cost deviation is measured by the absolute difference between the expected eradication cost and the realised eradication cost.

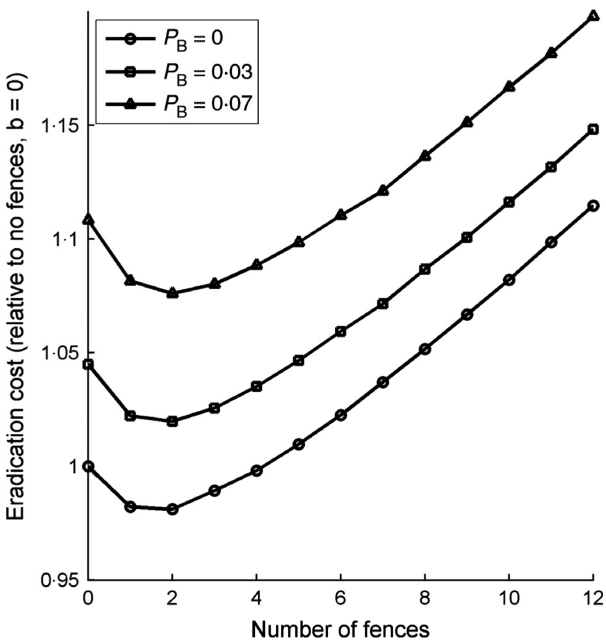

Fig. 5. Relative expected cost of successful eradication on DHI for a given number of interior fences, when fences can be breached with three different probabilities, *pB*. Costs given (*y*-axis) are relative to the cost of DHI eradication without fences, when the probability of breach is zero.

interior fences will cost within USD $9·8 M of the expected cost. Thus, in comparison with no fences, two interior fences will reduce the magnitude of a cost overrun or underrun by 43%. An accurate manager would prefer to construct as many interior fences as feasible, since the average over/under estimate continues to decline monotonically as the number of fences increases.

The decision to construct interior fences is quite robust to model uncertainty, as well as the parametric uncertainty indicated by the confidence bounds. If we estimate treatment costs using the Martins *et al.* (2006) nonlinear model, rather than the more parsimonious linear model (Fig. 2a), the expected cost of eradication is minimised by a single interior fence. If we recalculate the optimal strategy allowing for a nonzero probability of fence failure (eqns 4 and 5), interior fences are still required to minimise the expected cost of cat eradication for three different breach probabilities (Fig. 5). Managers should choose to construct two interior fences if they are less confident in the security of each fence. Changes in the probability of a breach parameter also have no effect on the optimal number of fences.

Discussion

Our general model indicates that interior fences can lower the expected cost of island eradication by converting a single large island into a series of smaller regions, from which eradication is more likely on any given attempt (Fig. 3). With a fixed conservation budget, an interior fencing strategy will enable the creation of more IAS-free islands and is therefore justified. For our DHI parameterisation, the optimal interior fencing strategy reduces the cost of eradication by 3%, or approximately USD $500 000. Moreover, analyses such as that presented here can help explain and justify management decisions to auditors and stakeholders, a key requirement for expensive management actions such as fencing (Ferraro & Pattanayak 2006; Bode *et al.* 2012).

A much more significant potential benefit of interior fences is their effect on budgetary risk. As well as reducing expected eradication costs, interior fences drastically reduce the magnitude of project cost overruns and underruns, allowing managers to predict project costs with much more certainty. This reduction is achieved for a number of reasons. First, once interior fences have been constructed, a failed eradication attempt no longer incurs the cost of retreating the entire island. Secondly, interior fences make the eradication status of separate regions independent; the project therefore spreads the risk of any misfortune. The converse is also true; however, interior fences also reduce the probability that projects will be unexpectedly inexpensive. The lower confidence intervals in Fig. 4a are monotonically increasing with additional interior fences. Finally, eradications from large regions are outside management experience and are therefore highly uncertain (Fig. 2). Interior fences create management regions that more closely resemble previous eradication projects, where treatment costs and success probabilities can be predicted with more certainty.

Even for smaller areas than DHI, the limited data available on island eradication costs and successes creates considerable uncertainty, reducing our ability to accurately predict management outcomes. This is demonstrated by the wide confidence bounds around our estimates (Figs 2 and 4a), and the variability in observed outcomes (e.g. the Faure Island eradication). We note, however, that this uncertainty is primarily due to poor information about the cost and success of eradication projects, not the interior fence strategy. This is apparent in the width of the confidence intervals when there are no interior fences (i.e. when $F = 0$). The poor quality of eradication data has been highlighted in the past (Brooke, Hilton & Martins 2007b; Donlan & Wilcox 2007; Pluess *et al.* 2012b) and is particularly pertinent for islands outside the range of previous eradications. When these broad uncertainty bounds are incorporated into a decision support model, the resultant management predictions vary considerably. Nevertheless, an explicit acknowledgement of this uncertainty has two key benefits. First, by highlighting our limited understanding of island eradication, this uncertainty provides a strong justification for additional research that focuses specifically on the factors that determine the cost and success of island eradication projects. Secondly, a quantitative treatment of uncertainty can itself provide management guidance. For example, it is the uncertainty itself that provides the most compelling rationale for interior fences on DHI. While interior fences will slightly lower the expected cost of eradication, their primary benefit will be to reduce the magnitude of the uncertainty.

Moreover, when our assumptions were relaxed to incorporate known uncertainties – either in our parameter estimates or in our assumptions about fence breaches – interior fences were still part of the optimal eradication strategy. However, uncertainties that we did not include could potentially have altered this conclusions. For example, our analyses have not captured all the potential environmental costs of utilising fences. On DHI we did not consider the local degradation of habitat caused by clearing vegetation to build fences, which on sand islands may create the risk of soil instability. As we are unable to estimate the risk of such an event we did not include it in our modelling, but it would act to reduce the optimal number of interior fences. If the use of an interior fencing strategy extends the duration of the eradication project (e.g. each fenced region is tackled in a separate year), the additional logistic costs would also undermine the efficiency of fences. Such fixed costs could be readily included in our method by changing c_F in eqn 3.

An interior fencing strategy also offers benefits that our analyses did not acknowledge. From the perspective of uncertainty, an interior fencing strategy will provide learning opportunities that are not available when entire islands are treated as single regions. Specifically, the effect of island size on eradication success and costs will be a crucial determinant of relative island priority (Brooke, Hilton & Martins 2007a), and as we show here, will influence the performance of alternative eradication strategies. By partitioning islands into multiple smaller regions, eradications with interior fences will produce data on a larger number of eradication areas, and thus allow more rapid learning. If the chosen fence locations create regions with different sizing – either artificially or to take advantage of natural features – the resulting data could offer powerful statistical insight into the true functional forms shown in Fig. 2. Different eradication regions will also allow experimental comparisons of different eradication techniques. The resulting information benefits will further increase the attractiveness of an interior fence approach to island eradication and can be incorporated using an active adaptive management framework (Walters 1986). Finally, interior fences create a beneficial 'ratchet' effect, where incremental and sporadic eradication gains can eventually add up to complete eradication because they are unlikely to be reversed. This ratchet effect allows managers to hedge against the risk of a project being temporarily suspended while only partially complete, due to funding uncertainty (McBride *et al.* 2007), environmental variability or legal challenges (Morrison 2007). It also allows eradication attempts using techniques that are limited to small spatial scales (e.g. by personnel constraints) or short temporal scales (e.g. summer in the Subantarctic).

Island eradications have increased in scale, speed and technology over the last two decades (Parkes & Panetta 2009). With the aim of ensuring successful, cost-effective eradication from large islands, managers are increasingly considering interior fencing strategies. In this paper, we have provided a process-based rationale for the construction of interior fences, and quantitatively assessed whether existing data support this approach. Distilled into models of treatment cost and the probability of success, the outcomes of previous eradication projects suggest that interior fences can reduce the total cost of eradication projects, and greatly reduce project uncertainty. For large islands, multiple interior fences will be an effective conservation investment.

Acknowledgements

This research was funded by the Australian Government's National Environmental Research Program, the ARC Centre of Excellence for Environmental Decisions, an ARC DECRA (DE130100572), and the Western Australian Department of Environment and Conservation. We thank Ashley Millar, John

Asher and participants of the 2011 Shark Bay fencing workshop for ideas and perspectives.

References

Abbott, I. (2000) Improving the conservation of threatened and rare mammal species through translocation to islands: case study Western Australia. *Biological Conservation*, **93**, 195–201.

Abbott, I. & Burbidge, A. (1995) The occurrence of mammal species on the islands of Australia: a summary of existing knowledge. *CALM Science*, **1**, 259–324.

Algar, D., Johnson, M. & Hilmer, S.S. (2011) A pilot study for the proposed eradication of feral cats on Dirk Hartog Island, Western Australia. *Island Invasives: Eradication and Management* (eds C.R. Veitch, M.N. Clout & D.R. Towns), pp. 10–16. IUCN, Gland, Switzerland.

Algar, D., Angus, G.J., Brazell, R.I., Gilbert, C. & Withnell, G.B. (2010) Eradication of feral cats on Faure Island, Western Australia. *Journal of the Royal Society of Western Australia*, **93**, 133–140.

Beaven, B. (2008) *Scoping the Potential to Eradicate Rats, Wild Cats and Possums from Stewart Island/Rakiura* (ed. N.Z.D.o. Conservation). New Zealand Department of Conservation, Invercargill, New Zealand.

Blackburn, T.M., Cassey, P., Duncan, R.P., Evans, K.L. & Gaston, K.J. (2004) Avian extinction and mammalian introductions on oceanic islands. *Science*, **305**, 1955–1958.

Bode, M. & Wintle, B. (2010) How to build an efficient conservation fence. *Conservation Biology*, **24**, 182–188.

Bode, M., Brennan, K.E.C., Morris, K., Burrows, N. & Hague, N. (2012) Choosing cost-effective locations for conservation fences in the local landscape. *Wildlife Research*, **39**, 192–201.

Brooke, M., Hilton, G. & Martins, T. (2007a) Prioritizing the world's islands for vertebrate-eradication programmes. *Animal Conservation*, **10**, 380–390.

Brooke, M., Hilton, G.M. & Martins, T. (2007b) The complexities of costing eradications: a reply to Donlan & Wilcox. *Animal Conservation*, **10**, 157–158.

Burbidge, A. & Morris, K. (2002) Introduced mammal eradication for nature conservation on Western Australian islands: a review. *Turning the Tide: The Eradication of Invasive Species* (eds C.R. Veitch & M.N. Clout), pp. 64–70. IUCN Species Specialist Group, Gland, Switzerland, and Cambridge, UK.

Burbidge, A., McKenzie, N.L., Brennan, K.E.C., Woinarski, J.C.Z., Dickman, C.R., Baynes, A. *et al.* (2008) Conservation status and biogeography of Australia's terrestrial mammals. *Australian Journal of Zoology*, **56**, 411–422.

Campbell, K.J., Harper, G., Algar, D., Hanson, C.C., Keitt, B.S. & Robinson, S. (2011) Review of feral cat eradications on islands. *Island Invasives: Eradication and Management* (eds C.R. Veitch, M.N. Clout & D.R. Towns), pp. 37–46. IUCN, Gland, Switzerland.

Carlquist, S. (1974) *Island Biology*. Columbia University Press, New York & London. pp. 660. 581, 5279.

Choquenot, D., Hone, J. & Saunders, G. (1999) Using aspects of predator-prey theory to evaluate helicopter shooting for feral pig control. *Wildlife Research*, **26**, 251–261.

Connolly, T.A., Day, T.D. & King, C.M. (2009) Estimating the potential for reinvasion by mammalian pests through pest-exclusion fencing. *Wildlife Research*, **36**, 410–421.

Courchamp, F., Chapuis, J.-L. & Pascal, M. (2003) Mammal invaders on islands: impact, control and control impact. *Biological Reviews*, **78**, 347–383.

Courchamp, F., Caut, S., Bonnaud, E., Bourgeois, K., Angulo, E. & Watari, Y. (2011) Eradication of alien invasive species: surprise effects and conservation successes. *Island Invasives: Eradication and Management* (eds C.R. Veitch, M.N. Clout & D.R. Towns), pp. 285–289. IUCN, Gland, Switzerland.

Donlan, C.J. & Wilcox, C. (2007) The complexities of costing eradications. *Animal Conservation*, **10**, 157–158.

Ferraro, P.J. & Pattanayak, S.K. (2006) Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *Public Library of Science, Biology*, **4**, 482–488.

Island Conservation (2012) Database of Island Invasive Species Eradications. http://eradicationsdb.fos.auckland.ac.nz/ (accessed 5 April 2013).

Johnston, M., Algar, D., Onus, M., Hamilton, N., Hilmer, S., Withnell, B. *et al.* (2010) A bait efficacy trial for the management of feral cats on dirk Hartog Island. *Technical Report Series*. Arthur Rylah Institute for Environmental Research Heidelberg, Australia.

Keitt, B.S., Campbell, K.J., Saunders, A., Clout, M., Wang, Y., Heinz, R. *et al.* (2011) The global islands invasive vertebrate eradication database: a tool to improve and facilitate restoration of island ecosystems. *Island Invasives: Eradication and Management* (eds C.R. Veitch, M.N. Clout & D.R. Towns), pp. 74–77. IUCN, Gland, Switzerland.

Kier, G., Kreft, H., Lee, T.M., Jetz, W., Ibisch, P.L., Nowicki, C. *et al.* (2009) A global assessment of endemism and species richness across island and mainland regions. *Proceedings of the National Academy of Sciences of the United States of America*, **106**, 9322–9327.

Long, K. & Robley, A. (2004) *Cost Effective Feral Animal Exclusion Fencing for Areas of High Conservation Value in Australia*. Department of Environment and Heritage, Canberra, ACT.

Martins, T., Brooke, M., Hilton, G., Farmsworth, S., Gould, J. & Pain, D.J. (2006) Costing eradications of alien mammals from islands. *Animal Conservation*, **9**, 439–444.

McBride, M.F., Wilson, K.A., Bode, M. & Possingham, H.P. (2007) Incorporating the effects of socioeconomic uncertainty into priority setting for conservation investment. *Conservation Biology*, **21**, 1463–1474.

McKenzie, N.L., Burbidge, A.A., Baynes, A., Brereton, R.N., Dickman, C.R., Gordon, G. *et al.* (2007) Analysis of factors implicated in the recent decline of Australia's mammal fauna. *Journal of Biogeography*, **34**, 597–611.

Micol, T. & Jouventin, P. (2002) Eradication of rats and rabbits from Saint-Paul Island, French Southern territories. *Turning the Tide: The Eradication of Invasive Species* (eds C.R. Veitch & M.N. Clout), pp. 199–205. IUCN Species Specialist Group, Gland, Switzerland.

Mittermeier, R.A., Gil, P.R., Hoffman, M., Pilgrim, J., Brooks, T., Mittermeier, C.G. *et al.* (2005) *Hotspots Revisited*. University of Chicago Press, Chicago, IL.

Morrison, S.A. (2007) Reducing risk and enhancing efficiency in non-native vertebrate removal efforts on islands: a 25 year multi-taxa retrospective from Santa Cruz island, California. *Managing Vertebrate Invasive Species* (eds G.W. Witmer, W.C. Pitt & K.A. Fagerstone), pp. 398–409. University of Nebraska, Lincoln.

Myers, J.H., Savoie, A. & van Randen, E. (1998) Eradication and pest management. *Annual review of entomology*, **43**, 471–491.

Parkes, J.P. (1990) Eradication of feral goats on islands and habitat islands. *Journal of the Royal Society of New Zealand*, **20**, 297–304.

Parkes, J. & Panetta, F. (2009) Eradication of invasive species: progress and emerging issues in the 21st century. *Invasive Species Management. A Handbook of Principles and Techniques* (eds M.N. Clout & P.A. Williams), pp. 47–60. Oxford University Press, Oxford.

Pluess, T., Cannon, R., Jarošík, V., Pergl, J., Pyšek, P. & Bacher, S. (2012a) When are eradication campaigns successful? A test of common assumptions. *Biological Invasions*, **14**, 1365–1378.

Pluess, T., Jarošík, V., Pyšek, P., Cannon, R., Pergl, J., Breukers, A. *et al.* (2012b) Which factors affect the success or failure of eradication campaigns against alien species? *PLoS ONE*, **7**, e48157. doi:10.1371/journal.pone.0048157

Rudge, M.R. (1986) The decline and increase of feral sheep (*Ovis aries* L.) on Campbell Island. *New Zealand Journal of Ecology*, **9**, 89–100.

Sax, D.F. & Gaines, S.D. (2008) Species invasions and extinction: the future of native biodiversity on islands. *Proceedings of the National Academy of Sciences of the United States of America*, **105**, 11490–11497.

Schuyler, P.T., Garcelon, D.K. & Escover, S. (2002) Eradication of feral pigs (*Sus scrofa*) on Santa Catalina Island, California, USA. *Turning the Tide: The Eradication of Invasive Species* (eds C.R. Veitch & M.N. Clout), pp. 274–286. IUCN Species Specialist Group, Gland, Switzerland.

Short, J. & Turner, B. (2005) Control of feral cats for nature conservation. IV. Population dynamics and morphological attributes of feral cats at Shark Bay. *Western Australia. Wildlife Research*, **32**, 489–501.

Towns, D.R. & Broome, K.G. (2003) From small Maria to massive Campbell: forty years of rat eradications from New Zealand islands. *New Zealand Journal of Zoology*, **30**, 377–398.

Usher, M.B. (1989) Ecological effects of controlling invasive terrestrial vertebrates. *Biological Invasions: A Global Perspective* (eds J.A. Drake & H.A. Mooney), pp. 463–489. John Wiley & Sons, New York.

Walters, C.J. (1986) *Adaptive Management of Renewable Resources*. MacMillan, New York.

Weesakul, B. (1961) The random walk between a reflecting and an absorbing barrier. *The Annals of Mathematical Statistics*, **32**, 765–769.

Received 15 February 2013; accepted 23 May 2013
Handling Editor: Robert B. O'Hara