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CORRESPONDENCE

Complexities of costing eradications

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Conservation dollars are and likely always will be limiting. Spending those dollars effectively and efficiently is of pivotal importance (Wilson *et al.*, 2006). Martins *et al.* (2006) present an analysis of economic costs of invasive mammal eradications on islands that they claim 'provide conservation planners with a robust, if preliminary, estimate of the cost of any proposed eradication programme.' Given that invasive mammal eradication is one society's most powerful conservation tools, this is a badly needed instrument (Donlan *et al.*, 2003). However, while we commend Martins and colleagues' efforts and goal, the available data are simply and sadly insufficient to provide such a tool. More importantly, providing a naïve tool is potentially misleading and dangerous for land managers, conservation practitioners and philanthropists who may turn to the pages of conservation journals for guidance.

The costing of eradication campaigns is a complex process. As quantitatively demonstrated by Martins and colleagues, island area plays a major factor in the variation of costs. Using general linear models with a dataset that included 41 eradication campaigns, the authors conclude 'decision makers considering potential eradication programmes need only know island area, distance from airport (at least in the New Zealand region), and species to be eradicated to make internally consistent and robust first-pass estimates of likely costs.' In reality, a number of factors and examples strongly suggest that this is not the case. We briefly address five of those issues here.

First is the issue of fixed costs. Martins and colleagues claim that due to efficiency gains in the ability to remove invasive mammals from islands, campaigns should be significantly cheaper today than 20 years ago. They go on to state 'thus, for a hypothetical 10 km^2 island situated 100 km from an airport, costs would decrease from US$ 251 000 if rodents were eradicated in 1983 ... to only US$ 31 200 if it were done in 2003.' While the authors are correct in suspecting that the detected effect in their regression model is indeed real, due to certain fixed costs of eradication campaigns, the relationship does not decrease linearly with gains in efficiency. For example, a conservative estimate of the cost of rodenticide bait alone for an aerial eradication campaign on a 10 km^2 island is US$60 000 – twice their predicted total eradication cost (assuming $4 per kg of bait with an application rate of 15 kg ha^{-1}, cost estimate from Bell Labs, USA). While a few of the rodent eradication campaigns used by Martins *et al.* reported total costs less than the bait cost estimate at the above application rate, these were conducted on small islands ($\leq 1\,km^2$) where either a bait-station approach was used, or perhaps a lower application rate was used but still successful due to the small area of the island. Of the data available worldwide for application rates of past aerial broadcasts, now the most common method used in rodent eradications, the mean application rate is 17.6 kg ha^{-1}, which is a conservative estimate as more than one bait application is sometimes necessary (range 10–35, $n = 16$; C. J. Donlan, unpubl. data).

Secondly, variable costs have a substantial impact on the realized cost of an eradication campaign. While Martins *et al.* acknowledge that local factors may be important, they fail to appreciate the impact of these factors. Important variable costs of eradication campaigns include (1) needs to mitigate for potential non-target species, (2) the techniques used (e.g. bait stations versus aerial broadcast of bait for rodent eradications), (3) the level of local capacity present, (4) amount of environmental compliance required and (5) the levels of bureaucracy. For example, the cost of campaigns for rodent eradication using aerial broadcast by helicopter depends largely on isolation (e.g. nearest port not airport as remote campaigns are increasingly boat based), while the cost of a campaign using a bait station approach depends largely on labor costs. These variable costs can influence the overall costs of eradication campaigns by an order of magnitude. Black rats *Rattus rattus* were recently eradicated from Anacapa Island, CA, USA (Howald *et al.*, 2005). Anacapa Island was the first aerial-based rodent eradication campaign in the United States, and thus was subject to substantial environmental compliance requirements as dictated by a suite of federal laws. Further, the eradication campaign required mitigation for a endemic rodent present on the island due to potential non-target effects of the rodenticide (Howald *et al.*, 2005). Thus, the eradication campaign on Anacapa Island, a small 3 km^2 island just 80 miles from Los Angeles International Airport, cost ~US$2 million (G. Howald and C. J. Donlan, unpubl. data). While environmental compliance requirements differ widely among countries, navigating bureaucracy (e.g. permitting) is a pervasive challenge and can require substantial resources.

Thirdly, increases in efficiency with time (as noted by the authors) are likely a function of the reduction in variable costs with experience rather than time *per se*. These reductions are a product of experience in a given system (i.e. with a particular target species, using certain eradication methods in an ecological system, and navigating local bureaucracy and environmental compliance requirements within a socio-political unit). In contrast to New Zealand, where massive resources and effort have been dedicated to island conservation efforts over the past three decades, invasive mammal eradications in new systems will incur relatively high variable costs that will be difficult to estimate, requiring expert knowledge and past experience in other systems (for an example, see U.S. National Park Service, 2000). This is an important point that should be communicated to land managers and conservation funders.

Fourthly, while the data used by Martin *et al.* may be statistically adequate for exploring rat eradications ($n = 29$), their data are inadequate for other invasive mammals ($n = 3$ cats, $n = 4$ ungulates, two of which are atypical eradications; see Parkes, 1990; Campbell & Donlan, 2005). The scant dataset leads them to a likely incorrect conclusion that rodent eradications are 1.7–3 times more expensive than ungulate eradications. The small amount of ungulate data, and its narrow distribution with respect to island area (in log space), causes their statistical model to produce a spurious relationship for taxon. This is driven by the assumption of a common slope across species; if relationships are fit separately, ungulates are cheaper to eradicate on small islands, but much more expensive on large islands (Fig. 1). In reality, this may or may not be true – more data are needed. However, two important differences between rat and ungulate eradications suggest that their area–cost slopes are different. First, on both small and large islands, rodents are commonly eradicated using a one-time poisoning effort that puts all animals at risk. In contrast, ungulate eradication campaigns on large islands call for multiple, specialized methods that are used over a longer period of time (e.g. aerial hunting, ground hunting with specialized hunting dogs and Judas goats), and consequently removing the last animals is increasingly expensive (Parkes, 1984). Second, because of these differences, rodent eradications are likely more influenced by fixed costs (e.g. bait costs and helicopter time), while with ungulates, eradications can be made more cost-effective by leveraging new technology and techniques (Campbell & Donlan, 2005; Lavoie *et al.*, 2007). For

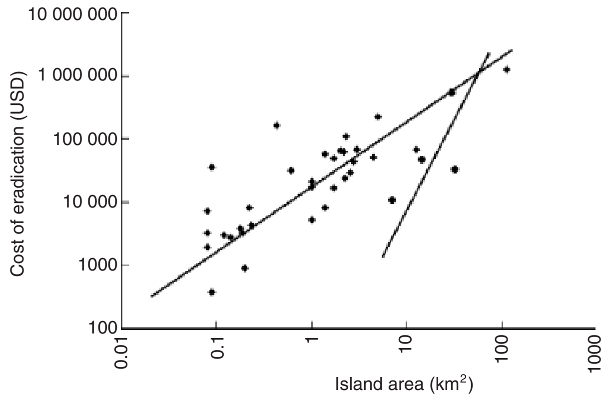

Figure 1 Area and cost of invasive rodent (♦) and ungulate (•) eradication campaigns redrawn from Martins *et al.* (2006) not assuming a common slope between taxa. Lines for each species are based on a linear regression model in log–log space, assuming only an intercept and an area effect.

example, goats were recently removed from Santiago Island, Galapagos (58 465 ha), at a cost of ~$US 5.1 million. Following the success, experience and refinement of techniques on Santiago Island, goats were removed from Isabela Island, Galapagos (458 812 ha), at a cost of ~$US 3.5 million (F. Cruz, V. Carrion, K. Campbell & C. J. Donlan, unpubl. data).

A final point is one concerning drawing real-world conclusions from power functions (i.e. analyses of log-transformed data). Linear relationships in log–log space are notoriously hard to interpret (Smith, 1984), which is exacerbated by the strong human bias toward linear perceptions (Ross, 1974; Cleveland, Diaconis & McGill, 1982). For example, it is difficult to appreciate the difference between the log values of 0.4 and 2.0; yet, they represent the difference between the small island of Anacapa and the largest successful rodent eradication to date (Campbell Island, NZ; 3 vs. 113 km^2). Yet, due to variable costs driven not by area or remoteness, eradication on Anacapa, a small island near of one of the busiest airports in the world, costs more than Campbell Island – a massive one in the remote subantarctic ($2 vs. $1.2 million).

The foremost point Martins and colleagues bring to light is the lack of economic data for invasive mammal eradication campaigns (Campbell & Donlan, 2005). The authors do conservation practitioners a great service by calling upon them to report on the economic data of conservation interventions. Until such data are available, prioritization of invasive mammal eradications should proceed without input costs, and rather should rely on biodiversity value, risk, urgency and other important factors. Economic costs will drastically improve prioritization efforts, and we fervently agree with Martins and colleagues that if we do not take on such bio-economic challenges, our conservation efforts will continue to fall short (Naidoo *et al.*, 2006). But we must do so carefully and close to the realities of on-the-ground conservation action.

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