Threatened Species Recovery Hub

National Environmental Science Programme

Improved budgetary planning for threatened species and ecological community recovery plans

Stephen T. Garnett, Linda Luck, Kerstin K. Zander, Ram Pandit, Asha Gunawardena, David Pannell

31 July 2019

Charles Darwin University Australia

The University of Western Australia

Improved budgetary planning for threatened species and ecological community recovery plans

31 July 2019

Authors:

Stephen T. Garnett [1]

Linda Luck[1]

Kerstin K. Zander[1]

Ram Pandit [2]

Asha Gunawardena[2]

David Pannell[2]

[1] Northern Institute and Research Institute for the Environment and Livelihoods, Charles Darwin University, Northern Territory 0909, Australia

[2] Centre for Environmental Economics & Policy, UWA School of Agriculture and Environment, M087, University of Western Australia, Perth WA 6009, Australia

Garnett, S. T., Luck, L., Zander, K. K., Pandit, R., Gunawardena, A. and Pannell, D. 2019. Improved budgetary planning for threatened species and ecological community recovery plans. Report to the National Environment Science Program, Department of the Environment and Energy, Canberra. 31 July 2019.

Cover image: A feral predator exclusion fence constructed by the Australian Wildlife Conservancy at Mt Gibson Sanctuary, WA. Photo: Jaana Dielenberg

Table of contents

Abstract

Every recovery plan for a threatened species or ecological community requires a budget that estimates the cost of the actions specified in the plan. When first initiated in the 1990s, these budgets were routinely used as the basis for funding subsequent actions. Gradually, however, they have been simplified to become indicative of costs with no commitment to funding. This has reduced their utility and quality as there is no motivation for either the authors or reviewers to check the accuracy of the estimates, which in turn means they are of little use for budgetary planning. There is also a trend for the budget estimates to be specified only at the highest level with no transparency about how they have been compiled. In this report we have (i) recalculated budget estimates for four recovery plans – Boggomoss Snail, Swift Parrot, Macquarie Perch and Central Rock-Rat – to explore how these budgets could contribute to understanding both the costs and the benefits of supporting the recovery; (ii) developed a framework for calculating the costs of predator exclusion fencing; (iii) obtained initial costs of ex situ conservation; (iv) calculated the improvements in threat alleviation that can be obtained from investment in different actions. For (i) we have broken down each action in each recovery plan into their component parts and the year, over a five year period, in which the action will occur so that we can include provision for inflation and changes in employment costs. As all actions need to be undertaken by people, we determined the employment category and institutional affiliation of the personnel likely to be undertaking those actions, the time that will be required to carry out each action and the employment status (e.g. full-time, casual, contract). We have also calculated the travel costs, operational costs, the costs of equipment and, where relevant, the costs of (ii) and (iii) above. We have separated the costs between those that need to be covered by institutions, including private businesses, and those that are provided voluntarily as in-kind support by private citizens. For institutions we have estimated the additional costs associated with employment. We have also matched actions against threats and assessed where the investment in threatened species conservation is most likely to occur.

Estimating costs in such detail allows aggregation in ways that can contribute not just to the recovery plan but to also forward planning for threatened species recovery as a whole. Budgetary summaries are provided for the whole plan, for each strategic aim, each individual action and for each institutional partner. Similarly, employment is summarised for the plan and for each contributing institution. Expenditure can also be estimated for each priority category attributed to an action and cost estimates can also be estimated for different levels of uncertainty about action completion. Classification of actions into the categories developed by the IUCN and for Conservation Action Planning also enables the relative costs of different action types to be compared and aggregated across plans. By attributing each threat to an action and estimating the extent to which a threat will be alleviated by delivering on that action, makes it possible to calculate monetary return on investment in terms of threat alleviation. Finally, based on estimates gleaned from the literature, indirect and induced benefits are then estimated in terms of expenditure per dollar of direct investment and employment per year for each unit of investment. Potential additional benefits to the budget planning are incorporation of non-market values, partnership benefits and use benefits (tourism etc.).

1. Introduction

Accurate and detailed costing are fundamental to conservation planning (Cook et al. 2017). The arguments, outlined in detail by Iacona et al. (2018), are that they allow estimates of cost-effectiveness of different actions within and between organisations, enable planning of expenditure and are fundamental to efficient resource allocation such as those espoused by Gerber et al. (2018). Arguments on the efficient allocation of funds can then be used to argue that additional funds are required for conservation to be effective, as happened in New South Wales (Office of Environment and Heritage 2016). Yet accurate costings are rarely documented sufficiently well to allow predictive modelling. A review of 30 peer-reviewed articles on conservation costings by Iacona et al. (2018) found that essential information was missing from most and proposed a framework for budgetary reporting that would contribute to a global database.

Recovery planning for threatened species, which was initiated in Australia in the 1990s (Stephens and Maxwell 1996), aims to identify, prioritise and plan actions that will prevent extinctions and promote species recovery. When first introduced in the 1990s, plans were routinely funded to their full extent. Gradually, however, this practice was eroded and budgets were requested with increasingly little detail and explanation of how they had been derived. This developed into a negative feedback loop with a lack of belief in funding being provided reducing the effort put into estimating the budget required, making it less likely for the recovery plan budget estimates to be used for budgetary planning.

The current project aimed to initiate restitution of accuracy in cost estimates, identifying advantages in budget planning beyond those identified by Iacona et al. (2018) that can help policy-makers understand more fully the benefits of investment in threatened species conservation. These include the contributions that threatened species investment can make to remote and regional communities where investment is often lacking as well as to understanding the mix of threatened species conservation expenditure classes that are needed for different stages in the recovery process and the costs of different types of threat amelioration.

The former contribution of threatened species investment explores the benefits derived in terms of the indirect and induced expenditure within the communities where the expenditure occurs. Given the flight of skills and resources from remote and regional Australia, direct expenditure for threatened species can potentially deliver the same types of benefit as resource development at a community level (Fleming and Measham 2015) or tourism (Gibson and Connell 2016; Rolfe 2019). The high level of volunteer involvement with threatened species monitoring and management can bring additional benefits (Brightsmith et al. 2008).

Accurate assessments of budgetary costs can also be combined with a new metric of assessing progress in threat amelioration (Garnett et al. 2019) to evaluate the effectiveness of different actions in reducing the threats to threatened species. The metrics developed combine IUCN measures of threat intensity with those of the Royal Society for the Protection of Birds on progress in conservation to calculate rates of amelioration given the implementation of specific actions.

As noted by Iacona et al. (2018), information on actual expenditure is sparse and poorly reported. The approach adopted here, therefore, has been to reconstruct the budgets of recovery plans. This has two benefits. First, it deals with all the actions specified by the authors of the plans as being required not just those that have been performed which may be subject to budgetary, social or political constraints. Secondly, it permits comparison between the published estimates of costs of actions and those estimated using a comprehensive costing framework.

2. Methods

The cost analysis presented in this report has four components and the method adopted for the analysis is described below.

i. Recovery plans: Four recovery plans were selected at random for budget analysis. Actions and their associated budgets were then disaggregated into their component parts so that costs could be allocated to different budgetary categories, action categories and organisations
ii. Exclusion fencing: Fencing costs were identified as cost type that warrants separate estimates as a unit of cost in some recovery plans, particularly for mammals
iii. Ex situ conservation: Ex situ costs were identified as cost type that warrants separate estimates as a unit of cost in some recovery plans, particularly for mammals
iv. Cost data aggregation: All costs were then aggregated into categories for subsequent reassembly so they could be used for other policy-related purposes.

2.1 Recovery plans

Four recovery plans were chosen from those published on the EPBC web site or provided by the Department of the Environment and Energy with the aim of choosing examples that would provide a breadth of taxonomic type and management complexity as recommended for case study analysis (Yin 2017). All selected recovery plans were for animals. An initial review of recent plans for plants or communities suggested that too much additional work was required to link suggested actions to plan objectives and there were no recovery teams extant with whom these costs could be confirmed. The plans selected included those for the Boggomoss Snail (*Adclarkia dawsonensis*) (Stanisic 2008), Macquarie Perch (*Macquaria australasica*) (Commonwealth of Australia 2018), Central Rock-Rat (*Zyzomys pedunculatus*) (McDonald et al. 2017) and the Swift Parrot (*Lathamus discolor*) (Commonwealth of Australia 2019).

The actions in each plan were then extracted and listed against the plan objectives. Actions as described in the recovery plans that involved multiple activity types were broken down into their component parts – e.g. "plan and implement pest control" was separated into a planning sub-action and an implementation sub-action as these would involve different people in different places. For a few plans extra actions were suggested because they had objectives listed without actions aimed at meeting those objectives. However, where threats were listed in recovery plans but described no actions that would deal with those threats, the threats without specific actions were simply identified in the threat analysis.

Each action was then costed in as much detail as possible, using common budgeting categories as described by Iacona et al. (2018). All actions had an allocation of staff time, whether full-time, casual or contract. The level of employment, the time required to undertake an action and the organisation responsible for funding that employee were recorded with salaries and on-costs calculated from the mean of multiple government and academic organisations. Travel for any action was calculated for vehicle, air or other travel types as well as accommodation and other travel allowances. Vehicle costs were all calculated as though they were hired with the assumptions that hire rates cover maintenance and depreciation costs of vehicles owned outright by organisations. Equipment costs, room hire costs and other potential expenses required to carry out an action were also estimated or calculated from market information or provided empirically by conservation practitioners involved in recovery plan implementation. As volunteers are routinely involved with threatened species conservation work, the hours they donate and the travelling they undertake to participate in activities were also estimated. The value of their work was determined by the nature of the activities they would be expected to undertake so was pegged against equivalent professional rates.

While not a feature of the four case studies, modules for exclusion fencing and captive breeding were developed separately for use with other species requiring these conservation interventions. Work is underway to incorporate the costing estimates for feral animal eradication from islands recently published by Wenger et al. (2018).

Institutional infrastructure costs were estimated as a percentage of the total budget, as is common practice among organisations estimating the costs of projects, with 100% of project costs being the default following standard government practice (Allen Consulting Group 2008).

2.2 Exclusion Fencing

Fencing costs were extracted from all known published sources. Each was interrogated for information on the costs of labour and material for construction and maintenance. Initial eradication costs were not included because they have recently been comprehensively modelled for similar systems by Wenger et al. (2018). Similar model parameterisation for fencing will be undertaken when partner investigators provide the relevant data. Costs of depreciation (Scofield et al. 2011) were absorbed into replacement costs every 25 years (Norbury et al. 2014).

Net Present Value of the fencing costs (NPV) was calculated using the discounting formula adopted by the Department of Prime Minister and Cabinet (2016) as follows:

$NPV = C_t/(1+r)^t$

where C_t = the cost at time t

r = the discount rate

t = the number of years over which the future costs are expected to occur

2.3 Ex situ conservation

For ex situ conservation, detailed costing data on 40 collections of 31 threatened species were obtained from three large public institutions involved in the captive breeding of threatened species. This included details of the time spent by all levels of personnel in the institution on each taxon and their salary levels as well as information on the costs of feeding and other consumables, operating costs, maintenance and, where possible, the cost of the facilities in which they were housed. This latter was more difficult because many of the collections were in multi-use or old facilities and it was hard to calculate current replacement costs for a dedicated facility. Data is also being gathered on the costs or maintaining collections of threatened fish and plants with data sharing agreements currently being arranged with the 12 organisations that collect seed in Australia for conservation purposes. This data will be incorporated into the final analysis of the cost of ex situ conservation in Australia. Missing data from that collected so far includes the costs of obtaining breeding stock from the wild and the costs of institutional support.

2.4 Cost Data Aggregation

Costs were aggregated in five ways.

i. Objectives and priorities: This approach is based on the costs required to meet each objective, allowing creation of a total plan budget including all costs, including the costs of volunteered time and of the institutional infrastructure required to support on-ground action. This is the standard for recovery plans so these costs were compared with the budgets provided by the recovery plan itself, excluding the infrastructure costs which are known to be omitted from recovery plans. Actions in recovery plans are also routinely given a high, medium or low priority and budgets can also readily be aggregated across different priority categories.

ii. Action class: The IUCN developed a categorisation of threats to species in association with the Conservation Measures Partnership (CMP) (IUCN & CMP 2012). Subsequently the CMP developed a second generation classification (CMP 2016). Both are used by conservation practitioners to classify threats, a classification that can then be used to influence investment decisions. However, any process of threat amelioration must first understand the relative costs of different types of action. As far as we are aware, this has never been estimated across multiple threatened species so there is no model to determine, at a broad level, how much may need to be allocated to, for example, land acquisition, fire management or population research.

iii. Threats and progress: Each action was attributed in whole or in part to one of the threats listed in the plan. A category was also created for actions that could not be attributed to the amelioration of a particular threat. The severity of all threats was then assessed using the IUCN criteria (IUCN) on their currency (whether they were currently operating or had operated but might return in the near or distant future), the extent of the species range to which they pertained and the rate of loss they were causing within their area of influence. Following Garnett et al. (2019), an assessment was then made on the current extent of knowledge of how to manage each threat and the extent to which management was currently being implemented. Based on the performance criteria listed in the recovery plan and the current threat impact, an assessment was then made on the threat intensity, the knowledge of how to manage a threat and the extent to which management would be applied to the threat at the end of the recovery period. The recovery period was taken as five years for each of the case studies. From this it was possible to calculate the financial resources potentially required for each unit of improvement in knowledge or management or each unit of threat reduction. While not as sophisticated as modelled changes in percentage threat reduction using population viability analysis (Kissel et al. 2017), it provides a common currency for comparing the costs of threat reduction across species that, with more examples, will then allow modelling for entire threat classes.

iv. Employment: The number of people, in total and across different employment categories, could be estimated for the entire recovery plan for each organisation involved and for the plan in aggregate. Such data are important for workforce planning and understanding the role of threatened species recovery as part of the local, regional and national employment profile.

v. Social impacts: Type I and II multipliers were derived from published estimates of indirect and induced expenditure and job creation (Nielsen-Pincus et al. 2010, Allen Consulting Group 2011, Stoeckl et al. 2011, Edwards et al. 2013, Nichol et al. 2013, Laughland et al. 2014, Kelmenson et al. 2016, Clarke et al. 2017, Royuela et al. 2017, Australian Bureau of Statistics 2018). Each action was allocated to a theatre of probable expenditure based on the regionalisation of Australia (Australian Bureau of Statistics 2018). While it was expected that most planning, for example, would be undertaken in major capitals, much of the field activity would be expected to result in expenditure in regional and remote parts of Australia where investment can have major impacts with benefits to other policies related to maintaining economic activity in these areas.

3. Results and key findings

3.1 Cost categories

Analysis of the re-estimated Recovery Plan costs were much higher than the original budget estimates, primarily because they included infrastructure costs, although it was evident that some actions had not been fully costed in the original budget (Table 1). There was some level of consistency across categories but enough variability that there is potential for modelling with a wider range of examples. Such modelling would make it possible to estimate likely costs of recovery for different species types in different types of physical and human geography. This in turn would allow budgetary planning at multispecies scales, especially if complementary actions could also be identified.

Table 1. Costs of Recovery Plan case studies and the percentage allocation to budgetary categories

Description	Boggomoss Snail	Swift Parrot	Macquarie Perch	Central Rock-Rat	Overall total
Original budget ($'000)	$146	Not available	$3,010	$1,020	
Reconstituted budget* ($'000)	$741 ($812)	$10,227 ($11,555)	$7,186 ($8,150)	$1,373 ($1,460)	
Budget breakdown (% of total)					
Salary	37.5	23.3	35.1	22.9	28.2
Consultants	2.8	0.0	0.0	0.8	0.2
Travel / Transport	5.3	9.8	0.6	0.4	5.6
Consumables and Facilities	1.5	0.1	1.9	18.3	2.0
Equipment	4.2	0.8	0.1	5.7	1.0
Contingency	1.3	0.9	1.0	1.2	0.9
GST (where applicable)	0.6	0.1	0.2	2.4	0.3
Infrastructure	46.9	28.5	43.5	48.4	36.1
In-kind	0.0	36.5	17.6	0.0	25.7

** The figures for reconstructed budgets are based on 7% and 3% (in parenthesis) discount rates.*

Infrastructure costs were the highest cost items proportionately. This was never included in any of the recovery plan budgets examined but is a largely hidden contribution from organisations hosting recovery programs. It may have been particularly high in the examples examined because it was assumed that the government agencies responsible for most of the work would charge at a rate of 100% of project costs. It was assumed that some non-government organisations and private providers, such as farmers, may provide infrastructure support at a lower rate. Governments, academic institutions and other large organisations, however, do need to maintain support staff, buildings and other physical infrastructure, insurance and many other costs (i.e., overhead costs), a proportion of which is used to support those undertaking the field research, money that is usually allocated from public expenditure.

Of the cash contributions (employment, operating, capital costs), employment was always the largest element of any recovery plan cost, though this too is never transparent in published recovery plan budgets. Across the four case studies, it constituted 74% of all costs (47% for Central Rock-Rat to 91% for Macquarie Perch) of all non-infrastructure costs. Employment costs were calculated only for those directly involved in undertaking or managing work directly related to recovery with the time of more senior managers being absorbed into infrastructure costs.

In-kind labour varied greatly between projects. Swift parrot surveys are undertaken in their non-breeding range on a large scale involving hundreds of hours of volunteer time and travel. Macquarie perch research and management are often assisted by the angler community. No volunteer time was identified for the boggomoss snail or the central rock-rat. Some of the in-kind support is hidden. Employment costs were always calculated at standard rates when, in fact, much threatened species management happens at unconventional hours without overtime but no data is available on this contribution.

Travel was often expensive, with allowances making up a substantial proportion of field costs. Often such allowances, mandated in enterprise agreements, are foregone by dedicated professionals in the interest of stretching programme funds. In such cases they should be allocated to in-kind support but, as with time volunteered by employees working on threatened species, there is no data to support the extent to which this supposition is true.

Operational expenditure and equipment costs were usually bespoke. For example, central rock-rat are monitored with imported camera traps, Macquarie perch require weirs, and swift parrots require specialised nest boxes. Such expenditure is harder to model but is usually a relatively small component in the overall budget. Although helicopter hire increased the proportion of costs allocated to consumables in the rock-rat recovery plan budget. Some of these items may also be used across multiple recovery plans, with items like camera traps becoming standard for monitoring of mammals.

Many elements of recovery plans could be covered by costing protocols, such as those developed for Western Australian islands by Wenger et al. (2018) for the eradication of feral animals and weeds. Here we have developed preliminary costings for exclusion fencing and captive breeding.

3.2 Exclusion fencing

Given the enthusiasm for landscape-scale predator-proof fencing as a means of conserving Australian native mammals in particular (Legge et al. 2018), there is limited data on its costs. Such fences are complex and so more expensive than fences constructed to keep out wild dogs, kangaroos or emus (URS 2007). For Australia best data is for material costs, though most of the estimates are now getting quite old: Arid Recovery (Moseby and Reid 2006) had details for one area with greater detail in Long and Robley (2004a). More recently costs were provided for predator-proof exclosures by Hayward et al. (2014), a recent model for a toad proof fence in Western Australia (Southwell et al. 2017) and a cat proof fence in central Australia (Ireland et al. 2018). Australian studies referring to cost-effectiveness (e.g. Bode and Wintle 2010, Helmstedt and Possingham 2017) either do not actually refer to costs but just the items likely to influence them or quote the few existing sources.

Information from overseas is of limited value for Australia because of differences in cost structures and the types of animal the fences are designed to contain, either outside (New Zealand) or inside (Africa). For New Zealand, where fences have to keep out mustellids, rats and sometime mice, Campbell-Hunt (2008) estimated the construction cost of predator proof fencing at NZ$200,000/km in 2008, translated to US$150,000/km in 2011 (Burns et al. 2012) and then modelled by Norbury et al. (2014) to produce a cost of NZ$250,000/km. Scofield et al. (2011) in New Zealand provided costs for 10 existing and three mainland 'island' fences and five Peninsula fences and estimated a cost of NZ$220,000/km with maintenance at NZ$880,000. In current terms, the costs of construction would be AU$229,000-238,000/km based on appropriate inflation and exchange rate multipliers (Reserve Bank of New Zealand 2019, Reserve Bank of Australia 2019a).

In Africa, where wages are lower, construction costs were US$20,000/km in 2008 in Kenya for a fence designed to retain elephants and lions (Lamarque et al. 2008) and, in South Africa, US$4,500-6,500/km for antelope fences and $5,250-$7,250/km for carnivore fences (Lindsey et al. 2012). These translate to current estimates of AU$7,700/km (antelopes), AU$11,700/km (lions) and AU$34,400/km (elephants) (Reserve Bank of Australia 2019a, Coinnews Media Group Llc 2019).

Lacking antelopes, lions, elephants or mustellids and with no recognised threat from mice, we have summarised only the limited data from Australia. We had intended to obtain additional data but the schedules of partner practitioners with original empirical data on their fencing costs have so far prevented them from providing it – though there is every intention that it will be made available for the final publication. This report therefore identifies the types of costs involved in fence construction and maintenance, summarises existing published information, and updates it based on costs of inflation then calculates an annualised Net Present Value.

Overall costs of fence materials extracted from the literature are shown in Table 2. All costs of Moseby and Reid (2005) exclude costs of fenceline clearing, freight, foot aprons in erodable areas and labour and the potentially substantial costs of steel posts were 'minimised by using recycled bore casing'. It is unclear what was included in the costs of Hayward et al. (2014) for Scotia but they are assumed to be the same as for Moseby and Reid (2006). All costs have been updated using the inflation rate adopted by the Reserve Bank of Australia (Reserve Bank of Australia 2019b) with no adjustment for variation among the different elements (e.g. a fence costing $10,000 in 2004 is estimated to cost $14,022 in 2018).

Labour costs were supplied explicitly only by Southwell et al. (2017) who calculated that, in 2017, they were about the same as those for materials based on contracting commercial rates.

Maintenance costs related to area were provided by Hayward et al. (2014) who estimated $25,000 per year ($26,700 in 2018) for the 60 km^2 of Arid Recovery, including periodic eradications, $13,000 per year ($13,885 in 2018) for the 650 km^2 of Scotia but it is not clear what the maintenance involved. The cost of maintaining the toad fencing was estimated at $20,800 per year (Southwell et al. 2017) but it was unclear how many kilometres this covered. Norbury et al. (2014) estimated the cost of maintenance at 4% of capital costs, including 'fence inspections and eradication costs following occasional pest incursions', for New Zealand fences with Scofield et al. (2011) estimating a very similar maintenance costs (3.67% of capital costs). For Australian fences (Long and Robley 2004b) estimated that fence maintenance alone took 0.9 hrs/km/week. Assuming this is undertaken by one person at the 2019 basic wage of $18.93/hour (Australian Government 2019) plus a vehicle cost of $0.91/km (average for four wheel drive utes in (RACQ 2019), this amounts to an annual cost/km of $1,820. For Arid Recovery this would amount to a cost of about $93,000 per year for the approximately 100 km of fencing shown in the Reserve Layout (Arid Recovery 2019) ($760/km^2) and $75,000 per year for the 80 km (De Torres and Marlow 2011) perimeter fence ($933/km^2).

Additional costs for such fences are likely to include scrub clearance every four years along the fenceline ($200/hr x 4 km/hr x 4 blade widths) and solar power unit replacement every 10 years ($3000).

The Net Present Value of fence costs/km at a discount rate of 7.0% (Department of Prime Minister and Cabinet 2016) gives values in 20, 50 and 100 years (Table 2).

Table 2. Fencing costs and Net Present Value/km of a selection of costed fences designed to exclude feral animals in Australia including material, labour and maintenance costs.

Target animal	Fence type	Year	Original price	Price 2018	Net Present Value (years)			Information source
					20	50	100	
Cat/fox/rabbit	Electric wire overhang with 50 mm posts	2004	$11,368	$15,941	$31,894	$31,898	$31,899	Long and Robley 2004a
Cat/fox/rabbit	Electric wire overhang with C-sction droppers	2004	$9,758	$13,683	$27,379	$27,382	$27,384	Long and Robley 2004a
Cat/fox/rabbit	Floppy-top fence with electirc wires	2004	$10,300	$14,443	$28,899	$28,902	$28,904	Long and Robley 2004a
Cat/fox/rabbit	Floppy-top fence with electirc wires	2005	$12,433	$16,976	$46,617	$50,448	$52,026	Moseby and Reid 2006
Cat/fox/rabbit	Floppy-top fence without electric wires	2004	$9,685	$13,581	$27,174	$27,178	$27,179	Long and Robley 2004a
Cat/fox/rabbit	Floppy-top fence without electric wires	2005	$11,670	$15,935	$44,535	$48,365	$49,944	Moseby and Reid 2006
Cat/fox/rabbit	Low floppy-top fence with electric wires and 30 mm netting	2005	$8,815	$12,036	$36,737	$40,567	$42,146	Moseby and Reid 2006
Cat/fox/rabbit	Low floppy-top fence with electric wires and 40 mm netting	2005	$6,939	$9,475	$31,615	$35,445	$37,024	Moseby and Reid 2006
Cat/fox/rabbit	Rigid overhang	2004	$9,895	$13,875	$27,763	$27,767	$27,768	Long and Robley 2004a
Cat/fox/rabbit	Rigid overhang	2014	$17,000	$18,158	$48,981	$52,811	$54,389	Hayward et al. 2014

Target animal	Fence type	Year	Original price	Price 2018	Net Present Value (years)			Information source
					20	50	100	
Cat/fox/rabbit	Wire netting/electric wire composite fence	2004	$7,950	$11,148	$22,308	$22,312	$22,313	Long and Robley 2004a
Feral dog/dingo	Composite	2004	$4,601	$6,452	$12,916	$12,920	$12,921	Long and Robley 2004a
Feral dog/dingo	Fabricated dog fence with no trip wire	2004	$6,663	$9,343	$18,699	$18,703	$18,704	Long and Robley 2004a
Feral dog/dingo	Fabricated dog fence with trip wire	2004	$7,522	$10,548	$21,108	$21,112	$21,113	Long and Robley 2004a
Feral dog/dingo	Sloping	2004	$2,899	$4,065	$8,143	$8,147	$8,148	Long and Robley 2004a
Fox	Sloping fox fence 1	2004	$6,237	$8,746	$17,504	$17,508	$17,509	Long and Robley 2004a
Fox	Sloping fox fence 2	2004	$6,797	$9,531	$19,075	$19,078	$19,080	Long and Robley 2004a
Goat	Fabricated 1	2004	$4,203	$5,894	$11,800	$11,804	$11,805	Long and Robley 2004a
Goat	Fabricated 2	2004	$3,355	$4,704	$9,422	$9,425	$9,427	Long and Robley 2004a
Goat	Prefabricated fence 2	2004	$3,185	$4,466	$8,945	$8,949	$8,950	Long and Robley 2004a
Pig	Electrified trip-wire fence	2004	$3,914	$5,488	$10,989	$10,993	$10,995	Long and Robley 2004a
Pig	Prefabricated fence 1	2004	$3,510	$4,922	$9,856	$9,860	$9,862	Long and Robley 2004a
Pig	Prefabricated fence 2	2004	$3,085	$4,326	$8,664	$8,668	$8,670	Long and Robley 2004a
Rabbit	With 40 mm netting	2004	$5,565	$7,803	$15,619	$15,623	$15,625	Long and Robley 2004a
Rabbit	With imported 30 mm wire	2004	$3,985	$5,588	$11,188	$11,192	$11,194	Long and Robley 2004a
Rabbit	With stock-wires	2004	$3,715	$5,209	$10,431	$10,435	$10,437	Long and Robley 2004a
Toads	Fine mesh	2017	$30,000	$31,165	$75,003	$78,834	$80,412	Southwell et al. 2017

None of the published data fulfils the sorts of detail suggested as being necessary for predictive modelling of conservation costs suggested by Iacona et al. (2018) or which have been calculated for the costs of eradication of pests on islands (Wenger et al. 2018). That will have to wait until the extra data is obtained from partner practitioners. Nevertheless, this initial analysis provides some estimate of the costs of construction per kilometre for fences designed to last from 20 to 100 years which averages $45-50,000 per kilometre at NPV.

The NPVs of the fencing costs presented in Table 2 do not include the costs of eradication of feral animals, either initially or after incursions, or the costs of stocking with native animals. These additional costs, which we have not estimated, will then be incurred from managing the fauna inside and outside the fence, both desired and undesired. These costs vary with the species, the standard of fence and the nature of the terrain, as will the costs of fence construction. Some costs will involve trade-offs. For example Norbury et al. (2012) examined the trade-offs between the cost of constructing a fence that effectively excluded all undesirable species with one that was cheaper but leaky, requiring repeated eradication following incursions. This in turn will be affected by the speed with which incursions are detected, with rapid detection greatly reducing subsequent costs of eradication. A long delay, on the other hand, may require the extra expenses of restocking should the predator eradicate the species being protected.

3.3 Ex situ conservation

Regardless of the extent to which captive breeding has contributed to conservation historically (Brichieri-Colombi 2019) and the evidence that facilities, at least for large mammals globally, are now saturated (Alroy 2015), for some species there is no alternative. For example, Christmas island skinks (though with maintenance costs much lower than quoted in Andrew et al. 2018), Orange-bellied Parrot and Regent Honeyeater are all likely to be extinct had there not been captive breeding programs. Given the inevitability of the need for captive breeding, therefore, cost estimates are helpful for planning for other species for which it will be essential, perhaps as a result of climate change (Garnett et al. 2014). Indeed, they are useful for understanding the true cost of best practice and for developing cost-effective strategies for species where there may be a choice between ex situ and in situ conservation (Kissel et al. 2017).

Detailed data on the ex situ costs of management were made available for 39 collections of 31 threatened species in five animal taxa (Table 3) with relatively few reptiles and only one invertebrate. The difference between the two is that some institutions had multiple collections across their portfolio of sites and there were some species being maintained in multiple collections. The costs we have calculated to date are currently incomplete and more are being assembled. For example, the 44 species that are part of regional species management programs were costing $350,000 for genetic management in 2012 (Hogg 2013), the equivalent of $8,900 each species at current pricing, but this is not yet reflected in the costs obtained.

Table 3. Number of species and collections for which costing data was available.

Animal type	No species	No. collections
Invertebrate	1	1
Amphibian	8	11
Reptile	7	6
Bird	6	11
Mammal	9	11
All	31	39

In total the 39 collections were cared for by an average of about one person each although the FTE (Table 4) were sometimes spread widely across multiple keepers, managerial, support and research staff although keeper staff dominated the employment with over 80% of employees involved with the threatened species being keepers. Science support may have been under-estimated as some of the research was undertaken by PhD students and the costs of their tuition and supervision are unlikely to have been included. There was also variable acknowledgement of institutional infrastructure costs supporting the on-ground work by keepers.

Table 4. Average and total number of people employed (FTE) for each animal type across three captive breeding institutions

	Employee type					
Animal type	Keeper	Manager	Science	Support	All types	Total employees
Invertebrate	0.14	0.00	0.05	0.07	0.26	0.26
Amphibian	0.71	0.16	0.06	0.00	0.93	10.20
Reptile	4.00	0.67	0.20	0.00	4.88	29.25
Bird	0.65	0.10	0.07	0.00	0.82	8.98
Mammal	1.30	0.10	0.10	0.01	1.50	16.50
Average FTE	0.86	0.12	0.07	0.00	1.06	41.28
Percentage	81.5	11.1	7.0	0.4		

It is generally cheaper to keep invertebrates and frogs or reptiles than birds or mammals (Table 5) with the latter costing a little more than double per collection. Of this, salaries makes up a little under two thirds of the costs of keeping the animals with consumables and equipment much of the rest. Food is generally a small part of the cost except under exceptional circumstances, such as the $30,000 p.a. contract required for the collection of termites to feed captive numbats.

Table 5. Annual costs of keeping threatened species at three Australian institutions

Animal type	Salaries	Consumables	Equipment	Maintenance	Research	Facilities[1]	Total
Invertebrate	$27,239	$58,411	-	-	-	$9,105	$85,650
Amphibian	$100,595	$17,830	$2,812	$545	-	$6,070	$121,782
Reptile	$80,481	$14,375	$4,229	$833	-	$664,419	$99,918
Bird	$81,066	$27,908	$15,517	$13,676	$2,091	$341,121	$223,166
Mammal	$159,448	$26,124	$1,139	$4,011	$12,728	$108,181	$203,451
All types	$110,793	$24,070	$29,526	$5,271	$3,590		$173,840
Percentage[2]	63.9	13.9	17.0	3.0	2.1		

1. Facility costs based on information from 2 frogs, 3 birds, 1 mammal and 2 reptiles; 2. Excludes facility costs

Excluding facility costs, the costs per individual animal maintained could be estimated for 21 species (Table 6). Mammals were the most expensive, but the amphibians were surprisingly high on a per capita basis, although tadpoles were excluded from this analysis. Reptiles were much the cheapest, requiring little keeper time or food.

Table 6. Cost per animal for 21 species kept in captivity (Numbers in brackets are the number of species collections for which numbers were available)

Animal type	$/individual
Amphibian (6)	$9,918
Bird (6)	$6,332
Mammal (7)	$12,036
Reptile (2)	$1,902
Average (5.25)	$7,727

There is little other information in the international literature on keeping threatened species so costs are likely to be most relevant if calculated at a national level. For example, three species kept in Brazil, Pekin robin (*Leiothrix lutea*), Livingstone's turaco (*Tauraco livingstonii*) and Valley quail (*Callipepla californica*), cost $469, $331 and $182 respectively for their daily upkeep (Cruz et al. 2016). These costs are substantially lower than what we found for Australia. Of more relevance are the cost proportions. For example, labour costs for lammergeyer (*Gypaetus barbatus*) captive breeding in Andorra are about 80% of total costs (Margalida et al. 2017) as they are with the three costed examples of frog breeding in Canada (Kissel et al. 2017).

3.4 Cost Data Aggregation

3.4.1 Objectives and priorities

All recovery plans report budgets against their objectives, and sometimes against individual actions. The more detailed plan allows managers to ensure that all actions do have budgets – several of the case studies included actions but failed to budget for anything to happen. The detailed budget also allows recovery plan managers to understand resource allocations in terms of staff time etc. that need to be committed for actions to proceed, especially where limited budgets means that only high priority actions are affordable. However, for the four case study plans, just under half the actions were considered very high priority and there were none below medium priority with much variation (Table 7). These results suggest a need for standardisation in prioritisation for the exercise to be sufficiently rigorous to be used for budgetary setting (see threat section below). Priorities also tended to be set at the level of objectives rather than at the level of individual sub-actions, making it difficult to separate the actions that would make most difference to lowering risk of extinction in a short time from those needed for long term threat amelioration and population growth.

Table 7. Proportion of costs in different priority categories in four reconstructed recovery plan budgets

Priorities	Boggomoss Snail	Swift Parrot	Macquarie Perch	Central Rock-Rat	Average
Very High (81-100%)	17.86	63.29	56.73	65.37	50.81
High (61-80%)	55.38	22.15	27.88	1.72	26.78
Medium (41-60%)	26.76	14.56	15.39	32.91	22.40

3.4.2 IUCN and Conservation Measures Partnership action categories

Most recovery plans list actions with an internal logic that is not readily comparable between plans. For any single objective, the action types may vary from on-ground action through planning and public outreach. However, each proposed action can readily be classified within one of the action classification schemes. These in turn can be aggregated across plans to understand the scale of investment needed for different areas. Differences among plans partly reflect the level of knowledge about how to manage threats. For some there is a heavy research component because little is known about the target organism, the threats facing it or how to manage those threats. For others, the actions required are known and it is action on the ground that is required. However, such analyses can also reveal weightings in the proposed budget allocations that do not seem commensurate with the threats faced.

While extra funding does not necessarily mean that an action is more effective, it can lead to questioning of recovery plan structure. For example, the allocation in the Swift Parrot to research currently seems disproportionate to the urgency of action needed to prevent its extinction. Also, the summaries in Table 8 may reflect a patchy commitment to monitoring given that about 10% is allocated to monitoring of public expenditure in fields like health and education (Lindenmayer et al. 2012). However, research and monitoring are listed separately only under the IUCN system with the Conservation Measures Partnership approach giving greater emphasis to investments in socially-directed actions.

Table 8. Percentage allocations to IUCN and Conservation Measures Partnership action categories in four reconstructed recovery plan budgets

Action Type	Boggomoss Snail	Swift Parrot	Macquarie Perch	Central Rock-Rat	Average
IUCN					
1. Land/water protection	17.46	8.25	1.25	0	6.74
2. Land/water management	15.18	4.98	30.50	20.09	17.69
3. Species management	0	5.04	3.74	7.06	3.96
4. Education and awareness	3.73	12.05	4.86	0	5.16
5. Law and policy	1.57	10.24	4.37	0.31	4.12
6. Livelihood, economic and other incentives	0	0	0.98	0	0.25
7. External capacity building	0	1.39	0	1.47	0.72
8. Research	57.76	29.60	26.15	52.48	41.50
9. Conservation planning	1.06	14.13	20.72	0.92	9.21
10. Monitoring	3.24	10.93	7.43	17.68	9.82
11. Other	0	3.39	0	0	0.85
Action Type	**Boggomoss Snail**	**Swift Parrot**	**Macquarie Perch**	**Central Rock-Rat**	**Average**
Conservation Measures Partnership					
1. Land/Water Management	32.64	2.23	32.39	20.09	21.83
2. Species Management	0	2.34	4.05	1.85	2.06
3. Awareness Raising	3.73	7.10	5.44	1.47	4.44
4. Law Enforcement & Prosecution	0	1.62	3.56	0	1.30
5. Livelihood, Economic & Moral Incentives	0	0	5.23	0	1.31
6. Conservation Designation & Planning	0	18.83	14.57	0	8.35
7. Legal & Policy Frameworks	1.57	10.96	0.91	0.31	3.44
8. Research & Monitoring	62.06	53.87	33.84	70.16	54.98
9. Education & Training	0	0	0	0	0
10. Institutional Development	0	3.05	0	6.13	2.30

3.4.3 Partners

Recovery plans currently list the roles partners can potentially play in each action but do not clarify the scale of commitment required. Many organisations, particularly different branches of government, are implicitly expected to fulfil roles and commit funds either as cash or as a form of in-kind, though still publicly funded, through support for the institutional infrastructure that underpins the recovery effort. Landholders are also often listed as supporting a recovery programme, but their time and investment is omitted from budgets. In the four case study recovery plans most of the financial responsibility is assumed to be taken by state/territory governments and research organisations with volunteers contributing in two of the cases, a pattern not immediately evident from the list of actions (Table 9). The analysis also suggests that Indigenous benefit from most of these examples may be relatively low.

Table 9. The percentage of the budget attributed to different types of recovery plan partners in four reconstructed plan budgets

Organisation	Boggomoss Snail	Swift Parrot	Macquarie Perch	Central Rock-Rat	Average
Commonwealth Government	0	0	0	0.57	0.14
State/territory government	40.50	31.59	56.12	53.51	45.43
Local government	5.04	0	0.59	0	1.41
Indigenous organisations	0	1.07	2.96	10.60	3.66
Research organisation	41.54	22.60	22.21	34.68	30.26
Conservation NGO	0	4.13	3.02	0.64	1.95
Private landholders	12.92	0	0	0	3.23
Volunteers	0	40.62	15.09	0	13.93

3.4.4 Costs of threat alleviation and downlisting

The relationship between the list of actions and the list of threats in a recovery plan is rarely explicit. When a methodology recently developed by Garnett et al. (2019) is applied to the threats and the expected progress estimated in terms of changes in threat intensity, increases in knowledge of how to manage the threats and increases in the management of the threat, four things become apparent. First, a substantial proportion of the investment (on average 30% in total) could not be attributed to any of the threat categories (Table 10). This suggests that either money is being misdirected to activities that do not deal with substantive threats, the threats have not been fully described in the recovery plans or the actions are not sufficiently well justified so do not explain which threats they are intended to alleviate or how.

Table 10. Investment directly attributed to threats in four reconstructed recovery plans

Investment category	Boggomoss Snail	Swift Parrot	Macquarie Perch	Central Rock-Rat
01 Residential & commercial development		$115,424		
02 Agriculture & aquaculture	$58,215	$599,416		
03 Energy production & mining		$31,157		
04 Transportation & service corridors		$19,437		
05 Biological resource use		$1,022,082	$301,691	
07 Natural system modifications	$51,586		$744,620	$172,498
08 Invasive & other problematic species, genes & diseases	$65,752	$1,599,007	$1,702,219	$784,661
09 Pollution			$870,013	
11 Climate change & severe weather		$47,096		
12 Other options			$11,547	
Not directly attributable to any action	$408,201	$3,387,013	$371,477	$454,601
% not directly attributable to any action	69.93	49.66	10.04	32.2

Second, with additional analyses, it will be possible to model the expected costs of dealing with threats across multiple threatened species. Current costings tend to deal solely with actions for individual species or with individual threats without there being specific attribution to individual species. Attribution of costs to threats in recovery plans makes the link clear.

Third, it is possible to estimate the amount of money that it is anticipated will be required to reduce a threat, increase knowledge or achieve a level of threat reduction by a percentage point. In doing so the recovery plan sets up a set of testable hypotheses in addition to targets. By making explicit in the recovery plan the expected change in threat intensity for a given investment, subsequent accounting at the end of the recovery planning period will not only make it possible to determine whether a target has been met but also the relationship between the cost estimate and the actual amount eventually expended on the specified action. Work aiming to make this link is currently under way but has not been completed.

3.5 Employment, multipliers and regional development

The direct investment of funds for threatened species conservation, as with any enterprise, has economic flow-on effects in terms of expenditure beyond its original purpose including the creation of extra employment, income, and tax-receipts. Initial recipients of payments for services involved in implementing a program or project then spend some of that money locally, and the shop keepers then spend some of those additional receipts locally, and so on. The additional benefits beyond the original investment are reflected in indirect and induced multipliers. For example, if the multiplier is 2.0, an initial investment of $1 ends up increasing local expenditures by an addition $1, giving $2 in total.

Normal practice in BCA is to exclude multipliers from the calculation of benefits and costs (Maddison Jacob Association 2005), because the process of collecting revenue to spend on the program or project would itself have negative multiplier effects elsewhere in the economy, which may cancel out the overall multiplier benefits. However, if supporting a particular region such as a rural area is a particular social priority, then information about local multiplier effects can be relevant.

We aim to estimate the level of local multiplier effects resulting from investment in threatened species conservation. The indirect multipliers (Type I) are those that relate to changes in sales, employment and income within business or society that are directly linked to the investment (second-round effect). For example, the supplier of fuel for helicopters or bait for traps. Each business that provides goods or services to a threatened species project benefits indirectly from the amount spent by that project.

Type I multiplier = (direct investment + indirect investment)/ direct investment

Induced multipliers (later-round effects) are those that relate to changes in sales, employment and household spending of income earned either directly or indirectly from spending by those involved with the threatened species project. For example, a threatened species manager will buy food at local shops, visit local hairdressers and send their children to local schools; all of which then generates other jobs and other expenditure. Indirect and induced effects are termed a Type II multiplier.

Type II multiplier = (direct investment + indirect investment + induced investment)/ direct investment

Such multiplier effects can have important economic and social benefits for local communities. For example, Fleming and Measham (2015) describe how unconventional gas development is attracting young skilled workers to rural Queensland and reversing rural decline while Stoeckl et al. (2011), Gibson and Connell (2016) and Rolfe (2019) examine the same effects emanating from tourism. BenDor et al. (2015) analyse this for environmental investments, describing it as part of the restoration economy which they calculate as employing about 126,000 workers and generating US$9.5 billion in annual sales, thereby creating an additional 95,000 jobs and $15 billion in economic indirect spending. In Australia, investment in Indigenous land and sea management has substantial flow-on benefits to communities beyond the direct environmental benefits, exceeding those of mining and agriculture (Jarvis et al. 2018).

Multiplier effects are easily over-estimated (Sinclair and Sutcliffe 1982) with the extent and rate of leakage of direct investment benefits varying greatly based on local conditions. Induced expenditure is particularly likely to be spent beyond the local community so that multipliers derived from national studies often exaggerate benefits at the regional or local level (Archer and Owen 1971). The type of model is also important, with social accounting matrices producing higher multiplier effects than input-output models or computable general equilibrium models (Crompton et al. 2016). Importantly, as recognised with computable general equilibrium models, investment in threatened species incurs opportunity costs elsewhere in the economy.

Against this, however, is the argument that threatened species are mostly away from major urban centres that generate much of the taxable wealth used to support threatened species investment and the multiplier effects of direct investment are a major reason governments support rural and regional communities in an effort to stem or reverse rural decline. Thus, acknowledging greater sensitivity in input-output data in regional economies over time and notwithstanding concerns about the accuracy of multiplier effects, they nevertheless represent real phenomena at a regional level and warrant estimating as an outcome of threatened species investment.

For the four case study recovery plans, substantial (Table 11) and employment (Table 12) benefits are likely to be derived if they were funded. International expenditure can be seen as a loss to the Australian economy (it is for the import of specialist tracking equipment and for camera traps) but these losses are far outweighed by the benefits that could be generated in outer regional and remote communities. If actions are made spatially explicit across all the regions in which threatened species actions could occur, modelling will be possible of the extent to which threatened species can contribute to larger policy goals with respect to rural and regional Australia.

Table 11. Application of economic multipliers to potential investment in threatened species as a result of funding recovery plans* ($'000s, all investments are without in-kind or infrastructure costs)

Remoteness	Direct investment	Indirect investment	Induced investment	Total
Expenditure				
Major city	$5,651 ($6,369)	$3,387 ($3,817)	$6,674 ($7,522)	$15,712 ($17,707)
Inner Regional	$9,568 ($10,867)	$6,204 ($7,053)	$12,227 ($13,899)	$27,999 ($31,819)
Outer Regional	$3,003 ($3,350)	$1,937 ($2,167)	$3,816 ($4,271)	$8,756 ($9,789)
Remote	$1,048 ($1,125)	$462 ($489)	$911 ($963)	$2,422 ($2,577)
Very Remote	$9 ($9)	$3 ($3)	$5 ($6)	$16 ($18)
International	$240 ($248)	$143 ($148)	$282 ($291)	$665 ($688)
Total	$19,520 ($21,969)	$12,136 ($13,675)	$23,916 ($26,951)	$55,572 ($62,596)
Percentage of total				
Major city	28.86 (28.99)	27.87 (27.91)	27.87 (27.91)	28.22 (28.29)
Inner Regional	48.81 (49.57)	50.99 (51.57)	50.99 (51.57)	50.22 (50.83)
Outer Regional	15.32 (15.25)	15.92 (15.85)	15.92 (15.85)	15.71 (15.64)
Remote	5.74 (5.12)	4.02 (3.57)	4.02 (3.57)	4.62 (4.12)
Very Remote	0.05 (0.04)	0.02 (0.02)	0.02 (0.02)	0.03 (0.03)
International	1.23 (1.13)	1.18 (1.08)	1.18 (1.08)	1.20 (1.10)

** The figures are based on 7% and 3% (in parenthesis) discount rates.*

Table 12. Potential employment (FTE) resulting from funding four case study recovery plans*
(all investments are without in-kind or infrastructure costs)

Remoteness	Direct investment	Indirect investment	Induced investment	Total
Potential employment				
Major city	6.79 (6.79)	10.13 (11.40)	11.93 (13.42)	28.84 (31.61)
Inner Regional	5.72 (5.72)	18.53 (21.07)	21.82 (24.81)	46.08 (51.60)
Outer Regional	4.43 (4.43)	5.78 (6.47)	6.81 (7.62)	17.03 (18.53)
Remote	1.25 (1.25)	1.46 (1.46)	1.72 (1.72)	4.43 (4.43)
Very Remote	0.0 (0.0)	0.01 (0.01)	0.01 (0.01)	0.02 (0.02)
International	0.0 (0.0)	0.43 (0.44)	0.51 (0.52)	0.94 (0.96)
Total	18.20 (18.20)	36.34 (40.85)	42.79 (48.10)	97.33 (107.15)
Percentage of total				
Major city	37.30 (37.30)	27.87 (27.91)	27.91 (27.91)	29.63 (29.50)
Inner Regional	31.46 (31.46)	50.99 (51.57)	51.57 (51.57)	47.34 (48.15)
Outer Regional	24.36 (24.36)	15.92 (15.85)	15.85 (15.85	17.49 (17.29)
Remote	6.89 (6.89)	4.02 (3.57)	4.02 (3.57)	4.55 (4.14)
Very Remote	0.0 (0.0)	0.02 (0.02)	0.02 (0.02)	0.02 (0.02)
International	0.0 (0.0)	1.18 (1.08)	1.18 (1.08)	0.96 (0.90)

** The figures are based on 7% and 3% (in parenthesis) discount rates.*

4. Future research direction

Based on the recovery plan costing analysis presented, conservation investment decisions ought to be based on estimation of direct and indirect as well as market and non-market benefits and costs. A range of future research directions are suggested to generate further evidence which in turn will help improve the efficiency and benefit from species recovery investments in Australia.

4.1 Direct use benefits

Wildlife tourism is a large and growing industry with some threatened species being suitable for the industry where threatened species would be the prime source to generate direct tourism revenue in the form of user fees and indirect spill-over benefits on local businesses and communities. Identifying such direct, indirect and induced benefits of threatened species in a range of conservation contexts would provide policy relevant information to decision makers to weigh species recovery costs against the direct, indirect and induced benefits they generate.

4.2 Non-market values and benefits

Not all benefits of threatened species are monetary. Valuation and consideration of non-monetary benefits of threatened species would help raise societal awareness as well as justifying species recovery plan budgets. Research just initiated within the same project (NESP 6.1), of which recovery plan costing is one component, will provide a non-market value for threatened species that can also be incorporated into recovery plan budgets. Estimation of the non-market values of a greater number of threatened species, either through original valuation studies or through benefit transfer approach, where relevant, would help in making species conservation investment decisions.

4.3 Indigenous benefits

In the case study recovery plans, Indigenous benefits were relatively low. However, the analysis of recovery plan budgets makes it possible not only to consider Indigenous interests in threatened species but also plan from the start opportunities for their direct involvement in their conservation and the links between Indigenous land and sea management and recovery objectives. Generating and expanding the evidence base on direct and indirect/induced benefits of investment in threatened species conservation to Indigenous communities, particularly in rural and remote locations are useful to measure effectiveness of such investments.

4.4 Partnership costs and benefits

Threatened species recovery involves a great deal of collaboration and partnership among different stakeholders and agencies. Partnership has both costs and benefits, but a good understanding of the value and effectiveness of such partnerships is generally lacking from the literature. Estimating the costs and benefits of partnerships as well as identifying the success factors for effective partnerships would be highly useful for developing species recovery plans and allocating budgets for specific actions under partnerships. This action would complement the work being undertaken under NESP project 6.6.

4.5 Modelling recovery plan costs

With additional case studies, the costs of actions described in this report could be modelled and extrapolated across multiple threatened species to understand the likely costs of threatened species recovery at larger scales. This modelling could also include an understanding of realistic costs of threat abatement for threatened species in multiple settings.

Acknowledgements

We are grateful to the many partner organisations and professionals who have helped conceptualise and develop this report. We are particularly grateful to the officials of the Department of the Environment and Energy (Terry Hills, Sandra Walpole, Peter Latch, Juanita Watters) for providing guidance through meetings and discussion on recovery plan costing work. We are also thankful to James Brazill-Boast, New South Wales Office of Environment and Heritage, for his valuable insights and information on recovery plan costing in NSW. For advice on the costs of ex situ conservation we wish to thank Michael Magrath, Zoos Victoria, Andrew Elphinstone and Bronwyn Parker of Taronga Park Zoo, Peter Mawson of Perth Zoo, Michael Johnson of Moonlit Sanctuary and Peter Gowland of Priam Australia Pty Ltd and for advice on fencing costs we appreciate the assistance of Katherine Moseby and Michael Bode.

Appendix

The appendix for this report is available online here:
https://www.dropbox.com/sh/9wne7h3z1m5mbux/AAB5rC02NZDP-Rh15n2CE0ria?dl=0

It includes excel spreadsheets of detailed recovery plan costings for four example threatened species and a combined table. The example species are:

- Boggomoss snail (EPBC Status: Critically Endangered)
- Central rock-rat (EPBC Status: Critically Endangered)
- Swift parrot (EPBC Status: Critically Endangered)
- Macquarie perch (EPBC Status: Endangered)

The spreadsheets include tabs for:

1. Budget overview
2. Strategy summary
3. Economic impact
4. Threat alleviation
5. IUCN actions
6. CAP actions
7. Actions total
8. Organisation totals
9. Staff summary
10. Recovery plan comparison
11. Actions
12. Partners
13. Staff costs non-casuals
14. Staff costs casuals
15. Consultants
16. Travel Vehicles
17. Travel airfares
18. Travel allowances
19. Travel other
20. Consumables
21. Facilities
22. Equipment
23. Personnel costs in-kind
24. Travel vehicles in-kind
25. Travel accommodation in-kind
26. Threat progress scores
27. Threat action attribution
28. Threat weighting
29. Threat types
30. Salary Table (internal only)
31. Lists (internal only)
32. Remoteness
33. Action Categories

References

Allen Consulting Group. 2008. Recognising the full costs of university research. Report to the Department of Innovation, Industry Science and Research, Canberra.

Allen Consulting Group. 2011. Assessment of the economic and employment outcomes of the working on country program. Report to the Department of Sustainability, Environment, Water, Population and Communities. Allen Consulting Group, Canberra.

Alroy, J. 2015. Limits to captive breeding of mammals in zoos. *Conservation Biology*, 29(3), pp. 926–931.

Archer, B. H. & Owen, C. B. 1971. Towards a tourist regional multiplier. *Regional Studies*, 5, 289–294.

Arid Recovery. 2019. Reserve layout. Arid Recovery. <https://aridrecovery.org.au/reserve-layout> accessed 05 Aug 2019.

Australian Bureau of Statistics. 2018. 1270.0. 55.005 – Australian Statistical Geography Standard (ASGS): Volume 5 – Remoteness Structure, July 2016, Canberra. <https://www.abs.gov.au/ausstats/abs@.nsf/mf/1270.0.55.005> accessed 05 Aug 2019.

Australian Government. 2019. Paying your employees. Australian Government. <https://www.business.gov.au/people/hiring/pay-and- conditions/paying-your-employees> accessed 05 August 2019.

Bendor, T., Lester, T. W., Livengood, A., Davis, A. & Yonavjak, L. 2015. Estimating the size and impact of the ecological restoration economy. *PLoS one*, 10(6), e0128339.

Bode, M. & Wintle, B. 2010. How to build an efficient conservation fence. *Conservation Biology*, 24(1), pp.182–188.

Brichier-Colombi, T. A., Lloyd, N. A., McPherson, J. M. & Moehrenschlager, A. 2019. Limited contributions of released animals from zoos to North American conservation translocations. *Conservation Biology*, 33(1), 33–39.

Brightsmith, D. J., Stronza, A. & Holle, K. 2008. Ecotourism, conservation biology, and volunteer tourism: A mutually beneficial triumvirate. *Biological Conservation*, 141, 2832–2842.

Burns, B., Innes, J. & Day, T. 2012. The use and potential of pest-proof fencing for ecosystem restoration and fauna conservation in New Zealand. In *Fencing for Conservation: Restriction of Evolutionary Potential or a Riposte to Threatening Processes?* (eds M. J. Somers & M. W. Hayward), pp. 65–90. Springer, New York.

Campbell-Hunt, D. M. 2008. Ecotourism and sustainability in community driven ecological restoration: Case studies from New Zealand. *Sustainable Tourism III* (eds C. A. Brebbia & F. D. Pineda). *Ecology and the Environment*, 115, pp. 231–240. WIT Press, Southampton.

Clarke, M., Gillespie, R. & Cunningham, S. 2017. *Regional Economic Multiplier Impacts Potential Pollinator Deficits Across Crops.* Rural Industries Research and Development Corporation. Barton, Australia.

Coinnews Media Group LLC. 2019. US inflation calculator. <https://www.usinflationcalculator.com/> accessed 05 Aug 2019.

Commonwealth of Australia. 2018. National Recovery Plan for Macquarie perch (*Macquaria australasica*). Commonwealth of Australia, Canberra.

Commonwealth of Australia. 2019. National Recovery Plan for the swift parrot (*Lathamus discolor*). Commonwealth of Australia, Canberra.

Conservation Measures Partnership (CMP). 2016. Threats and Actions Classifications (v2.0). <http://cmp-openstandards.org/tools/threats-and-actions-taxonomies> accessed 22 Nov 2019.

Cook, C., Pullin, A., Sutherland, W., Stewart, G. & Carrasco, L. 2017. Considering cost alongside the effectiveness of management in evidence based conservation: A systematic reporting protocol. *Biological Conservation*, 209, 508–516.

Crompton, J. L., Jeong, J. Y. & Dudensing, R. M. 2016. Sources of variation in economic impact multipliers. *Journal of Travel Research*, 55(8), 1051–1064.

Cruz, C. E. F., Cerva, C. & Andretta, I. 2016. Financial costs of conserving captive-bred wild birds. *Der Zoologische Garten*, 85(6), 354–362.

de Torres P. J. & Marlow, N. 2011. The relative merits of predator exclusion fencing and repeated fox baiting for protection of native fauna: Five examples from Western Australia. Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes? *Biological Conservation*, 142(1), 1–13.

Department of Prime Minister and Cabinet. 2016. Guidance Note – Cost Benefit Analysis. <https://www.pmc.gov.au/sites/default/files/publications/cosst-benefit-analysis.docx> accessed 05 Aug 2019.

Edwards, P. E. T., Sutton-Grier, A. E. & Coyle, G. E. 2013. Investing in nature: Restoring coastal habitat blue infrastructure and green job creation. *Marine Policy*, 38, 65–71.

Fleming, D. A. & Measham, T. G. 2015. Local economic impacts of an unconventional energy boom: The coal seam gas industry in Australia. *Australian Journal of Agricultural and Resource Economics*, 59(1), 78–94.

Garnett, S. & Franklin, D. Eds. 2014. *Climate Change Adaptation Plan for Australian Birds*. CSIRO Publishing, Melbourne.

Garnett, S. T., Butchart, S. H. M., Baker, G. B., Bayraktarov, E., Buchanan, K. L., Burbidge, A. A., Chauvenet, A. L. M., Christidis, L., Ehmke, G., Grace, M. & Hoccom, D. G. 2019. Metrics of progress in the understanding and management of threats to Australian birds. *Conservation Biology*, 33(2), 456–468.

Gerber, L. R., Runge, M. C., Maloney, R. F., Iacona, G. D., Drew, C. A., Avery-Gomm, S., Brazill-Boast, J., Crouse, D., Epanchin-Niell, R. S., Hall, S. B. & Maguire, L. A. 2018. Endangered species recovery: A resource allocation problem. *Science*, 362(6412), 284–286.

Gibson, C. & Connell, J. 2016. *Music Festivals and Regional Development in Australia*. Routledge, Farnham, UK.

Hayward, M. W., Moseby, K. & Read, J. L. 2014. The role of predator exclosures in the conservation of Australian fauna. *Carnivores of Australia: Past, Present and Future* (eds Alistair Glen & Christopher Dickman), pp.353–371, CSIRO Publishing, Melbourne.

Helmstedt, K. J. & Possingham, H. P., 2017. Costs are key when reintroducing threatened species to multiple release sites. *Animal Conservation*, 20(4), 331–340.

Hogg, C. J. 2013. Preserving Australian native fauna: Zoo-based breeding programs as part of a more unified strategic approach. *Australian Journal of Zoology*, 61(1), 101–108.

Iacona, G. D., Sutherland, W. J., Mappin, B., Adams, V. M., Armsworth, P. R., Coleshaw, T., Cook, C., Craigie, I., Dicks, L. V., Fitzsimons, J. A. & McGowan, J. 2018. Standardized reporting of the costs of management interventions for biodiversity conservation. *Conservation Biology*, 32(5), 979–988.

International Union for Conservation Of Nature (IUCN) & Conservation Measures Partnership (CMP). 2012. Unified Classification of Direct Threats. Gland, Switzerland. <http:/s3.amazonaws.com/iucnredlist- newcms/staging/public/attachments/3127/dec_2012_guidance_threats_classification_scheme.pdf> accessed 22 Nov 2019.

Ireland, L., Zabek, M., Galindez-Silva, C., Weir, S., West, R., Olds, L., Backhouse, B., Copley, P. & Read, J. 2018. More than just the animals: Opportunities and costs of reintroducing threatened black-footed rock-wallabies to remote Indigenous land. *Pacific Conservation Biology*, 24(4), 388–396.

Jarvis, D., Stoeckl, N., Hill, R. & Pert, P. 2018. Indigenous land and sea management programs: Can they promote regional development and help "close the (income) gap"? *Australian Journal of Social Issues*, 53(3), 283–303.

Kelmenson, S., Bendor, T. & Lester, T. W. 2016. *The Economic Impacts of the US Ecological Restoration Sector.* University of North Carolina, Chapel Hill, US.

Kissel, A. M., Palen, W. J. & Govindarajulu, P. 2017. A decision-theory approach to cost-effective population supplementation for imperilled species. *Ecological Economics*, 142, 194–202.

Lamarque, F., Anderson, J., Chardonnet, P., Fergusson, R., La Grange, M., Osei-Owusu, Y., Bakker, L., Belemsobgo, U., Beytell, B., Boulet, H., Soto, B. & Tabi Tako-Eta, P. 2008. Human-wildlife conflict in Africa: An overview of causes, consequences and management strategies. FAO Internet Report, Washington D. C. <http://www.wildlife-conservation.org/en/content/download/1034/4568/ fi le/HumanWildlifeConflict2008.pdf> accessed 22 Nov 2019.

Laughland, D., Phu, L. & Milmoe, J. 2014. *Restoration Returns: The Contribution of Partners for Fish and Wildlife Program and Coastal Program Restoration Projects to Local U.S. Economies.* U.S. Fish & Wildlife Service. University of Southern Maine, Portland, US. <https://digitalcommons.usm.maine.edu/cgi/viewcontent.cgi?referer=https://scholar.google.com.au/&httpsredir=1&article=1042 &context=cbep-publications> accessed 10 Jul 2019.

Legge, S., Woinarski, J.C., Burbidge, A.A., Palmer, R., Ringma, J., Radford, J.Q., Mitchell, N., Bode, M., Wintle, B., Baseler, M. & Bentley, J., 2018. Havens for threatened Australian mammals: The contributions of fenced areas and offshore islands to the protection of mammal species susceptible to introduced predators. *Wildlife Research*, 45(7), 627–644.

Lindenmayer, D. B., Gibbons, P., Bourke, M. A. X., Burgman, M., Dickman, C. R., Ferrier, S., Fitzsimons, J., Freudenberger, D., Garnett, S. T., Groves, C. & Hobbs, R. J. 2012. Improving biodiversity monitoring. *Austral Ecology*, 37(3), 285–294.

Lindsey, P. A., Masterson, C. L., Beck, A. L. & Romañach, S., 2012. Ecological, social and financial issues related to fencing as a conservation tool in Africa. *Fencing for Conservation*, pp. 215–234. Springer, New York.

Long, K & Robley, A. 2004. *Catalogue of Fence Designs*. Natural Heritage Trust, Australia.

Long, K & Robley, A. 2004. *Cost Effective Feral Animal Exclusion Fencing for Areas of High Conservation Value in Australia.* Natural Heritage Trust, Australia.

Maddison Jacob Associates. 2005. *Frameworks for Economic Impact Analysis and Benefit-Cost Analysis.* Maddison Jacob Associates, Camberwell, Vic.

Margalida, A., Martínez, J. M., Gómez De Segura, A., Colomer, M. A., Arlettaz, R. & Serrano, D. 2017. Supplementary feeding and young extraction from the wild are not a sensible alternative to captive breeding for reintroducing bearded vultures *Gypaetus barbatus*. *Journal of Applied Ecology*, 54(1), 334–340.

McDonald, P., Ward, S., Nano, C., Pavey, C., Nano, T. & Nunn, P. 2017. National Recovery Plan for the central Rock-rat, *Zyzomys pedunculatus*. Northern Territory Department of Environment and Natural Resources, Alice Springs, NT.

Moseby K. E. & Read J. L. 2006. The efficacy of feral cat, fox and rabbit exclusion fence designs for threatened species protection. *Biological Conservation*, 127(4), 429–437.

Nichol, M., Shi, H. & Campi, S. 2013. Economic Impact Analysis on Tourism on Tasmania's King Island. Report prepared for Tourism Tasmania and the Cradle Coast Authority.

Nielsen-Pincus, M. & Moseley, C. 2010. Economic and Employment Impacts of Forest and Watershed Restoration in Oregon. Working Paper No. 24. University of Oregon, Eugene, US.

Norbury, G., Hutcheon, A., Reardon, J. & Daigneault, A., 2014. Pest fencing or pest trapping: A bio-economic analysis of cost-effectiveness. *Austral Ecology*, 39(7), 795–807.

Office of Environment And Heritage. 2016. More Plants and Animals to be Saved from Extinction. OEH, New South Wales Government, Sydney.

RACQ. 2019. Private vehicle expenses 2019. <https://www.racq.com.au/cars-and-driving/cars/owning-and-maintaining-a-car/car- running-costs> accessed 22 Nov 2019.

Reserve Bank of Australia. 2019a. Exchange rates. Reserve Bank of Australia. <https://www.rba.gov.au/statistics/frequency/exchange- rates.html> accessed 05 Aug 2019.

Reserve Bank of Australia. 2019b. Inflation calculator. Reserve Bank of Australia. <https://www.rba.gov.au/calculator/annualDecimal.html> accessed 05 Aug 2019.

Reserve Bank of New Zealand. 2019. Inflation calculator. <https://www.rbnz.govt.nz/monetary-policy/inflation-calculator> accessed 05 Aug 2019.

Rolfe, J. 2019. Simple economic frameworks to evaluate public investments in sporting events in regional Australia. *Economic Analysis and Policy*, 63, 35–43.

Royuela, J. B., Parejo, S. H., De La Cruz, A., Geraldes, P., Costa, L. T. & Gil, A. 2017. The socio-economic impact of conservation: The Safe Islands for Seabirds LIFE project. *Oryx*, 53(1), 1–8.

Scofield, R. P., Cullen, R. & Wang, M. 2011. Are predator-proof fences the answer to New Zealand's terrestrial faunal biodiversity crisis? *New Zealand Ecological Society*, 35, 312–317.

Sinclair T. & Sutcliffe C. 1982. Keynesian income multipliers with first and second round effects: An application to tourist expenditure. *Oxford Bulletin of Economics and Statistics*, 44, 321–338.

Southwell, D., Tingley, R., Bode, M., Nicholson, E. & Phillips, B. L. 2017. Cost and feasibility of a barrier to halt the spread of invasive cane toads in arid Australia: Incorporating expert knowledge into model-based decision-making. *Journal of Applied Ecology*, 54(1), 216–224.

Stanisic, J. 2008. Recovery Plan for the boggomoss snail *Adclarkia dawsonensis*. Department of the Environment, Water, Heritage and the Arts, Canberra and Environmental Protection Agency, Brisbane.

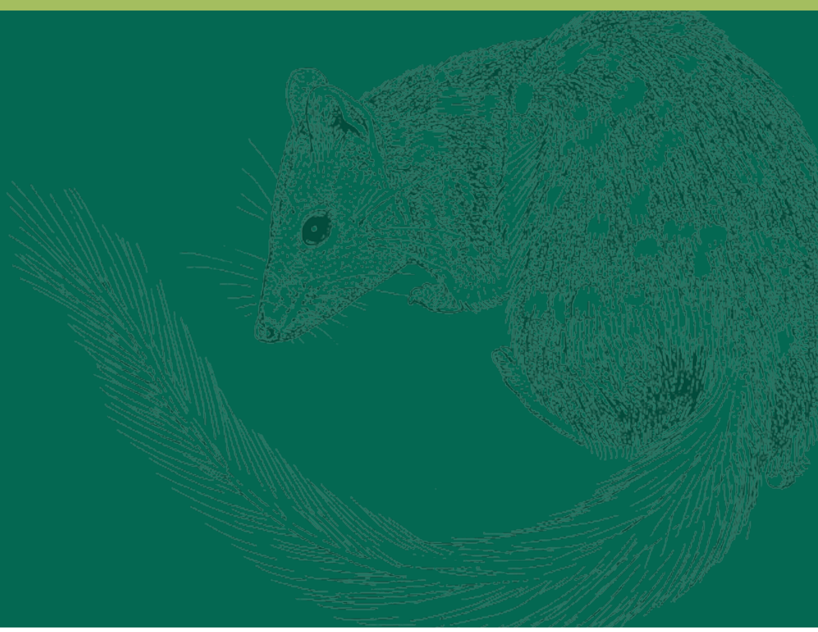
Stephens, S. & Maxwell, S. (Eds). 1996. *Back from the Brink: Refining the Threatened Species Recovery Process.* Surrey Beatty & Sons, Chipping Norton, UK.

Stoeckl, N., Esparon, M., Stanley, O., Farr, M., Delisle, A. & Altai, Z. 2011. Socio-economic Activity and Water Use in Australia's Tropical Rivers: A Case Study in the Mitchell and Daly River Catchments. Final report for The Tropical Rivers and Coastal Knowledge Research Consortium. Charles Darwin University, Darwin, NT.

URS. 2007. Final Report: Benefit-Cost Analysis of the State Barrier Fence. Report to the Department of Agriculture and Food, Western Australia.

Wenger, A. S., Adams, V. M., Iacona, G. D., Lohr, C., Pressey, R. L., Morris, K. & Craigie, I. D., 2018. Estimating realistic costs for strategic management planning of invasive species eradications on islands. *Biological Invasions*, 20(5), 1287–1305.

Yin, R. K. 2017. *Case Study Research and Applications: Design and Methods*. Sage, Thousand Oaks, US.



Further information:

http://www.nespthreatenedspecies.edu.au/

This project is supported through funding from the
Australian Government's National Environmental Science Program.

Threatened Species Recovery Hub

National Environmental Science Programme