A comparative approach to assess drivers of success in mammalian conservation recovery programmes

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Abstract

The outcomes of species recovery programmes have been mixed, with high-profile population recoveries contrasting with species-level extinctions. Although each conservation intervention faces its own challenges, it is imperative to assess whether such lessons have wider general applicability. To contribute towards evidence-based improvement of future conservation strategies, we conducted global-scale quantitative analysis of 48 mammalian recovery programmes based on peer-reviewed literature and semi-structured interviews with conservation scientists and practitioners, investigating ecological, management and political factors associated with population recoveries or declines. The importance of identifying and removing threats was shown strongly by our results, emphasizing that populations are likely to continue to be negatively impacted if threats are not reduced or removed. Our analysis also highlights the importance of management strategies such as robust threat monitoring. Small population size and lack of habitat were associated with longer-term dependence on conservation intervention; this demonstrates the importance of increasing population numbers quickly, and restoring and protecting habitat to ensure long-term population recovery. Informants also cited poor stakeholder coordination and management as key weaknesses in recovery programmes, indicating the importance of effective leadership and shared goals and management plans. Project outcomes were not influenced by ecological variables, suggesting that recommendations from our results are applicable to other recovery programmes. Our study demonstrates the value in conducting quantitative comparative assessments of factors influencing success in conservation interventions. We encourage further such studies, particularly at more geographically localised scales, and recommend that the conservation community continues to evaluate and learn lessons from past experiences and adapt future strategies accordingly.

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Introduction

It is widely accepted that we are experiencing a global biodiversity crisis. Vertebrate populations for which long-term data are available have on average declined globally by 52% since 1970 (WWF 2014), and ≥25% of mammal species are threatened with extinction (Schipper et al. 2008). Wide-scale and long-term attempts to mitigate anthropogenic impacts have been insufficient to halt or reverse global biodiversity loss (Butchart et al. 2010). Threatened species recovery is considered an important example of 'micro-scale' conservation (Sodhi et al. 2011) but the outcomes of such interventions remain mixed. Indeed, only 24 species in one recent analysis underwent a positive change in conservation status from 1996-2008 compared with 171 that deteriorated (Hoffmann et al. 2011).

High-profile recent mammalian conservation successes include population recovery of southern white rhino (*Ceratotherium simum simum*) (Amin et al. 2006) and black-footed ferret (*Mustela nigripes*) (Miller et al. 1996), whilst well-publicised losses include extinction of the Yangtze River dolphin (*Lipotes vexillifer*) (Turvey 2008) and the Vietnamese subspecies of Javan rhino (*Rhinoceros sondaicus annamiticus*) (Brook et al. 2014). These mammals were all the focus of recovery programmes, and it is not immediately clear why certain conservation strategies succeeded and others failed. The combination of a species' biology, its ecological, political and social environment (the 'operating environment'), and threat type all interact to create unique conservation challenges requiring diverse, often bespoke approaches and responses. However, some of these factors may transcend context and predispose a project towards certain outcomes. It is therefore imperative to learn lessons from past successes and failures in conservation recovery, to maximize effectiveness of future interventions and minimize ongoing biodiversity loss (Ferraro 2009).

Conservation evaluation is a small but growing area of conservation science, with recent industry-wide efforts to establish guidelines on conservation project evaluation (Kapos et al. 2008; CMP 2013; Hopkins et al. 2015). Previous qualitative evaluations of single or contrasting case studies have highlighted specific issues driving past conservation success and failure (Miller et al. 1996; Martin et al. 2012). While these are insightful on a case-by-case basis, it is unclear whether such issues are context-specific or have wider applicability (Hutchings et al. 2012). Conversely, several studies that have carried out reviews of multiple conservation projects have largely focused on integrated conservation and development programmes, with limited attention on correlates of success in species recovery (Waylen et al. 2010; Brooks et al. 2013). Few studies have included a large enough sample size to conduct quantitative analyses that assess the effect of different operating environments on species recovery outcomes, but this approach would contribute greatly to the current scientific evidence-base for informing conservation planning.

As species recovery programmes have now been underway for several decades, ample data are potentially available for long-term analyses of their efficacy. Statistical analysis across a wide range of projects can permit identification of common features associated with varying likelihood of project success or failure (e.g., Abbitt & Scott 2001). These factors may be intrinsic (e.g., species biology) or extrinsic (e.g., project management). Such analysis could constitute a powerful tool for determining likely success of future conservation programmes operating under different scenarios. This in turn could help managers and policy-makers choose appropriate strategies to maximise likely effectiveness of potential recovery activities.

We aimed to provide a new baseline of conservation evidence by conducting global-scale quantitative analysis of a large set of mammalian recovery programmes, representing a wide range of taxonomic groups, life-histories and conservation challenges. This analysis aimed to determine whether it was possible to identify common factors associated with population recovery or decline, and thus assist with improvement of future conservation strategies under different operating environments.

Methods

To understand the relationships between causal factors and conservation outcomes, and identify potential determinants of conservation success, we generated an initial list of possible interventions and variables through focus-group discussion with conservation scientists and practitioners. This list was subsequently refined by three of the authors (JJC, HMRM, STT).

Project selection

Conservation interventions vary enormously in scope. To avoid comparison of projects with substantially different aims, we chose targeted recovery programmes that aimed to increase population size of the focal species. We defined a recovery programme as '*a coordinated initiative comprising linked conservation actions that seek to directly mitigate threats to a species and increase its population (or the populations of interest)*'. To minimize taxonomic variation within our sample we only investigated conservation activities for mammals, a well-studied group that has been the focus of numerous recovery programmes.

We selected projects by contacting Specialist Group coordinators and chairs from the International Union for Conservation of Nature (IUCN) Species Survival Commission (SSC) and requesting information on species within their group that were subject to recovery programmes as defined above, and by using the IUCN Species Information Service (SIS) database. We only included species with sufficient information in the literature to complete as a case study and/or for which we were able to find a relevant contact to interview. Our final dataset comprised 48 recovery programmes focused on an entire threatened species, subspecies, or specific population, and included a wide range of species and locations (Table 1).

Response variables

Numerous methods have been proposed for evaluating conservation ‘success’ (Kapos et al. 2008; Howe and Milner-Gulland 2012). However, success is frequently determined through achievement of project goals (Saterson et al. 2004). We therefore assessed population trend (overall trajectory of population/species since start of recovery programme) as a primary measure of success, defined as a binary variable: 1=Extinct/decline; 2=Stable/increase (Table 1).

However, most recovery programmes also aim to ensure that target populations are self-sustaining with minimal need for long-term direct management (Redford et al. 2011). We therefore included the additional variable of conservation dependence, which quantified the degree to which focal population(s) required ongoing conservation intervention to maintain recovery; this was considered a secondary measure of success, and defined as: 1=Extinct; 2=Intensively managed; 3=Lightly managed; 4=Conservation-dependent; 5=Self-sustaining (Table 1), based on definitions in Redford et al. (2011). A sixth category, “Captive managed”, was excluded as our dataset contained no species representing this category.

Explanatory variables

We organised our final set of explanatory variables into six areas (species biology/ecology; geopolitical environment; threats; baseline information; stakeholders and management; funding), and developed a standardized questionnaire based on these categories, with each question representing a potential variable (Supplementary Information). Life-history data were obtained from Jones et al. (2009), habitat types were based on IUCN (2013), biogeographic realms were defined according to Olson et al. (2001), and Human Development Index data were taken from UNDP (2013). We obtained information on recovery programmes from peer-reviewed conservation literature, and semi-structured interviews conducted verbally or through correspondence with relevant contacts involved with specific recovery programmes currently or in the past. Due to time constraints on data collection, a maximum of two people were interviewed per recovery programme. To account for potential differences in informant perspectives on factors associated with project outcomes, we gathered information from both the literature and interviews where possible, or used multiple, independent literature sources when interviewees were unavailable. All subsequent statistical and descriptive analysis preserved respondent anonymity. Interviews also gathered extensive qualitative data on examples of good and bad practice in species recovery, which are discussed below.

Data analysis

We employed initial exploratory tests (Pearson's correlation and chi-square tests) to eliminate variables that were correlated or lacked substantive explanatory power, resulting in a set of 20 explanatory variables for subsequent analysis (Table 2). To explore factors influencing recovery programme outcomes, we modelled the response variables of population trend and conservation dependence against explanatory variables under univariate analysis using, respectively, binomial logistic regression and ordinal logistic regression in the R package 'ordinal'. We included significant variables into full models and applied model simplification, deleting variables with highest p-values to produce a Minimum Adequate Model (Crawley 2007). To assess significance of changes in deviance resulting from removal of terms, we compared models using *F*-tests rather than chi-square tests due to overdispersion in our data (Crawley 2007). All analyses were undertaken in R version 2.15.2 (R Core Team 2013).

Results

Of the 48 mammalian populations in our study for which conservation action had been undertaken, 33 were stable or increasing and 15 were declining or extinct (Table 1). The commonest 'intensive' conservation interventions (where individual animals were manipulated/managed to some degree) were *ex situ* conservation breeding and translocation, whilst the commonest 'non-intensive' interventions (where only the environment was manipulated/managed) included community engagement and habitat protection/restoration. Although intervention type was treated as a single variable within our analysis, we were unable to include it as a predictor variable as all projects had >1 intervention, and our overall sample was too small to account for this. Average project length was 24.3±11.4 years and was not a significant predictor of improved conservation outcome. No explanatory variables related to species biology/ecology, geopolitical environment, baseline information or funding were significantly related to either response variable. We did not encounter conflicting informant responses associated with specific project outcomes.

Threat reduction was significantly associated with both increasing population trend and decreasing conservation dependence under univariate analysis, and was retained under both multivariate models (Table 3). The commonest threat was habitat loss (reduction, degradation and fragmentation); human-induced mortality, primarily hunting and persecution, was also a major threat. Novel threat emergence (e.g., dam development for Irrawaddy dolphins (*Orcaella brevirostris*); increase in disease prevalence in mountain gorillas (*Gorilla beringei*) through tourism) was associated, although not significantly, with increased likelihood of population decrease/extinction (Table 3). Although 85% of focal populations were protected by national-level legislation, this did not predict recovery programme outcomes; however, low levels of law enforcement were significantly associated with increased likelihood of population decrease/extinction (Table 3). Lack of available habitat and small population size were cited as limiting factors within 56% and 42% of recovery programmes respectively, but were not statistically associated with population recovery or decline. However, both were associated with increased conservation dependence under univariate analysis; small population size remained a significant predictor of long-term conservation dependence in multivariate analysis (Table 3).

Neither response variable was statistically associated with predictor variables around stakeholders or management structure. However, >55% of all projects with stable/increasing populations were associated with general stakeholder agreement, with only 21% associated with weak stakeholder agreement. By contrast, only 33% of projects with extinct/declining populations were associated with general stakeholder agreement, with 40% associated with weak stakeholder agreement (Figure 1).

Discussion

Our study investigating factors related to conservation recovery programme success constitutes a novel step towards developing a global quantitative comparative framework to identify mechanisms that improve likelihood of species recovery under different operating environments.

Identifying and mitigating threats

The importance of accurate identification and removal of threats to improve population trajectories in both the short- and long-term was demonstrated strongly in our analysis. Although seemingly intuitive, it highlights that even if certain aspects of a recovery programme (e.g., community engagement, captive breeding) are successful, wild populations will continue to be negatively impacted if threats are not reduced or eliminated. Indeed, recent species or population extinctions in our dataset (e.g., Yangtze River dolphin, Vietnam rhino) were closely associated with a lack of effective effort to mitigate continuing external threats. Whilst we agree that threat abatement is insufficient to ensure recovery (Hutchings et al. 2012), it is clearly a necessity that must be acknowledged from the outset of conservation planning.

We argue that the association between novel threat emergence and population decline demonstrates the need to undertake thorough threat analysis during project planning, continue monitoring threats, and adapt strategies in response to new and potential threats. Monitoring (and acting on monitoring-derived data) is a key part of conservation evaluation (Stem et al. 2005), but its usefulness within conservation, and the resources it should be allocated, are debated (McDonald-Madden et al. 2010; Geupel et al. 2011). Many conservation programmes have inadequate monitoring and evaluation systems (Stem et al. 2005), often due to absence of measurable goals (Lindenmayer & Likens 2009; Stephenson & Ntiamoa-Baidu 2010), and some populations or species such as the Christmas Island pipistrelle (*Pipistrellus murrayi*) have even been 'monitored to extinction' (Martin et al. 2012). Our results emphasize that threat identification and monitoring should be incorporated into project monitoring and evaluation plans, and data used regularly for adaptive management (McCarthy et al. 2012).

Although most focal populations were protected by national-level legislation, our case studies included numerous examples of considerable weakness in practical law enforcement. For example, Brook et al. (2014) specifically attributed the extinction of Javan rhino in Vietnam to "*poaching, facilitated by weak enforcement of anti-poaching and anti-trafficking laws*", and inadequate protection is linked to rhino declines elsewhere in Asia and Africa (Amin et al. 2006). Conversely, increased investment in anti-poaching enforcement has successfully protected populations of rhinos and other species (Amin et al. 2006; Hilborn et al. 2006). As a further example, the Alaotran gentle lemur (*Hapalemur alaotrensis*) is threatened by illegal burning of marshland for rice cultivation and access by fishermen around Madagascar's Lake

Alaotra, a temporary protected area and Ramsar site, but despite closed fishing seasons there is little enforcement, partly due to lack of government funding (Copsey et al. 2009). As one informant noted, "*the future is not that bright for this species unless there are radical changes in the way that environmental rules and laws, whether traditional local ones or national ones, are enforced*". We therefore strongly recommend that resources for enforcement must be fully integrated and costed into recovery plans where illegal activity is a known primary limitation to population recovery.

Short- versus long-term recovery goals

Young et al. (2014) found that a minimum 11-year recovery time was needed before species improved in conservation status. In our study project length was not associated with population trend or conservation dependence, and our dataset included young projects showing population recovery and well-established projects struggling to increase populations, as well as vice-versa. Although recovery time is likely linked to focal species life-history, our results suggest that other factors may be more important than time spent on project.

Small population size and habitat limitation were not associated with population recovery, but were associated with long-term conservation dependence. Our results therefore suggest that it may be helpful to distinguish different phases of recovery programmes (Linklater 2003). The first is removal of a species or population from immediate danger of extinction by increasing numbers as quickly as possible; this tends to be the primary goal of most recovery programmes and is a fundamental principle of conservation theory (Frankham & Ralls 1998; Courchamp et al. 2008).

The second phase is a longer-term process of recovery to achieve multiple robust, healthy and self-sustaining populations requiring minimal conservation input. This distinction may be beneficial in conservation recovery planning, as different phases may require distinct goals and timelines to be anticipated and managed proactively rather than reactively.

Where habitat limitation is a known barrier to recovery, restoring, protecting and increasing habitat should be a key conservation action, otherwise species may recover from near-extinction only to exist in captivity, or with wild populations that are non-viable or with little ecological function. Several of our case studies highlighted that long-term management may also require re-evaluation of species' habitat requirements, and recognition that areas where surviving individuals occur may constitute suboptimal habitat. For example, remnant populations of Cape mountain zebra persisted in montane fynbos-dominated areas assumed to constitute appropriate habitat. However, subsequent research has demonstrated that zebras would naturally have migrated up and down mountains to find suitable grazing, and only recently became restricted to isolated fynbos patches (Faith 2012). Even where meta-population management is the explicitly stated recovery strategy, expanding habitat or creating habitat corridors should still constitute a key conservation action in such situations, if species are not to remain heavily conservation-dependent.

Stakeholders and management

Informants commonly cited stakeholder conflict as a major reason for project failure, from obstructive individuals, to recovery teams unable to agree on common management approaches, to conflicts with political figures stalling conservation efforts at a policy level. Neither response variable was statistically associated with any predictor variables around stakeholders or management structure, possibly due to difficulties with capturing this complex information in a quantitative measure (Black et al. 2011). However, all projects where species became extinct and two-thirds of those where species were declining were characterised by partial or total lack of coordinated management, and stakeholders with separate agendas. This invariably led to lack of clear aims and delays in implementing conservation interventions.

In contrast, over half of projects associated with general stakeholder agreement had increasing target populations (Figure 1). In these instances an active centralised working group generally facilitated effective conservation efforts, with species managed as a coordinated whole rather than separate populations for long-term recovery. We found that effective working groups tended to meet at least annually and with more regular informal updating. Regularity of contact also facilitated adaptive management, with changes in recovery programme trajectory evaluated and updated as necessary with everyone's agreement. It should be noted that elsewhere in the conservation literature the establishment of special working groups was correlated with increased innovation in translocation programmes of rare species, but not increased organisation or decreased conflict as found in our study (Reading et al. 1997). The relationship between working groups and effective conservation is therefore likely to be complex and context-dependent. However, effective leadership and management practices are likely to improve project performance, e.g., through appropriate working group coordination (Black et al. 2011, 2013). Effective stakeholder coordination may also be related to different capacity levels of partners that affect their ability to influence conservation decision-making; for example, lack of investment and subsequent limited capacity in many government wildlife agencies was regularly cited as a key issue by informants in this study. It was interesting that funding was not related to conservation outcomes here, particularly as links between increased funding and improved conservation status have been found elsewhere (Kerkvliet & Langpap 2007). However, our informants regularly made the important distinction that continuity of funding was more important to effective long-term recovery programme management than simply amount of funding, a concern which has not been commonly discussed in the conservation literature.

Conflict with government or policy-makers was also a commonly reported issue. One informant, reflecting on a difficult political relationship with a national government, commented, "*We were right to go on record and say there's a problem […] you have also got to realise that the people who run the country, run the country; we are just an NGO and you need to work with them to get anything done […] I think really careful political engagement is absolutely vital, and we'd be a lot further along if we'd been more adept at that several years ago*". We suggest that many projects could benefit from involvement of specific politically-adept individuals to help liaise with governments, and this may be worth considering in difficult cases. Although this is not a commonly acknowledged challenge in the conservation literature, Martin et al. (2012) argued that institutional accountability was a vital prerequisite for avoiding species extinctions, notably lacking for the Christmas Island pipistrelle. Key decision-makers must be identified early and political engagement managed carefully; this can constitute an essential step towards successful conservation outcomes (Phillis et al. 2013).

Future directions

Our global-scale quantitative approach has revealed several common predictors of recovery or decline across a wide sample of recovery programmes, some of which have received relatively little attention in the conservation literature. No biological variables were found to influence project outcomes, so we would expect that recommendations from our results could be applied to other recovery programmes. This could reflect the strong understanding of mammalian biology and ecology that usually forms part of the evidence-base for species recovery; alternatively, external factors may simply be more influential in project outcomes. An interesting next step would be to repeat this study for other taxonomic groups to identify whether these patterns hold more widely.

Although we had a reasonable geographical spread of projects, >37% were from the USA and Australia. This was partly due to legislation in these countries supporting identification of threatened species and establishment of recovery programmes, meaning that there were more existing projects in these countries than in most others. However, to tackle potential bias, further analyses could be focused within specific geographical or political regions. This would also help to yield more localized insights into factors influencing recovery programme outcomes and improve ability to apply lessons learned in a targeted manner. Perhaps most importantly it would be useful to generate larger project sample sizes for purposes of statistical analysis, including comparable-sized sets of conservation successes and failures, to help identify stronger associations between explanatory variables and project outcomes.

Recovery programmes must be planned (including use of proper threat and stakeholder analysis), implemented and monitored according to best practices, as well as tailored to suit specific situations. Other factors that we have not explicitly considered in this study, such as project cost-effectiveness (Naidoo et al. 2006), must also be incorporated into decision-making for recovery programmes. However, our findings demonstrate the importance of considering management strategies such as robust threat monitoring, long-term habitat protection and effective stakeholder coordination. Above all the conservation community must recognize the importance of regular evaluation and learning lessons from past experiences, to replicate successful strategies and avoid repeating potentially grave and irreversible mistakes.

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Table 1. List of species for which recovery programmes were assessed[a].

Scientific name	**Common name(s)**	**Order**	**Country(ies) within which species assessed**	**Population trend**	**Conservation dependence**	**Project length (years)**
Ailuropoda melanoleuca	Giant panda	Carnivora	China	→	LM	34
Arvicola terrestris	Water vole	Rodentia	UK	↑	LM	16
Beatragus hunteri	Hirola	Artiodactyla	Kenya	↓	IM	19
Bettongia penicillata	Woylie	Diprotodontia	Australia	↓	LM	18
Brachylagus idahoensis	Columbia Basin pygmy rabbit	Lagomorpha	USA	→	IM	13
Bubalus mindorensis	Tamaraw	Artiodactyla	Philippines	→	CD	35
Bunolagus monticularis	Riverine rabbit	Lagomorpha	South Africa	→	CD	16
Canis lupus baileyi	Mexican gray wolf	Carnivora	USA	↑	IM	32
Capra falconeri jerdoni	Sulainam markhor	Artiodactyla	Pakistan	↑	CD	29
Ceratotherium simum simum	Southern white rhino	Perissodactyla	South Africa	↑	CD	56
Coleura seychellensis	Seychelles sheath-tailed bat	Chiroptera	Seychelles	↑	LM	17
Cynomys ludovicianus	Black-tailed prairie dog	Rodentia	USA	→	IM	16
Dasyurus geoffroii	Chuditch, western quoll	Dasyuromorphia	Australia	→	LM	23
Dicerorhinus sumatrensis	Sumatran rhino	Perissodactyla	Indonesia, Malaysia	↓	IM	30
Equus zebra zebra	Cape mountain zebra	Perissodactyla	South Africa	↑	IM	64
Gorilla beringei	Mountain gorilla	Primates	Uganda, Rwanda, Democratic Republic of Congo	↑	CD	23
Hapalemur alaotrensis	Alaotran gentle lemur	Primates	Madagascar	↓	LM	24
Hypogeomys antimena	Malagasy giant rat	Rodentia	Madagascar	↑	CD	12
Lasiorhinus krefftii	Northern hairy-nosed wombat	Diprotodontia	Australia	↑	IM	22
Leontopithecus chrysopygus	Black lion tamarin	Primates	Brazil	↑	IM	27
Leontopithecus rosalia	Golden lion tamarin	Primates	Brazil	↑	IM	31
Lipotes vexillifer	Baiji, Yangtze River dolphin	Cetacea	China	EX	EX	28

[a] Key: EX = Extinct, IM = Intensively Managed, LM = Lightly Managed, CD = Conservation Dependent, SS = Self-sustaining

Lycaon pictus	African wild dog	Carnivora	South Africa	↑	IM	16
Lynx pardinus	Iberian lynx	Carnivora	Spain	↑	IM	12
Macrotis lagotis	Greater bilby	Peramelemorphia	Australia	↓	IM	23
Marmota vancouverensis	Vancouver Island marmot	Rodentia	Canada	↑	IM	26
Monachus schauinslandi	Hawaiian monk seal	Carnivora	Hawaii, USA	↓	LM	34
Muscardinus avellanarius	Common dormouse, hazel dormouse	Rodentia	UK	↓	LM	20
Mustela nigripes	Black-footed ferret	Carnivora	USA	↑	IM	27
Myrmecobius fasciatus	Numbat	Dasyuromorphia	Australia	↑	LM	29
Neotoma floridana smalli	Key Largo wood rat	Rodentia	USA	→	IM	26
Nomascus hainanus	Hainan gibbon	Primates	China	→	IM	11
Onychogalea fraenata	Bridled nailtail wallaby	Diprotodontia	Australia	↓	IM	23
Orcaella brevirostris	Irrawaddy dolphin	Cetacea	Cambodia	↓	CD	13
Panthera pardus orientalis	Amur leopard	Carnivora	Russia	→	LM	18
Petrogale penicillata	Brush-tailed rock wallaby	Diprotodontia	Australia	→	IM	18
Phocarctos hookeri	New Zealand sea lion	Carnivora	New Zealand	↓	CD	19
Pipistrellus murrayi	Christmas Island pipistrelle	Chiroptera	Christmas Island, Australia	EX	EX	10
Porcula salvania	Pygmy hog	Artiodactyla	India	↓	IM	19
Potorous gilbertii	Gilbert's potoroo	Diprotodontia	Australia	→	IM	20
Puma concolor coryi	Florida panther	Carnivora	USA	↑	IM	33
Rhinoceros sondaicus annamiticus	Javan rhino	Perissodactyla	Vietnam	EX	EX	16
Rhinoceros unicornis	One-horned rhino, Indian rhino	Perissodactyla	Nepal	↑	CD	55
Saguinus oedipus	Cotton-top tamarin	Primates	Brazil	→	LM	29
Sarcophilus harrisii	Tasmanian devil	Dasyuromorphia	Australia	↓	IM	10
Trichechus manatus latirostris	Florida manatee	Sirenia	USA	↑	LM	38
Urocyon littoralis	Island fox	Carnivora	Channel Islands, USA	↑	LM	15
Vulpes velox	Swift fox	Carnivora	USA	↑	SS	20

Table 2. Independent variables investigated as possible predictors of mammalian recovery programme success or failure.

Variable	Categories
Biology/ecology	
Order	Artiodactyla, Carnivora, Cetacea, Chiroptera, Dasyuromorphia, Diprotodontia, Lagomorpha, Peramelemorphia, Perissodactyla, Primates, Rodentia, Sirenia
Body mass	Range: 4 – 2,285,939 g
Habitat type[b]	Forest, savannah, shrubland, grassland, desert, wetland, rocky, marine, mixed
Geopolitical environment	
Biogeographical realm[c]	Nearctic, Palearctic, Afrotropic, Indomalaya, Australasia, Neotropic, Oceania
Human Development Index (HDI)[d]	Range: 0.463 – 0.938
Political support	Conflict/No support, Passive/partial/intermittent support, Active/continuous support
Threats	
Threat reduction	None, some, most, all
Threat escalation	None, moderate, substantial
Novel threat emergence	No or yes
Law enforcement	Ineffective/weak across range, partial in Protected Areas (PAs) only, partial inside and outside PAs, effective in PAs only, effective across range
Lack of habitat as limiter to recovery	No or yes
Small population as limiter to recovery	No or yes
Baseline information	
Length of time since project start	Number of years
Data confidence	None/status unknown, low, reasonable, high
Number of publications since the start of the recovery programme	Range: 1 – 259

[b] IUCN (2013) Habitat Classification Scheme
[c] Olson et al. (2001)
[d] United Nations Development Programme (UNDP 2013)

Stakeholders and management	
Management structure	Informal collaboration between stakeholders, formal collaborative recovery team/working group, formal recovery team led by e.g. government
Stakeholder agreement	Weak, partial, general
Community support	Persistent conflict, intermittent conflict/polarised support, none/neutral, general support, strong support
Funding	
Continuity of funding	1 year or less, 1-3 years, 3+ years
Actions delayed due to funding	Never/rarely, occasionally, regularly, always

Table 3. Results of logistic regressions and ordinal logistic regressions to investigate potential predictors of population trend and conservation dependence in mammalian recovery programmes (n=48).

	Estimate	SE	z-value	p-value
1. UNIVARIATE REGRESSIONS				
POPULATION TREND				
(Intercept)	-0.981	0.677	-1.449	
Threat reduction (Some)	2.280	0.819	2.785	**
Threat reduction (Most)	2.590	1.288	2.011	*
Threat reduction (All)	18.547	2284.101	0.008	
(Intercept)	1.386	0.456	3.037	*
Novel threats (Yes)	-1.269	0.667	-1.903	·
(Intercept)	-1.179	1.080	-1.659	·
Level of enforcement (Partial PAs only)	2.197	1.414	1.554	
Level of enforcement (Partial across range)	19.358	2284.102	0.008	
Level of enforcement (Effective PAs only)	3.045	1.345	2.263	**
Level of enforcement (Effective across range)	3.127	1.191	2.625	*
CONSERVATION DEPENDENCE				
Threat reduction (Some)	2.091	0.910	2.298	*
Threat reduction (Most)	2.368	1.283	2.516	*
Threat reduction (All)	3.229	1.283	2.516	*
Habitat limitation (Yes)	-1.579	0.583	-2.708	**
Small population (Yes)	-1.386	0.595	-2.329	*
2. MULTIVARIATE REGRESSIONS				
POPULATION TREND				
(Intercept)	-0.981	0.677	-1.449	
Threat reduction (Some)	2.280	0.819	2.785	**
Threat reduction (Most)	2.590	1.288	2.011	*
Threat reduction (All)	18.547	2284.101	0.008	
CONSERVATION DEPENDENCE				
Threat reduction (Some)	2.315	0.935	2.477	*
Threat reduction (Most)	2.062	1.143	1.803	·
Threat reduction (All)	3.261	1.313	2.482	*
Small population (Yes)	-1.573	0.626	-2.511	*

· p<0.1, *p<0.05, **p<0.01

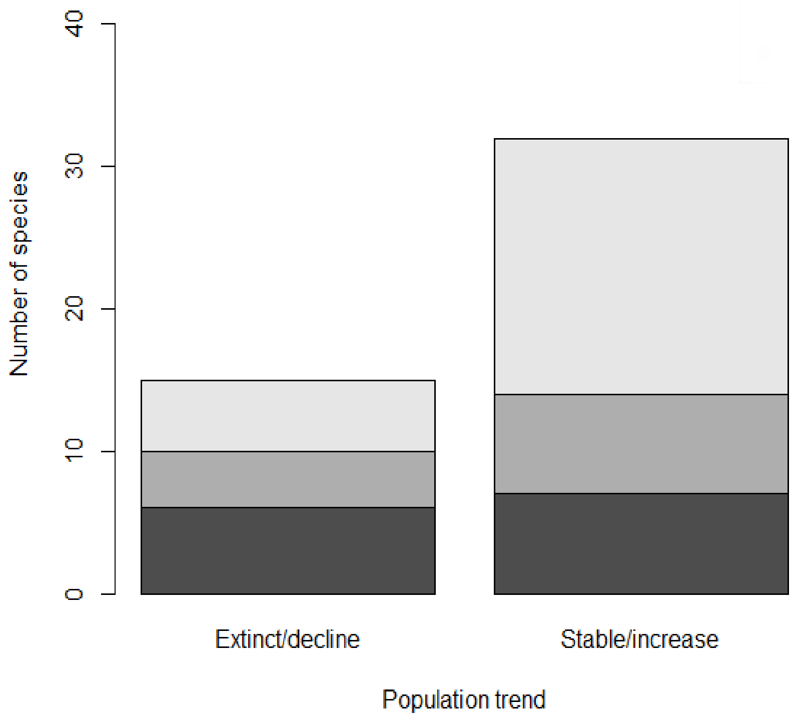

Figure 1. Population trend and level of agreement between stakeholders[e].

[e] Key: dark grey=weak agreement; medium grey=partial agreement; pale grey=general agreement.