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Science-based, stakeholder-inclusive and participatory conservation planning helps reverse the decline of threatened species

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ABSTRACT

Reversing the decline of threatened species is a target for the Convention on Biological Diversity but current efforts are failing. An integrative, multi-stakeholder approach to species conservation planning, which includes population viability analyses and both in situ and ex situ management consideration, could improve outcomes for some of the most challenging cases. The IUCN Species Survival Commission (SSC) uses such a planning approach, however, evidence of improved outcomes for species has to date been anecdotal. To assess the impact of planning, we accessed 35 species conservation plans completed in 23 countries over 13 years from the IUCN SSC database and matched them with independently generated Red List assessments of extinction risk. We used the Red List Index and a counterfactual approach, comparing the overall predicted extinction trend without planning with the observed trend after planning. Post-planning, threatened species declines continued, but gradually slowed, and then reversed, with an upward trend of recovery within 15 years. No species became extinct. Simulated counterfactual projections indicated outcomes would have been worse without the planning intervention; around eight species would have become extinct over that timeframe. To date, this planning approach has been applied to relatively high-profile species facing multiple threats, and where conflicting views, uncertainty, or lack of coordination among stakeholders constrain action. Opportunities to broaden application to other taxa are discussed. Our study provides evidence that science-based, participatory approaches to planning can create a turning point for threatened species by supporting stakeholders to transition quickly to more effective ways of working together.

1. Introduction

Aichi Target 12 of the 2011–2020 Convention on Biological Diversity (CBD) calls on countries to prevent extinction and ensure sustained improvement in the conservation status of known threatened species (CBD, 2010). Despite this, reviews show little progress on slowing declines (WWF, 2020; IPBES, 2019), the IUCN Red List currently reports 37,480 threatened species (IUCN, 2021), and future extinctions are predicted (Monroe et al., 2019).

Species conservation planning is one of a range of measures advocated to reverse extinction trends (Mace et al., 2018). Species conservation planning should aim to increase the effectiveness of conservation action, by ensuring that it is based on (*i*) relevant information for the species, (*ii*) well-defined goals, (*iii*) multiple perspectives, and (*iv*) agreement among those involved about what should be done (Boersma et al., 2001). Such planning, which ideally combines both social and analytical elements (Sande et al., 2005; Groves and Game, 2016), takes time and resources and is currently applied to few of the species that need it (e.g. Brazill-Boast et al., 2018; Watson et al., 2011). While recent studies provide compelling evidence that conservation action improves species status (Hoffmann et al., 2015; Butchart et al., 2006; Young et al., 2014), the way in which such successful action was planned, and whether planning supported outcomes, is rarely considered.

Evaluating the impact of planning on species is difficult, resulting in few attempts and conflicting conclusions. Although studies report that planning led to improved status of endangered species in the USA (Schultz and Gerber, 2002; Taylor et al., 2005), a further study showed it to be detrimental if not combined with substantial government funds (Ferraro et al., 2007) and an Australian study showed no effect once biases associated with prioritising species for planning were removed

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(Bottrill et al., 2011). The challenges of evaluating the impact of planning include insufficient data, protracted implementation time of plans, the potentially long timescale over which species might be expected to show signs of recovery and the difficulty of disentangling planning effects from those of other influences (Bottrill and Pressey, 2012; Watson et al., 2011). Furthermore, attempts to overcome the latter by comparing taxa with plans, to those without, require strong assumptions about equivalence that are often confounded by variables such as phylogeny and geography (Fuller et al., 2003). Finally, differences in purpose and approach complicate treatment of "planning" as a single type of intervention across multiple projects.

The Conservation Planning Specialist Group (CPSG) of the IUCN Species Survival Commission (IUCN SSC) supports diverse groups to develop species conservation plans collaboratively. Depending on project circumstances and emphases, the planning approach used is referred to variously as the "Population and Habitat Viability Assessment" (Miller and Lacy, 2003) or as the "One Plan Approach" (Byers et al., 2013; Conde et al., 2015), but its underlying principles, key elements and format are consistent (CPSG, 2020). Planning workshops are initiated and organised by government or non-government agencies in countries within the species' range. Wherever possible, all stakeholders are assembled (typically 20–60) for 3–4 days of facilitated analysis and discussion. Alongside government agencies, local communities, and academia, both in situ and ex situ species conservation communities are represented and decision-making is supported by population viability analyses. Stakeholders participate actively in decision-making, proceeding by consensus to agree a definition of successful species recovery or conservation, to analyse challenges to this, recommend solutions and commit to action. Outcomes are documented within 6–12 months (see supplementary material for further details). Though the planning tools and elements described are in use across the wider species conservation community, as far as we are aware the IUCN SSC CPSG approach is the only one that routinely integrates all these features within a standard workshop format. The approach is a good candidate for evaluation as the long period over which it has been used (>30 years), the relative stability of style and format, and the ready availability of information on planning projects, reduce some of the difficulties commonly encountered when assessing the impact of planning.

Past attempts to evaluate the impact of this approach have involved pre- and post-workshop surveys of participating stakeholders, to see how their work is affected by the planning deliberations and outputs (Vredenburg and Westley, 2003). Results indicate positive outcomes for participants, but to date no systematic studies have considered whether this is matched by an improvement in overall species conservation status. Given the effort and resources involved in this style of planning, evidence of impact would be useful to decision-makers charged with determining whether and how species planning is done. We therefore set out to fill this gap.

To assess the impact of this specific approach, we used a publicly available database of more than 250 well-documented species conservation planning projects, maintained by the IUCN SSC CPSG (http://cpsg.org/document-repository). Plans date from 1990 onwards and span more than 70 countries. To estimate progress on slowing or halting species extinctions following planning, we utilised the Red List Index (RLI) (Butchart et al., 2004, 2007; Mace et al., 2018). The RLI is calculated from the IUCN's published threat categories for individual species, which are generated by expert assessments of those species against independent criteria, with quantitative thresholds of extinction risk designed to be transparent and consistent across taxa (Mace et al., 2008). The RLI is widely used, readily interpreted by a range of audiences and has been adopted by the CBD for reporting on global species targets (IUCN, 2021).

Impact evaluation assesses the degree to which changes in outcome can be attributed to an intervention rather than to other factors, which requires knowing what outcomes would have looked like in absence of the intervention (Ferraro, 2009). In other studies, the necessary counterfactual comparison has been provided by econometric matching of species with plans, to those without them (Ferraro et al., 2007; Bottrill et al., 2011), or by eliciting the judgement of experts to estimate the counterfactual trajectories of species in absence of specific programs of conservation management (Butchart et al., 2006; Hoffmann et al., 2015; Young et al., 2014). Neither of these methods were available to us due to the wide geographic distribution of projects in the database, the long timeframe over which planning projects took place and the disproportionate number of highly threatened, high-profile, and phylogenetically distinct taxa included. As a result of these factors, no set of species without plans met the equivalence requirements of a control, and no group of experts available to us could provide informed counterfactual judgements across all projects. Instead, we used observed patterns in extinction trend *before* planning (though in the presence of conservation actions), to simulate a counterfactual extinction trend for the group *without* planning. We then compared the simulated *without* planning trends, to the observed *with* planning trends, to estimate the overall impact of the planning intervention on the species' conservation status (see Fig. 1).

This is the first use of the globally recognised Red List Index to evaluate the impact of a specific planning approach. Our work is relevant to those engaged in reversing the decline of threatened species and to planning practitioners seeking to evaluate longer-term impacts across multiple projects and taxa.

2. Methods

2.1. Building the dataset

We accessed species planning projects from the IUCN SSC CPSG database and, where possible, matched them with the IUCN Red List (RL) assessments for those species over time, to assess the impact of planning on conservation outcomes. IUCN Red List assessors assign species to one of seven RL extinction risk categories: Data Deficient (DD), Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR) and Extinct (EX). Assessments are repeated periodically (at least every 10 years for mammals and birds). To compare trends in population decline before and after planning, we selected projects for which the species involved had assessments extending either side of the planning workshop year. Projects were only included if the taxon had been assessed for the RL at least 5 years before, and at least 10 years after, the planning workshop. This asymmetry was considered reasonable because while a deterioration in species status can trigger an immediate elevation in risk category, 5 years of observed improvements are required to lower a risk category. Taxa also needed to have been assessed for the RL within 2 years of the planning workshop or to have identical categories before and after (to increase confidence in workshop-year category). Projects meeting these criteria were relatively rare in the database. Of 192 projects that were carried out before 2009 (thus allowing for at least 10 years of post-planning data): nine were excluded due to missing project information, two because they were area-based (and not species-based), and one because it was aimed at managing a feral species. Of the remaining 180 projects, 23 were either updates to a previous planning workshop or part of a workshop series and so were excluded on that basis. Thirty-six projects were for sub-populations of species and 24 for subspecies and all but one of these (*Gorilla b. beringei*) had not been assessed for the Red List, which is predominantly directed at species. A further six projects were for plant species that had also not been assessed for the Red List. Of the remaining 92 projects, one was excluded because it had a pre-existing conservation plan. Forty-six of the remaining projects were included in the study and the other 45 were excluded either because there were too few pre- or post-workshop RL assessments, because there were no assessments within 2 years of the workshop or, in a few cases, because the history of their assessments was interrupted by a 1990s change in RL categories that rendered some older categories with no

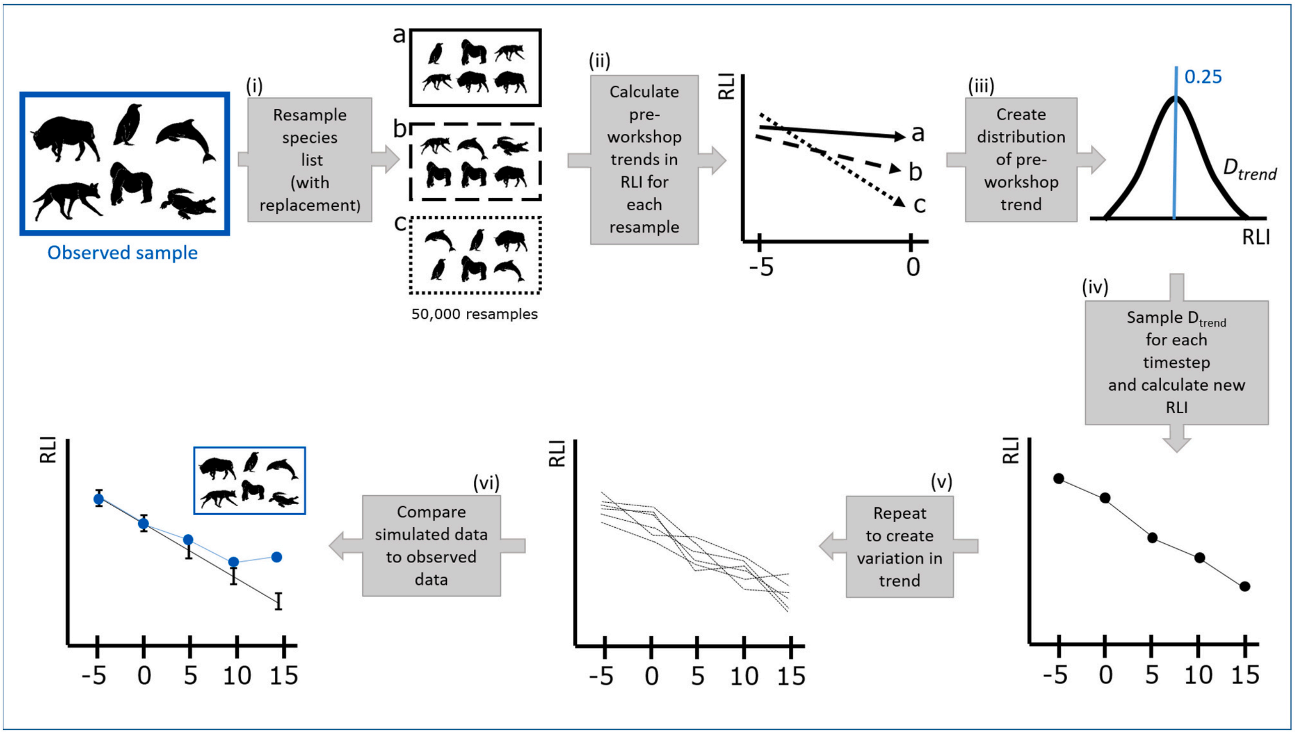

Fig. 1. Conceptual illustration of the planning impact evaluation method used in this study. We used a counterfactual approach, comparing the predicted extinction trend for species without planning with the observed trend after planning. Red List Index (RLI) uses IUCN Red List categories to measure the projected overall extinction risk over time. D_{trend} is the distribution of simulated trends in threat status over the pre-workshop period.

direct equivalent. Categories were considered current until superseded by a reassessment. For each taxon we recorded the RL category: once 5 years before the planning workshop; once at the time of the workshop; once each at 5 and 10 years following the workshop, and (data permitting) at 15 years following the workshop. The 46 taxa meeting the criteria included projects with workshops held between 1990 and 2008, in 23 countries; of these 35 had data up to 15 years post-planning. The list of 46 species included 33 mammals, nine birds, two reptiles, one amphibian and one fish. Five of the species were categorised as Critically Endangered at the time of the workshop, 30 as Endangered, four were Near Threatened or Lower Risk/Near Threatened (a pre-1995 iteration of the RL category designations, directly equivalent to NT in the current system), and seven were Vulnerable (Tables 1, S1).

2.2. The Red List Index calculation

The Red List Index (RLI) uses IUCN RL categories to measure the projected overall extinction risk of a set of species over time (Butchart et al., 2004). IUCN RL categories are weighted according to their extinction risk, ranging from $W_{LC} = 0$ for Least Concern species to $W_{EX} = 5$ for Extinct ones. The RLI reflects the proportion of species in each category and is defined by Butchart et al. (2007) as:

$$RLI_t = 1 - \frac{\sum_s W_{c(t,s)}}{W_{EX} N}$$

where $\sum_s W_{c(t,s)}$ is the sum of weights (W_c) for all assessed species (s) at a given time (t), N is the total number of assessed species and W_{EX} is the weight assigned to extinct species (i.e. = 5). For the subset of 35 species for which 15 years of post-planning data were available, we calculated five observed RLI values, ranging from 5 years before the workshop to 15 years after ($RLI_{-5,obs}$ to $RLI_{15,obs}$). In addition, for the group of 46 species with at least 10 years of post-planning data, we calculated four observed RLI values ($RLI_{-5,obs}$ to $RLI_{10,obs}$). A RL index near 1 indicates that most species in the group are Least Concern (i.e. not threatened), while a RL index near 0 indicates that most species are Critically Endangered, Extinct in the Wild or Extinct.

2.3. Statistical analyses

To evaluate the impact of planning, we developed a counterfactual prediction of group extinction trend without planning. To do so, we extrapolated the observed pre-workshop trend between RLI_{-5} and RLI_0 over the post-workshop period and compared it to the observed post-workshop trend. We used a multi-step procedure to project outcomes without planning (see Fig. 1). First, because our dataset only represents a *sample* of the global population of endangered species, we could only estimate the trend of RLI before the workshop, but no variability around this estimate, a crucial element to build the projection. To estimate the potential variation around the pre-workshop RLI trend, we used a classic bootstrap procedure (steps *i* to *iii*) (Efron, 1979). More precisely, we (*i*) resampled our dataset with replacement 50,000 times (some species can appear several times). For each resample, (*ii*) we calculated the number of changes in threat status (n) between years −5 and 0 (e.g. $n = 8$ changes) and from there, the overall trend in threat status as n divided by the total number of species (e.g. trend = 8/35 = 0.23). By combining all trends in threat status from the 50,000 bootstraps, we (iii) generated a distribution D_{trend} of simulated trends in threat status over the pre-workshop period and therefore captured variability around the observed trend of RLI before the workshop (mean = 0.25, s.d. = 0.07; i. e. one quarter of species are expected to decline by 1 category every 5 years, if pre-workshop extinction rates continue). Using these estimates, we were able to propagate this trend and the associated variation through the 20-year period for the $n = 35$ subset (steps *iv* to *v*). First (*iv*), we sampled a trend from the distribution D_{trend} (e.g. trend = 0.25) and randomly applied it to the observed dataset at year −5, thereby generating a simulated dataset of threat status at year 0. For example, for a trend of 0.23, the threat status of 23% of the species, selected randomly,

Table 1
Summary of the characteristics of projects included in the study: Wkshop Year = year in which the planning workshop was held; RL period = dates of first and last published Red List assessments (as of December 2019); Country = country in which the workshop was held; # People = number of participants listed as having attended part or all of the workshop; (orgs) = number of different organisations represented by participants (note that there may be some errors in this as some participants represented several institutions); PVA?/Ex situ recs? = presence (Y) or absence (N) of either PVA analyses or recommendations regarding ex situ management; −5 = Red List assessment category 5 years before the workshop; 0, 5, 10, 15 = Red List category at the year of the workshop and 5, 10, and 15 years after it; Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR) and Extinct (EX). LR/NT refers to Lowered Risk/Near Threatened and is equivalent to NT. All projects are housed on the CPSG website: (www.cpsg.org/document-repository).

Species common name	Scientific name	Wkshop year	RL period	Country	# People (orgs)	PVA?	Ex situ recs?	-5	0	5	10	15
Golden Lion Tamarin	*Leontopithecus rosalia*	1990	1982–2008	Brazil	46(32)	Y	Y	EN	EN	EN	CR	EN
Golden-headed Lion Tamarin	*Leontopithecus chrysomelas*	1990	1982–2003	Brazil	46(32)	Y	Y	EN	EN	EN	EN	EN
Black Lion Tamarin	*Leontopithecus chrysopygus*	1990	1982–2003	Brazil	46(32)	Y	Y	EN	EN	EN	CR	CR
Black-Footed Ferret	*Mustela nigripes*	1992	1965–2015	USA	?	Y	Y	EN	EN	EX	EN	EN
Cotton-top Tamarin	*Saguinus oedipus*	1992	1982–2008	Colombia	?	Y	Y	EN	EN	EN	EN	EN
Bornean Orangutan	*Pongo pygmaeus*	1993	1965–2016	Indonesia	40(23)	Y	Y	EN	EN	VU	EN	EN
Baiji Dolphin	*Lipotes vexillifer*	1993	1986–2008	China	43(18)	Y	Y	EN	EN	CR	CR	CR
Lion Tailed Macaque	*Macaca silenus*	1993	1986–2008	India	93(51)	Y	Y	EN	EN	EN	EN	EN
Sumatran Rhino	*Dicerorhinus sumatrensis*	1993	1986–2008	Indonesia	59(47)	Y	Y	EN	EN	CR	CR	CR
Indian Rhino	*Rhinoceros unicornis*	1993	1965–2008	India	68(60)	Y	Y	EN	EN	EN	EN	VU
Javan Gibbon	*Hylobates moloch*	1994	1986–2008	Indonesia	55(35)	Y	Y	EN	EN	CR	CR	EN
Houston Toad	*Anaxyrus houstonensis*	1994	1986–2004	USA	50(29)	Y	Y	EN	EN	EN	EN	
Marsh Deer	*Blastocerus dichotomus*	1994	1982–2016	Brazil	35(24)	Y	Y	VU	VU	VU	VU	VU
Baird's Tapir	*Tapirus bairdii*	1994	1965–2016	Panama	23(17)	Y	Y	VU	VU	VU	EN	EN
Gharial	*Gavialis gangeticus*	1995	1982–2017	India	48(31)	Y	Y	EN	EN	EN	EN	CR
European Bison	*Bison bonasus*	1995	1965–2008	Poland	29(26)	Y	Y	VU	EN	EN	EN	
Barasingha	*Rucervus duvaucelii*	1995	1986–2013	India	61(27)	Y	Y	EN	EN	VU	VU	VU
Orinoco crocodile	*Crocodylus intermedius*	1996	1986–2017	Venezuela	27(23)	Y	Y	EN	CR	CR	CR	CR
Babirusa	*Babyrousa babyrussa*	1996	1986–2008	Indonesia	37–62(?)	Y	Y	VU	VU	VU	VU	
Tamaraw	*Bubalus mindorensis*	1996	1965–2014	Philippines	37(?)	Y	Y	EN	EN	CR	CR	CR
Lowland Anoa	*Bubalus depressicornis*	1996	1965–2014	Indonesia	37–62(?)	Y	Y	EN	EN	EN	EN	EN
Mountain Anoa	*Bubalus quarlesi*	1996	1965–2014	Indonesia	37–62(?)	Y	Y	EN	EN	EN	EN	EN
Mountain Gorilla	*Gorilla b. beringei*	1997	1965–2018	Uganda	68(44)	Y	N	EN	CR	CR	CR	CR
Iberian Lynx	*Lynx pardinus*	1998	1965–2015	Spain	52(32)	Y	Y	EN	EN	CR	CR	CR
Muriqui	*Brachyteles arachnoides*	1998	1982–2016	Brazil	27(21)	Y	Y	EN	EN	EN	EN	EN
Goodfellow's Tree-kangaroo	*Dendrolagus goodfellowi*	1998	1982–2016	PNG	47(35)	Y	Y	VU	EN	EN	EN	EN
Doria's Tree-kangaroo	*Dendrolagus dorianus*	1998	1982–2016	PNG	47(35)	Y	Y	VU	VU	VU	VU	VU
Humboldt Penguin	*Spheniscus humboldti*	1998	1988–2018	Chile	31(23)	Y	Y	LR/NT	VU	VU	VU	VU
Red Wolf	*Canis rufus*	1999	1982–2018	USA	43(25)	Y	Y	EN	CR	CR	CR	CR
African Penguin	*Spheniscus demersus*	1999	1988–2018	S. Africa	35(18)	Y	Y	LR/NT	LR/NT	VU	VU	EN
Ethiopian Wolf	*Canis simensis*	1999	1986–2011	Ethiopia	68(44)	N	N	EN	CR	EN	EN	EN
Arabian Tahr	*Arabitragus jayakari*	2000	1965–2018	UAE	50(29)	Y	Y	VU	EN	EN	EN	EN
Riverine Rabbit	*Bunolagus monticularis*	2000	1986–2016	S. Africa	21(17)	Y	Y	EN	EN	CR	CR	CR
Magellanic Penguin	*Spheniscus magellanicus*	2000	1988 2016	Chile	43(35)	N	N	LC	LR/NT	NT	NT	NT
Galapagos Penguin	*Spheniscus mendiculus*	2000	1988–2016	Chile	43(35)	N	N	VU	EN	EN	EN	EN
Giant Jumping Rat	*Hypogeomys antimena*	2001	1994–2016	Madagascar	14(10)	Y	Y	EN	EN	EN	EN	EN
Blue Swallow	*Hirundo atrocaerulea*	2002	1988–2016	S. Africa	25(20)	N	N	VU	VU	VU	VU	VU
Horned Guan	*Oreophasis derbianu*	2002	1988–2016	Mexico	38(26)	Y	Y	EN	EN	EN	EN	EN
Malayan Tapir	*Tapirus indicus*	2003	1986–2014	Malaysia	32(14)	Y	Y	VU	EN	EN	EN	
Harpy Eagle	*Harpia harpyja*	2003	1988–2016	Mexico	?	Y	Y	LR/NT	NT	NT	NT	
Mountain Tapir	*Tapirus pinchaque*	2004	1965–2014	Colombia	66(48)	Y	Y	EN	EN	EN	EN	
Maned Wolf	*Chrysocyon brachyurus*	2005	1965–2015	Brazil	51(47)	Y	Y	LR/NT	NT	NT	NT	
Okinawa Rail	*Gallirallus okinawae*	2006	1988–2016	Japan	62–90 (?)	Y	Y	EN	EN	EN	EN	
Lowland Tapir	*Tapirus terrestris*	2007	1986–2018	Brazil	74(64)	Y	Y	VU	VU	VU	VU	
Rio Grande Silvery Minnow	*Hybognathus amarus*	2007	1990–2018	USA	41(22)	Y	N	EN	EN	EN	EN	
Mangrove Finch	*Geospiza heliobates*	2008	1988–2018	Ecuador	18(10)	Y	Y	CR	CR	CR	CR	

was increased by one RL category between years −5 and 0. From this simulated year 0 dataset (i.e. workshop year), we generated a year 5 dataset of threat status, by applying a new trend sampled from D_{trend}. The same operation was repeated to generate simulated datasets for years 10 and 15. Species reaching the maximum category of 5 (Extinct) were removed from the sampling pool. (*v*) The entire process was repeated 50,000 times to create variation in the propagation of the extinction trend past $t = 0$. For every simulated dataset RLI values were calculated for each time-step, to produce distributions of simulated $RLI_{t,sim}$ for the 35 species. Finally (*vi*), to assess the likelihood that the observed results arose by chance, we compared $RLI_{t,obs}$ (estimated from the observed data) to the simulated distribution of $RLI_{t,sim}$ at each time point. The observed data was considered significantly different from the simulated data when the observed RLI ($RLI_{t,obs}$) was larger than the 95th quantile of the simulated distribution ($RLI_{t,sim}$). *P*-values were calculated as the proportion of $RLI_{t,sim}$ values that were greater than $RLI_{t,obs}$.

Using the simulated dataset, (*vii*) we also calculated the proportion of species that reached Category 5 (Extinct) after 15 years, to provide an estimate of average extinction risk without planning. Steps (*i*) to (*vii*) were repeated between years −5 and 10 for the complete dataset (46 species).

All analyses were performed using the R software (R Development Core Team, v. 3.2.0).

3. Results

Compared to extinction risk at the time of the workshop, two of the 46 species with data for up to 10 years post-planning had improved in status after 10 years, 10 had declined and 34 were stable (with some of the latter declining initially before returning to their previous status). After 15 years, three of the 35 species with data for up to 15 years post-planning had improved in status, nine had declined and 23 were stable (Table 1). Before the planning workshop, mean status for these 35 species was between Vulnerable and Endangered. Afterwards, the mean extinction risk continued to increase until 10 years post-planning, after which it decreased, leaving mean status between Endangered and Critically Endangered by year 15 (Fig. 2). No species went extinct in the timeframe. One species temporarily classified as Extinct in the Wild (by year 10) underwent revision to Critically Endangered following reintroduction (Fig. 2).

For both the datasets ($n = 35$ and $n = 46$), there was no significant difference between simulated and observed RLIs for time-steps −5 and 0 (Tables 2a, 2b, Fig. 3, S1a, b), indicating the bootstrap procedure was unbiased and the overall pre-workshop trend was not driven by a few species with unusual trajectories, validating the pre- versus post-planning comparison.

For the species with data up to 15 years post-planning (n = 35), observed RLI values post-planning were consistently higher than the simulated means (without planning) and increasingly so as time after planning increased (Fig. 3, Table 2a). By years 10 and 15 the difference was statistically significant (p-values <0.04 and <0.001 respectively), signifying a post-planning improvement in overall extinction trend unlikely to have arisen by chance. An increase in sample size (to n = 46) strengthened the effect (p-values at 10 years are <0.04 and <0.01, for n = 35 and n = 46 respectively), as did increasing the number of years (p-values at 10 and 15 years for n = 35 are <0.04 and <0.001 respectively).

Table 2a
Observed and simulated Red List Index (RLI) values, 95th quantile and associated *p*-value comparing observed and simulated RLI for 35 species (the number of projects with 15-years data post-planning workshop). Time steps began 5 years before the planning workshop was held and extended to 15 years after.

Time step	RLI observed	RLI simulated	95th quantile	*p*-Value
−5	0.48	0.48 ± 0.03	0.52	0.52
0	0.43	0.43 ± 0.02	0.45	0.57
5	0.41	0.38 ± 0.03	0.41	0.11
10	0.37	0.32 ± 0.04	0.39	0.04
15	0.39	0.26 ± 0.04	0.34	0.00018

Table 2b
Observed and simulated Red List Index (RLI) values, 95th quantile and associated *p*-value comparing observed and simulated RLI for 46 species (the number of projects with 10-years data post-planning workshop). Time steps began 5 years before the planning workshop and extended to 10 years after.

Time step	RLI observed	RLI simulated	95th quantile	*p*-Value
−5	0.49	0.49 ± 0.03	0.49	0.52
0	0.44	0.44 ± 0.02	0.42	0.57
5	0.43	0.39 ± 0.03	0.42	0.06
10	0.37	0.35 ± 0.04	0.39	0.01

Without planning, over the 15-year timeframe following planning, the simulated trajectory predicted the extinction of 7.8 ± 2.5 species (15–29%) of the 35 considered (Year 15 RLI = 0.274).

4. Discussion

This is the first systematic study to demonstrate the benefits to species' conservation status resulting from an integrative, multi-stakeholder planning approach employed by the IUCN SSC. Systematic reviews of other approaches have drawn conflicting conclusions about the impact of planning, and the purpose of this study was to improve the information available to decision-makers charged with determining whether and how to invest in planning the conservation of threatened taxa. In this study, we measured the response of a group of species to this planning approach, to assess whether species' conservation prospects were better after planning than before it. Our results show that post-

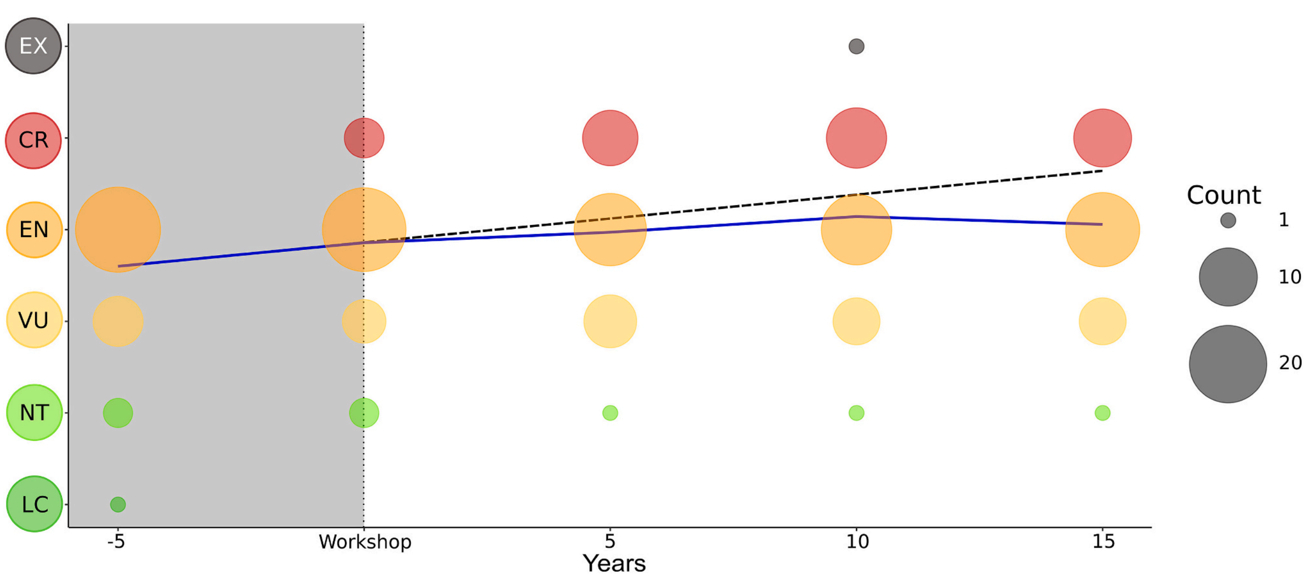

Fig. 2. Relative allocation of 35 species to extinction risk categories at 5-year intervals, beginning 5 years before the planning workshop and continuing to 15 years afterwards. Extinction risk categories are: Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR) and Extinct (EX). By weighting these such that LC = 0 and EX = 5, we calculated the “mean extinction category” (solid line) and compared this to the extrapolated pre-planning trend (dotted line).

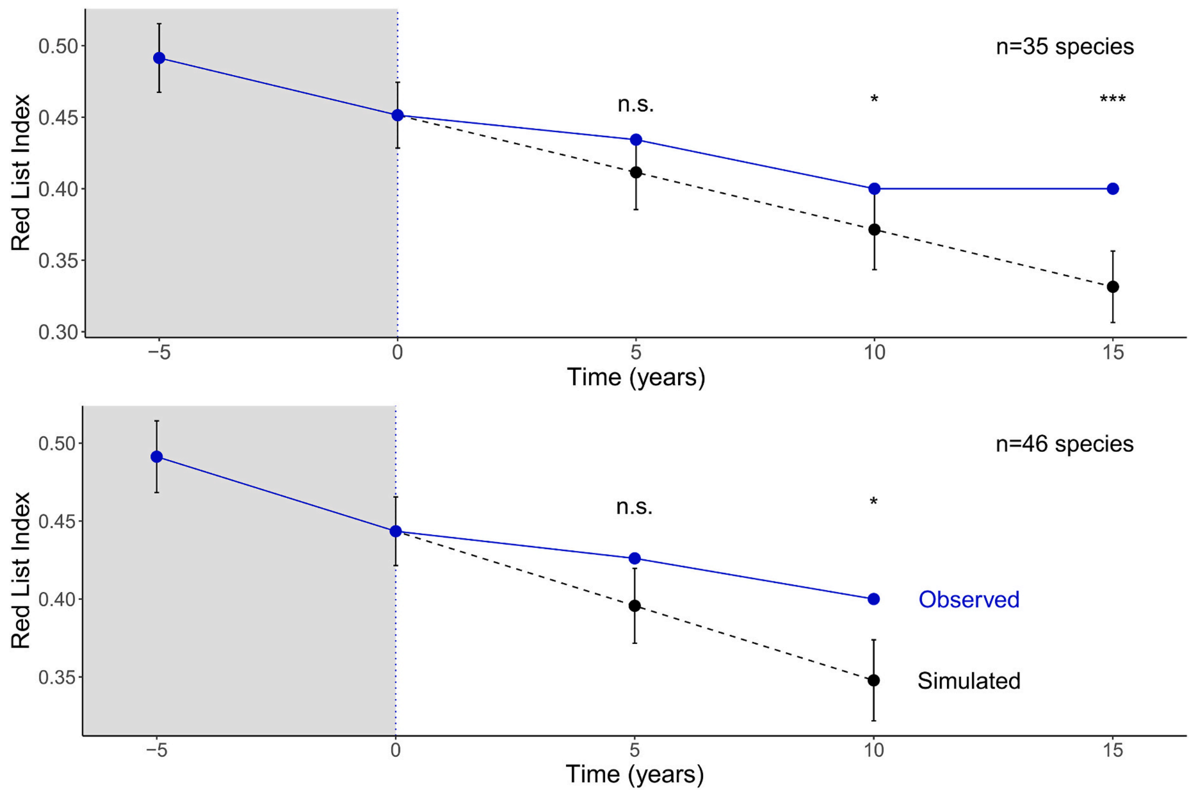

Fig. 3. Aggregate extinction trends for species before planning and for up to 10 (n = 46) and 15 (n = 35) years after planning (solid line). Pre-planning trends projected to 10 and 15 years afterwards (dotted line). Error bars depict standard deviations for simulated Red List Indices. **p*-value <0.05; ***p*-value <0.01; ****p*-value <0.001; n.s. non-significant.

planning, the aggregate rate of decline to extinction was slowed significantly by year 10 and reversed by year 15. Meanwhile, our simulated counterfactual scenario (projecting the expected declines without planning), predicted the extinction of 7.8 (±2.5) species over the same timeframe, in stark contrast to the zero extinctions observed with planning. However, because declines continued for a period after planning, only 3 (8.6%) of 35 species had improved in status by year 15, while 9 (25.7%) had declined and 23 (65.7%) had remained stable.

Before assuming the observed turning point to be the result of planning, it was necessary to review and eliminate rival alternatives (Ferraro, 2009). We discounted the possibility that changes in threat status of these species simply reflected global trends for similar taxa over the relevant period as during this time (1990–2018), declines are reported in the overall RLIs for mammals (33 of the species examined here), birds (nine species), and amphibians (one species) (WWF, 2020).

Next, we considered whether the turning point had been created or exaggerated by our project selection criteria. In relying on the availability of RL assessments pre-, post-, and at the time of planning, project selection may have been biased towards better-studied and potentially more recoverable species. However, of the 96 species-level projects available to us in the database, only six species (3%) were excluded because they had no Red List data. The 46 species that met the criteria for our study underwent a mean of 7.54 assessments (S.D. = 1.83) between 1986 and 2018 and the remaining 45 that did not, were assessed similarly often (mean = 6.09; S.D. = 3.23) with the timing of assessments relative to workshops the primary cause of exclusion. We conclude this eliminates project selection bias as the cause of the observed results.

We also considered whether the observed turning point could have been the result of measures set in place before the workshops, the results of which were only observed and recorded in the years after them; that is, that the workshops were a symptom of improved conservation efforts and not a cause. We ruled this out on the basis that workshop reports described the prevailing circumstances as those in which conservation efforts had stalled or were frustrated by, for example: conflicting views among stakeholders (e.g. projects 20, 28, 45, Table S1); uncertainty about how to proceed with conservation action (e.g. projects 15, 29, 46); or limited coordination or connectivity among implementers (e.g. projects 13, 24, 33).

Finally, we considered whether planning could have coincided with other events that were the real trigger of the turning point, such as the beginning of conservation action for the species or a sudden injection of resources. However, workshop reports and Red List accounts indicate that in all cases, conservation activities such as legal protections for the species or its habitat, had begun years and often decades before the workshops (see Supplementary material Table S1). We found no evidence of sudden resource investment, with conservation in the countries considered reported to be chronically under-resourced at the time (James et al., 1999).

On balance, the information available supports the proposition that the post-planning outcomes observed in this study were triggered by the planning intervention itself, rather than by coincidental factors or project selection bias. It is important to stress that we do not suggest that planning caused the observed changes in species conservation status - these were the result of conservation action taken by multiple agencies and conservation donors, working over several decades, in many countries. Our proposition is that this planning approach created a turning point in conservation efforts for these species that led to an overall improvement in outcomes in the years that followed.

The information gathered also allows insights into why this approach was beneficial for these species. The approach routinely integrates four elements that are not widely practised in combination: (*i*) population viability analyses (PVA); (*ii*) inclusion of both in situ and ex situ conservation expertise; (*iii*) facilitated participation of diverse stakeholders; and (*iv*) an emphasis on rapid production of outputs.

At the time of planning, most of the study species had experienced population declines or fragmentation (see Table S1). Population viability analyses were included in 91% of projects (see Table 1), using simulation models built with the program *Vortex* (Lacy and Pollak, 2021). These models supported not only investigation of the effects of deterministic threats to species, but also of the stochastic forces that can disproportionately influence the population dynamics of species with small or highly fragmented populations (Shaffer, 1981), leading to improved understanding and prioritisation of risks.

For species with elevated stochastic risks, mitigation of deterministic threats (such as habitat destruction and over-harvest) may not be sufficient to avert extinction (Foose et al., 1995). For the species in this study, the habitat and legal protections that were in place at the time for most species were clearly achieving only limited success (see Table S1). In such cases, urgent and intensive management at the level of populations and individuals, which explicitly targets demographic and genetic stochastic risks, may also be needed (Goodman, 1987; Foose et al., 1995; Frankham et al., 2017). These measures may involve in situ or ex situ activities, or a combination of both. Including the knowledge and know-how of both in situ and ex situ communities from the outset of planning can lead to better-integrated solutions, improving downstream results (Byers et al., 2013). Ex situ recommendations were included in plans for 87% of the study projects (see Table 1) and benefits accruing to several of these species as a direct result are reported elsewhere (e.g. CBSG, 2017; Young et al., 2014).

Though still relatively rare in species conservation planning, the inclusion of diverse stakeholders in planning decisions is widely advocated in environmental decision-making, premised on the understanding that science-based prescriptions alone will not improve outcomes (Pullin and Knight, 2004; Balmford and Cowling, 2006; Knight et al., 2008). The inclusion of stakeholders (e.g. those representing governments, agriculture, fisheries, academia, NGOs, local communities or the private sector) can confer multiple benefits, including lowered cost of enforcing regulations, benefits of local knowledge, increased project capacity and the sharing of responsibility (Forgie et al., 2001). However, within and between these sectors, differences in background, education, influences and agendas can lead to divergent views on whether or how action is taken, and interaction and dialogue can be key to resolving these differences (e.g. Cummins, 2004; Siebert et al., 2006; Brancalion et al., 2016; Maas et al., 2021). Success in this area is shown to increase where trust is secured (Young et al., 2016), conflicts are surfaced and managed (Madden and McQuinn, 2014) and those involved are united behind a clear and common purpose (Black, 2015). These outcomes can be advanced effectively in face-to-face workshops guided by third-party facilitation (Drolet and Morris, 2000; Mackelworth, 2012). The challenges that precipitated planning for the species in the study set included stakeholder conflicts and uncertainty, and limited coordination among implementers (see Table S1). In these circumstances a facilitated multi-stakeholder approach conferring the benefits described above, should improve outcomes. All projects adopted this method, with an average participation of 46 individuals (range 14–93) and 31 institutions (range 10–64) per project (see Table 1). We do not have specific information on how effective this approach was in all the study projects. However, earlier evaluations of some of the same projects concluded the participatory approach of the workshops was effective in fostering collaboration (CBSG, 2017; Vredenburg and Westley, 2003). In particular, surveyed stakeholders reported improved clarity of goals, and uniting of disparate groups over the short-term, promoting increased collaboration on action and research, improved understanding of other stakeholders' viewpoints and greater support for on-ground action, over the longer term. We assume that similar benefits were experienced by the study projects and that this contributed to the observed result.

Lastly, criticism has been levelled at plans that take years to produce (Tear et al., 1995) potentially creating a hiatus in decision-making, permitting or in undertaking key activities. For threatened species with urgent needs, such delays can facilitate further declines and exacerbate the difficulty of recovery (Martin et al., 2012; Hutchings, 2015). In the approach studied here, planning participants committed in each case to documenting the agreed plan swiftly (within 6–12 months), with the accompanying aim of minimising the inevitable trade-offs between speed, and quality or completeness, by siting plans within an iterative cycle of regular review and adaptation (Salafsky et al., 2002). It is assumed that this contributed, at least in part, to the post-workshop momentum described in CBSG (2017). In short, this planning approach was effective because it brought analytical tools well-tailored to the conservation needs of the species targeted, and a participatory decision-making environment that supported those involved to transition swiftly to more effective ways of working together.

To date, this approach, along with species-based conservation planning in general, has been applied mainly to vertebrates, and among those, to larger-bodied, higher-profile and more charismatic species, reflecting a well-recognised human bias in the value (and therefore the resources) apportioned to different taxa (Tear et al., 1995; Metrick and Weitzman, 1996; Laycock et al., 2009; Watson et al., 2011; Brambilla et al., 2013; Drinan et al., 2020). The planning principles and tools involved could benefit a wider range of threatened species but resources and time are obstacles to broader application. There are currently 37,480 taxa classified as threatened (IUCN, 2021) and though many of these may respond sufficiently well to general conservation measures targeted at area protection and threat mitigation, thousands may not. Planning for these species individually will be both too costly and too slow. Applying planning approaches such as this to well-chosen multi-species groups may be part of the solution.

Multi-species planning is not a new idea, though its application to date has received mixed reviews (e.g. Clark and Harvey, 2002; Moore and Wooller, 2004; Cullen et al., 2005; Baptista et al., 2019). Nevertheless, successful outcomes should be achievable with careful attention to the design of the planning approach and to the method of grouping species. Productive groupings for planning are expected to be among species that: share similar threats within a defined geographic or political area; rely on the same (threatened) ecosystem, habitat, or micro-habitat; are otherwise similarly affected by the same primary threats; share a need for intensive management either in situ or ex situ; or have needs that coincide closely with those of a higher-profile "umbrella species" (Burbidge, 1996; Machado, 2005; Foin et al., 1998; Jewell, 2000; Clark and Harvey, 2002; Branton and Richardson, 2011; Ward et al., 2019). The approach to planning described in this study, with some modification, has recently been trialled with multi-species groups of taxa including freshwater fish, reptiles, insects and trees (e.g. Gibson et al., 2020; Lees et al., 2020) and we recommend further application and evaluation, covering a broader array of taxa.

Studies have shown that time to recovery varies between species depending on biology and circumstances, with recovery particularly challenging for long-lived species, species with small and fragmented populations and species with particularly intractable threats (Abbitt & Abbitt and Scott, 2001; Cardillo et al., 2005; Davidson et al., 2009; Hutchings, 2015). Our study set was dominated by larger-bodied, longer-lived taxa with small or fragmented populations and some of the most difficult conservation challenges, including competition with people for habitat and food, and unsustainable harvesting (Ceballos and Ehrlich, 2002; Sutherland, 2001; Bennett, 2015; Table S1). Further, obstacles linked to social, institutional, and organisational factors that can delay effective action (Ortega-Argueta, 2020) were frequently reported (Table S1). As a result, we would have predicted the species in this study to show longer recovery times. Nevertheless, we chose this planning approach partly because of the long period over which it has been used (>30 years), which we hoped would provide enough time for plans to have been implemented, species to have responded to interventions, and for changes in status to have been measured and reported. However, available data provided only 10–15 years of post-planning information in most cases. Though by year 15 we were able to show an overall upward trajectory in species prospects, many taxa had not regained their pre-workshop conservation status in that time and a longer evaluation period is needed to confirm outcomes. This illustrates again one of the difficulties of evaluating planning impact.

In general, given that the species we examine do represent some of the most challenging for conservation, the timeframe to positive results shown in this study may sit at one extreme of the possible range. Though other studies also report times to recovery signals in excess of a decade (e.g. Beck et al., 1994; Schultz and Gerber, 2002; Young et al., 2014) this may again reflect a general bias in the conservation attention assigned to particular taxa. Such long timeframes present a challenge for nations aspiring to measurably improve the status of threatened species within a decade, in line with CBD commitments. However, in the previous section we recommended expanding this style of planning beyond the usual targets, to currently neglected species of animals and plants. Many of these are smaller-bodied, with shorter generation lengths, larger population sizes and consequently shorter potential recovery times. An expected added benefit then, of this expanded effort, would be an overall increase in the rate of species recovery.

One of the reasons that the RLI is such a valuable metric is that it is based on RL categories which are designed to be comparable across taxa. However, because of the need for broad applicability, a considerable change in species' prospects is required to trigger a shift in category (and therefore in RLI), such that hard-won improvements (or declines) are masked within shorter timeframes. The IUCN's new Green Status assessment, which scores the recovery status of species at finer scales and accounts for recovery potential (Akçakaya et al., 2018), could be a valuable additional metric for use in future evaluation. We recommend that government and non-government agencies responsible for generating large numbers of threatened species plans use the Red List Index as a primary aggregate measure for evaluating planning impact and consider combining it with the Green Status assessment once this metric is more widely available.

In summary, this study demonstrates the benefits of a science-based, participatory planning approach to a group of species facing multiple threats, and where conflicting views, uncertainty, or lack of coordination among stakeholders constrained action. These circumstances are common to many threatened taxa for which planning is needed but not currently resourced. Given the results described here, we recommend extending the use of this approach to more of the taxa that could benefit. In addition, as an efficient way to extend its contribution beyond the usual targets (longer-lived, charismatic mammal and bird species) we recommend evaluating its application to carefully selected multi-species groups featuring some of the more neglected (though sometimes more easily recovered) animal and plant taxa.

Though we stress that it is conservation action on the ground that generates good outcomes for species, our study provides evidence that with the right approach, species conservation planning can provide a turning point in species conservation efforts, supporting those involved to transition quickly to more effective ways of working together.

CRediT authorship contribution statement

Caroline Lees: Conceptualisation, Methodology, Investigation, Writing – Original draft, review & editing.

Alexis Rutschmann: Formal analysis, Visualisation, Review & editing.

Anna Santure: Conceptualisation, Supervision, Review & editing.

Jacqueline Beggs: Conceptualisation, Supervision, Review & editing.

All authors have approved this description of the partitioning of work.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

CML works for CPSG, AR has worked on CPSG projects; neither were involved in the study projects used to build the dataset. The authors were not involved in any of the IUCN Red List assessments used in the study. Funding was provided by the University of Auckland and CPSG.

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Appendix A. Supplementary data

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