Assessing the risks of changing ongoing management of endangered species

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Abstract

Recovery programmes for endangered species can become increasingly demanding over time, but managers may be reluctant to change ongoing actions that are believed to be assisting recovery. We used a quantitative risk assessment to choose support strategies for a reintroduced population of Mauritius olive white-eyes *Zosterops chloronothos*. Facing increasing costs, managers considered changing the ongoing supplementary feeding strategy, but at the same time worried this could jeopardize the observed positive population trend. We used a feeding experiment to compare the current feeding regime and a cheaper alternative (a simple sugar/water mix). Results suggested the cheaper alternative would only marginally reduce population vital rates. We assessed the influence of these results and the associated uncertainty on population recovery and management costs using two decision-analytic criteria, incremental cost-effectiveness ratio and stochastic dominance. The new feeding regime was expected to be, on average, more cost-effective than the status quo. Moreover, even negative outcomes would only likely mean a slower-growing population, not a declining one, whereas not changing feeding regime actually entailed greater risk. Because shifting from the current regime to a cheaper sugar/water mixture was both a risk-averse and a cost-effective choice, we decided to implement this change. Four years after the experiment, the population continues to grow and costs have been contained, matching predictions almost exactly. In this case, the field experiment provided useful empirical information about prospective actions; the risk analysis then helped us understand the real implications of changing the feeding regime. We encourage managers of recovery plans facing a similar situation to explicitly recognize trade-offs and risk aversion, and address them by combining targeted research and formal decision analysis.

Introduction

Many populations of threatened species require some ongoing support to improve their chances of persistence (Jones & Merton, 2012). For vertebrate species, support can include the provision of appropriate breeding and resting sites (Norris & Mcculloch, 2003), control of predators (Jones *et al.*, 2016) and parasites (Hudson *et al.*, 2016), and supplementary feeding (Ewen *et al.*, 2014). The decision of which population support strategy to apply is typically complicated by uncertainty about what is limiting population growth. Ideally, a choice should be based on *a priori* hypotheses of limiting factors; for example, a hypothesized lack of natural food resources may lead to a decision to supplement food (Ewen *et al.*, 2014). Alternatively, support may be provided within a broad range of actions targeting general regulating factors of populations (Jones & Merton, 2012). In this case, the hope is for populations to establish and grow without needing to know exactly what actions caused this. Once populations are secured then management components can be assessed and those having little influence on population recovery can be removed. This second scenario appears common in extreme rescue operations of critically endangered species (Jones, 2004, Jones & Merton, 2012).

When the time for change comes, however, managers may be reluctant to abandon support measures known or perceived to be successful, even when faced with the increasing long-term costs of providing intensive support (Goble *et al.*, 2012). Uncertainty about the outcomes of changing actions can create a situation where change might provide benefits, such as reducing costs, but also lead to negative results, such as an unwanted population decline. Managers can try to reduce uncertainty via experiments or monitoring (Armstrong, Castro & Griffiths, 2007), then make a decision accounting for the remaining uncertainty using suitable decision-support methods (e.g. Canessa *et al.*, 2016a).

We present a practical example of such a combined approach in deciding on a supplementary feeding strategy for the Mauritius olive white-eye *Zosterops chloronothos* (hereafter olive white-

68 eye). Supplementary feeding was provided as part of a management package to aid the species'

69 establishment following reintroduction to the 26-Ha Ile aux Aigrettes island off the south-east coast

70 of mainland Mauritius. The need for a decision about continued feeding was made necessary by the

71 cost and labour required to provide supplementation. However, any reduction in management effort

72 required careful evaluation, given the current feeding regime had likely played a role in the success

73 during the establishment phase of this offshore island population. We carried out a feeding

74 experiment to estimate the consequences of keeping or changing the current regime, quantifying

75 uncertainty. We then used two decision-analytic criteria to identify the best action in the face of

76 uncertainty and risk aversion.

77 **Materials and methods**

78 *Study species*

79 Olive white-eyes are a critically endangered and endemic passerine species of Mauritius (IUCN,

80 2014). They are a socially monogamous species where pairs establish and defend territories year

81 round. Once presumed to be widespread across Mauritius, the species has suffered continuing range

82 contraction and a crash in population size, with an estimated ~100 pairs in 2001 (Nichols, Woolaver

83 & Jones, 2004) currently projected to decline by 14% every year, mostly due to rat predation

84 (Maggs *et al.*, 2015). In response to this threat, a translocation was undertaken to Ile-aux-Aigrettes,

85 an offshore, low-lying coralline limestone island with an endemic and regenerating coastal forest.

86 Feral cats and black rats (*Rattus rattus*) were eradicated from Ile-aux-Aigrettes in 1991. A founding

87 cohort of 38 individual olive white-eyes were released between 2006 and 2010. To improve chances

88 of establishment, the population was provided supplementary feeding, which continued post-

89 release. Presumably aided by this food provision, the population continues to grow. At the

90 beginning of our study in September 2013 there were 11 known breeding pairs in the population,

91 with 11 feeders available; the most recent population count, in June 2019, recorded a minimum of

92 55 adult birds (S. Henshaw, Mauritian Wildlife Foundation, *pers. comm.*).

In addition to regular monitoring activities, supplementary feeding stations are established within each breeding pair's territory or in new sites that are subsequently occupied by new breeding pairs (additional details in Maggs *et al.*, 2019). At the beginning of our study, a complete diet with three types of supplements was provided at each station: (i) Aves© Nectar, a commercially available water-soluble powder deemed to provide a full dietary supplement to nectar-feeding birds, (ii) fresh fruit (grapes), and (iii) an insectivorous mix (Insectivorous feast®, Birdcare Company) mixed with egg, carrot and apple. Note that birds still consume natural food when available, particularly outside energy-demanding breeding phases (Maggs *et al.*, 2019). Feeding was done twice-daily (morning and midday) at all stations. The morning routine consisted of providing all the types of supplementary food mentioned above and the midday routine consisted of changing the nectar. Twice-daily feeding was necessary because Aves© Nectar fermented quickly in the hot field environment. The management effort required to sustain such intensive management was considerable. On the other hand, only one breeding pair is known to have successfully bred and fledged a chick without use of food from a feeding station, suggesting that current feeding is at least partly contributing to population growth. Because supplementary feeding has been provided since reintroduction, we have no additional evidence (such as a control group not supplementary fed) to understand how critical it is to the observed population growth.

Therefore, we identified two fundamental and competing management objectives: maximizing population growth, measured as total population size *N*, and minimizing management costs, measured in MUR Rupees and including the cost of food (per feeding station per month) and of staff time required (staff salary for hours worked per month). This was a short-term management decision, so we evaluated both objectives over a four-year time frame. We considered two alternative actions, either maintaining the current feeding regime (status quo), or replacing Aves© Nectar with cheaper sugar water that requires less frequent changes (sugar water). We initially used formal expert elicitation (Martin *et al.*, 2012) to screen additional alternatives, such as stopping the

feeding altogether or modifying it to more closely match demand (Maggs *et al.*, 2019). Such actions

might become viable in the future, if habitat restoration succeeds in reducing the species' reliance

on supplementary feeding. However, based on the information available at the time of our study,

those actions were considered to pose excessive risks to the population without sufficient benefit.

Only the two alternatives of status quo and sugar water were therefore chosen for the experimental

trial (see next section). To inform our decision, we then needed to predict the expected result of

either alternative action against both management objectives.

Feeding experiment

We obtained estimates of expected survival and fecundity under the two alternative feeding

strategies by carrying out a reciprocal feeding experiment over two years (2013/2014 & 2014/2015

September-August annual cycles), where the two feeding regimes were compared as experimental

treatments. Immediately prior to assigning a reciprocal feeding experiment design, all pairs were

given both Aves© Nectar and sugar water for one month and the consumption for each type was

measured twice weekly to test for any preference (significant preference toward the consumption of

sugar water compared to Aves© Nectar; $F_{1,333}= 3.81$, $P<0.0001$). If there had been a complete

refusal to consume sugar water, the proposed experiment would have been reassessed. Following

this preference test, all breeding pairs were assigned to one of two treatment groups for the duration

of the study (N=11 pairs; 6 pairs in group 1 and 5 pairs in group 2). In the 2013/2014 year, group 1

pairs were fed Aves© Nectar, and then switched to sugar water in 2014/2015. Group 2 pairs were

fed the reciprocal, with sugar water provided in 2013/2014 and Aves© Nectar provided in

2014/2015. Switching diets in different years provided a crossed design that allowed us to control

for background environmental stochasticity, while reducing the proportion of the population that

was exposed to a potentially risky change in management regime in a given year. Pairs in the two

groups were situated approximately on opposite sides of the island to further reduce the likelihood

of neighbouring pairs feeding on diets to which they were not assigned (similar to protocols

6

143   previously used to study supplementary feeding of threatened birds on small islands, e.g. Ewen *et*

144   *al.*, 2009). We assumed this small spatial bias would be compensated by the crossed study design

145   and would not influence our results because the site is a small and uniformly low-lying coralline

146   island with a restoring coastal plant community. Whilst Aves© Nectar continued to be changed

147   twice daily, the provision of sugar water allowed for once-daily changes (as sugar water is more

148   heat stable). All other aspects of the feeding remained unchanged (as described in the *Study species*

149   section). The total daily quantity provided under the two options was the same.

150   We estimated fecundity from the recorded number of fledglings produced by each female in each

151   season, and survival using monthly re-sighting records for a total of 24 survey occasions. The

152   survival data were a subset of the standard resighting records collected during the daily feeder

153   loading activity, carried out along a standardized route. Fecundity data were analysed using mixed-

154   effect Poisson regression, with female ID used as the random effect (to account for

155   pseudoreplication, as each female contributed data points to both food types and at different ages)

156   and parameters for supplementary food type, female age and female age squared to account for

157   possible senescence effects. Survival data were analysed using a Cormack-Jolly-Seber model with

158   two age classes: new individuals entered the population as juveniles in the month they fledged and

159   remained in this age class for four months before transitioning to the adult class. We used model

160   comparison to select the most supported model structures to be retained for the next step (Appendix

161   S1). For recruitment, generalized mixed linear models including the squared age term were

162   considerably more supported (ΔAICc>10 for models not including it). For survival, analysis in

163   program MARK (Cooch & White, 1999) suggested models including age-specific survival and

164   resighting probability were most supported (within ΔQAICc<1). Although for survival the model

165   including a treatment effect received limited support (ΔQAICc<6), we decided to retain this term

166   and assess its magnitude in the following analysis.

167 To fully quantify uncertainty about our estimated parameters, we then implemented the selected

168 models structures in JAGS (Plummer, 2005), running 100,000 iterations with a 50,000 burn-in and

169 a thinning rate of 10 (Appendix S2). We used uninformative priors for all parameters. We modelled

170 resighting probability as a uniform distribution, using an uninformative prior with range 0-1. We

171 retained the full posterior distributions of all estimated parameters (age-specific fecundity and

172 survival under the two treatments) to account for uncertainty in the next step.

173 *Demographic and cost predictions*

174 We used the results of our feeding experiment to parameterise an age-structured population matrix

175 with ten classes. Because the observed and estimated survival was high, but the maximum lifespan

176 recorded for the species with or without feeding is 11 years (Maggs, 2017; C. Ferrière, *pers. obs.*),

177 we assumed birds that reached this age would rapidly senesce (their survival decreased to 0.01 and

178 contribution to reproduction was negligible; see Results). We then projected population size over

179 four years, accounting for demographic stochasticity in survival and fecundity using binomial and

180 Poisson processes respectively. We started the simulations from 14 adult females, the number

181 present at the end of our experiment in August 2015. We propagated parametric uncertainty by

182 repeating the simulation 10,000 times: for each iteration, we generated a new matrix by randomly

183 drawing a value from the posterior distribution of each parameter. We modelled females only, then

184 estimated total population size assuming equal sex ratio (Maggs, 2017) for comparison with

185 monitoring data that usually include undetermined individuals.

186 We calculated costs of current staffing requirements and on budget projections prepared by the

187 Mauritian Wildlife Foundation's Fauna Manager and the olive white-eye project coordinator (CF

188 and NZ). The food costs of status quo management were known; we projected the costs of sugar

189 water based on a shelf price of 30 Rupees per kg of sugar and on mean monthly costs per feeding

190 station generated in the feeding experiment described below. We simply then replaced the Aves©

191 Nectar cost with sugar water cost and held all other supplementary food components the same

192   (grapes and insectivore mix laced with egg, carrot and apple). Staff salary cost was calculated for

193   one paid field staff at 50 Rupees per hour (this person normally assisted by one unpaid volunteer at

194   no cost and supervised by one coordinator with additional responsibilities not included in costing

195   here). We used current staffing requirements from the Mauritian Wildlife Foundation to reflect the

196   status quo and timed how long it took to prepare food, complete servicing of feeding stations and

197   then clean up after feeding. We then summed both component costs (food and staff time) to

198   generate an overall cost estimate per breeding pair for each feeding action. We estimated the annual

199   costs per feeder as 853.8 Rp for nectar (780 Rp for staff, 73.8 for food, 853.8 total) and 466.8 Rp

200   for sugar water (435 Rp for staff, 31.8 for food, 466.8 total). At each time step in each simulation

201   iteration we multiplied this annual rate by the predicted adult population size, obtaining a

202   distribution of estimated total costs for each feeding alternative.

203   *Decision analysis*

204   The final step in our analysis was to compare the simulated outcomes for different management

205   actions, to determine whether shifting from the status quo would imply an excessive risk of

206   jeopardizing population recovery. We carried out this risk analysis using two methods.

207   In the first step, we used stochastic dominance (Levy, 1998) to assess the risk that changing feeding

208   would negatively impact species recovery. Stochastic dominance allows decision-makers to rank

209   available actions, depending on their preferences and considering the full range of uncertainty about

210   the expected outcomes of actions. For a conservation example and a more detailed explanation, see

211   Canessa *et al.* (2016b). The full probability distributions of results are used to calculate the

212   cumulative distribution functions for each action, which are then compared to determine whether

213   dominance exists. First order dominance means that an action is preferred to another regardless of

214   whether the decision maker is risk averse. The only assumption required is that "more is better" (in

215   our case, higher population sizes are preferred). It is best understood by plotting the curves of the

216   cumulative distributions for each action. If the curves do not cross, the action with the lower curve

217   has first-order dominance over the other; it is always the better choice and choosing it entails no

218   risk.

219   If the curves of the cumulative distribution functions cross, no action dominates the other at the first

220   order. One action will have a greater chance of positive outcomes, but also a greater chance of

221   negative ones (i.e. longer distribution tails on both sides). Making this choice therefore entails some

222   risk, and the attitude of the decision maker becomes relevant to the choice. For example, in our

223   olive-white eye example, the decision makers are risk averse, that is, they prefer to avoid negative

224   outcomes. Knowing this, one can verify whether second-order stochastic dominance exists. The

225   same visual comparison is repeated, this time plotting the integral of the cumulative distribution

226   function for each action. If the curves do not cross, the action with the lower curve has second-order

227   dominance over the other; this action is the better choice for a risk-averse decision maker. Note that

228   actions that dominate at the first order also dominate at the second order. For the olive white-eye

229   case, we calculated the CDFs and their integrals for the empirical distributions of population sizes

230   under each action, then assessed whether an action had stochastic dominance over the others at the

231   first or second order.

232   In the second step of risk analysis, we assessed the balance between effectiveness (population size)

233   and cost of changing feeding regimes by calculating the incremental cost-effectiveness ratio

234   (ICER), a standard metric in evaluation of healthcare options (Cohen & Reynolds, 2008). We

235   calculated the ratio as

236    $$ICER = \frac{Cost_{change} - Cost_{status\ quo}}{Effectiveness_{change} - Effectiveness_{status\ quo}}$$ Eq. 1

237   where *change* was the sugar water feeding, and *status quo* the current Aves© Nectar feeding

238   (values are the totals over the four-year management time frame). This ratio could be positive or

239 negative as the result of different combinations. For example, it could be positive if sugar water

240 proved cheaper but produced fewer animals than nectar, but also if it were both more effective and

241 more expensive (for example by resulting in a much greater total population, offsetting the lower

242 cost per feeder). Therefore, we plotted simulation results for both objectives in the space defined by

243 the differences in cost and effectiveness (Cohen & Reynolds, 2008). Within this four-quadrant

244 space, a cheaper and more effective action would always be preferred, a more expensive and less

245 effective action would always be discarded, and situations presenting trade-offs (cheaper but less

246 effective or more expensive but more effective) would be assessed using the ICER metric. To

247 account for uncertainty, we counted the proportion of simulation iterations that fell in each

248 quadrant.

249 **Results**

250 *Predictions of expected management outcomes*

251 Breeding female olive white-eyes produced between 0 and 6 fledglings per year. The model

252 estimated slightly higher fecundity when females were fed sugar water than when they were fed

253 Aves© Nectar, and a clear quadratic relationship between fecundity and female age for both

254 methods (Fig. 1a). Peak fecundity was expected at age 3 and 4 for sugar and nectar feeding

255 respectively (mean±s.d, Nectar: f=1.925±0.787, Sugar: f=2.937±1.249; Fig. 1a). Survival was

256 generally high (Fig. 1b), higher for adults than for juveniles, and higher for sugar-water-fed than

257 nectar-fed juveniles (mean±s.d. for juveniles Nectar ϕ=0.676±0.132, Sugar ϕ=0.801±0.114; for

258 adults Nectar: ϕ=0.990±0.013, Sugar: ϕ=0.969±0.031; Fig. 1b).

259 The simulation suggested sugar water was expected to result, on average, in larger population sizes

260 and cheaper costs than the status quo of Aves© Nectar (mean adults [min-max], Nectar: $N_{t=5}$=39

261 [24-228], Sugar: $N_{t=5}$=65 [24-390]; Fig. 2a-b). Only a small proportion of simulation runs resulted

262 in a population decline from the initial size (Fig. 2a). The cumulative distribution functions of the

11

263   two actions did not cross, suggesting the change to sugar water was always better than maintaining

264   the status quo, regardless of the risk attitude of managers (Fig. 2c-d). Moreover, in 87% of the

265   simulation runs, sugar water dominated the status quo in terms of cost-effectiveness, providing a

266   cheaper and more effective alternative; the opposite never occurred (Fig. 3). In the remaining

267   simulation runs, switching to sugar water generated a trade-off between cost and outcomes (Fig. 3).

268   The average cost-effectiveness ratio was -921 Rp/pair (~25 US$) over four years, where the

269   negative value reflects the dominance of sugar water on both objectives (extra pairs came cheaper

270   than if maintaining the status quo). For simulation runs where the predicted population size was the

271   same under the two feeding regimes, and therefore the ratio could not be calculated, sugar water

272   was on average ~800 US$ cheaper. Given these values, and the results of the stochastic dominance,

273   the decision makers chose to switch to sugar water in September 2015.
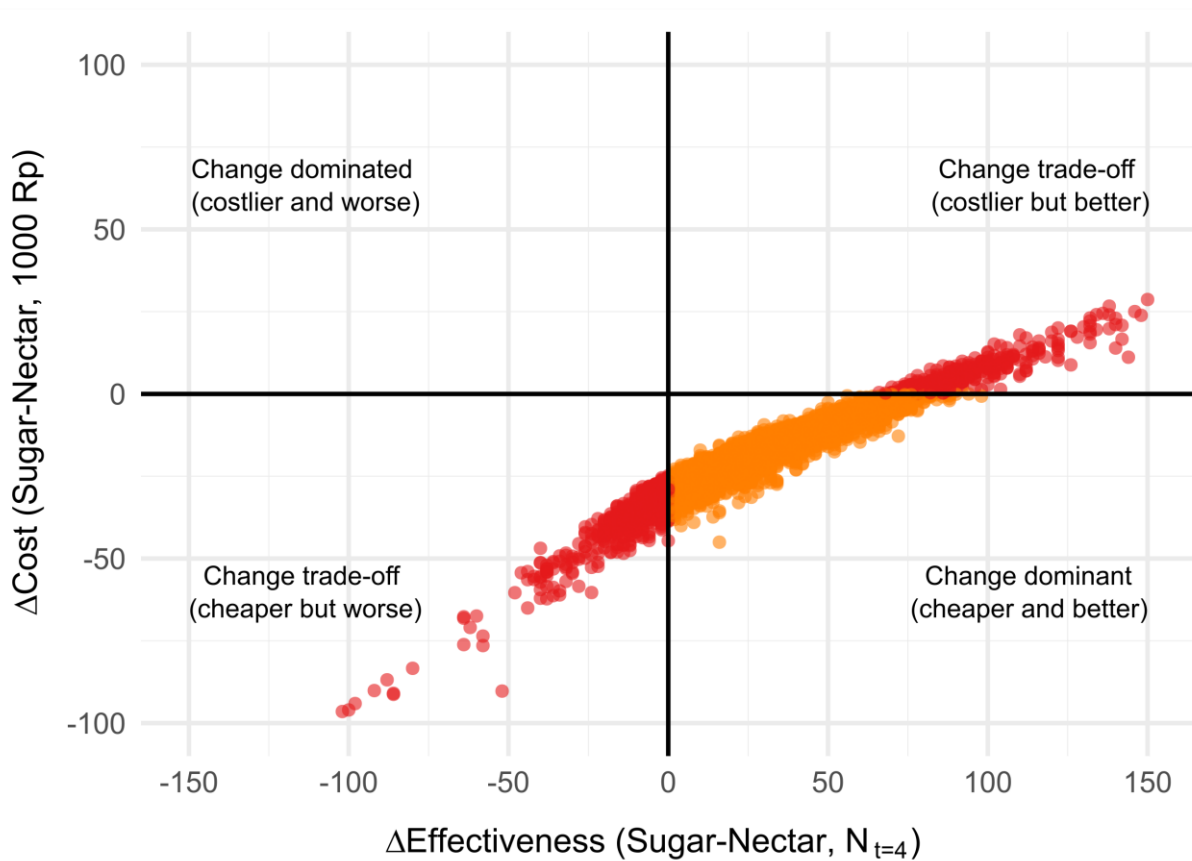
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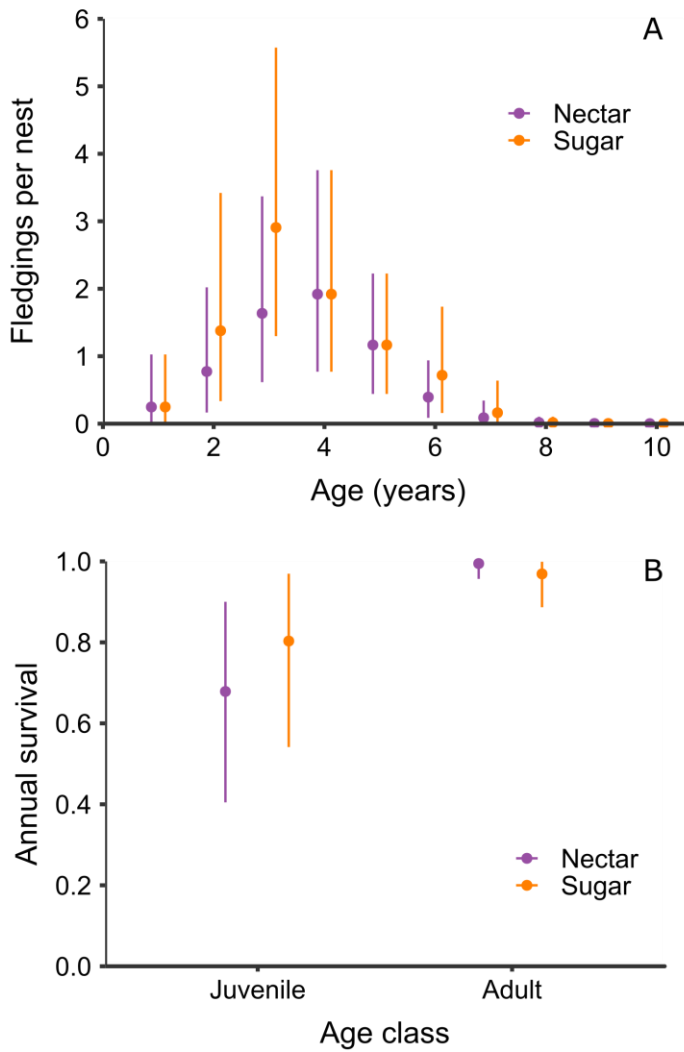
Figure 1. Annual recruitment (A) and survival (B) of olive white eyes as estimated from feeding experiments, under different feeding regimes (status quo of Aves© Nectar, orange; proposed change of sugar water, purple).
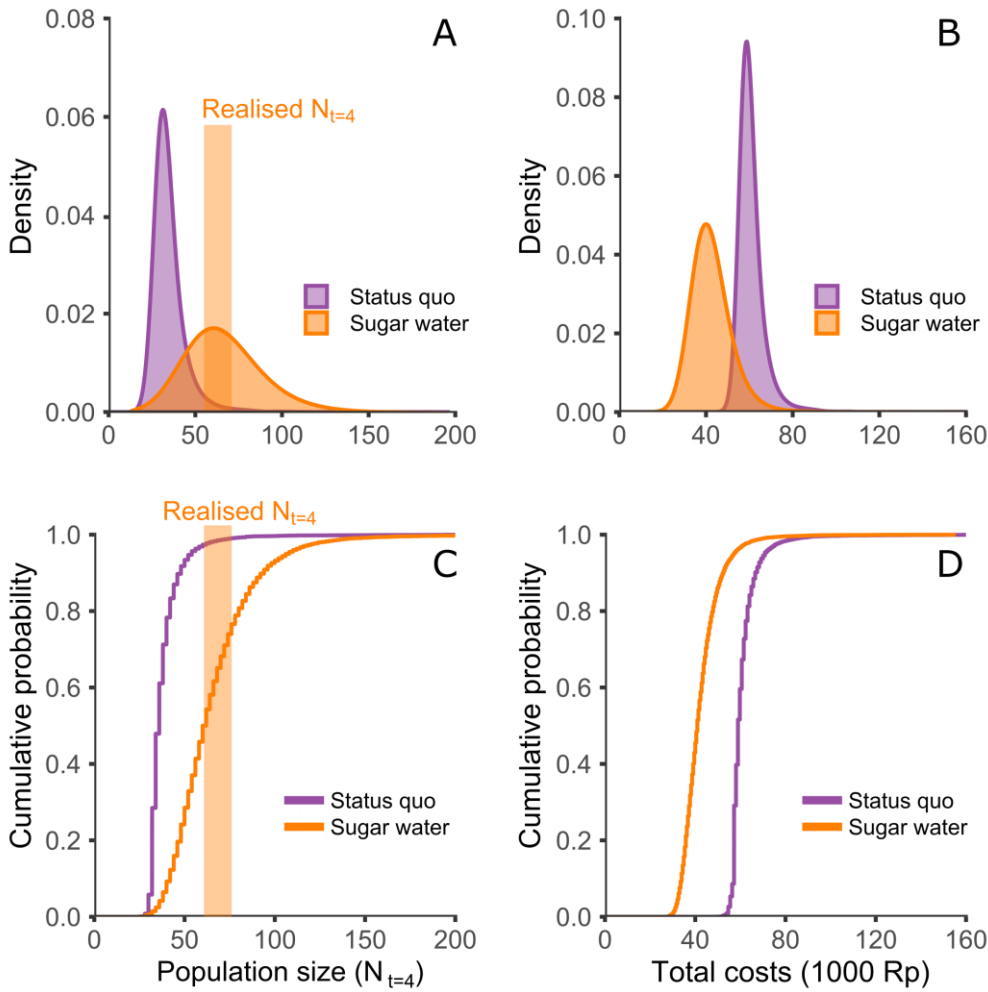
Figure 2. Stochastic dominance between alternative feeding regimes, with and without consideration of management costs. Top plots indicate the distribution across 10,000 simulation runs of predicted (A) adult population sizes and (B) total costs. In (A), the rectangle indicates the realised adult population size in June 2019, four years after the change to sugar water was implemented; the shaded area covers the range between the minimum and maximum population counts (respectively, known individuals alive and total count including unmarked animals). Bottom plots (C,D) indicate the cumulative distribution functions of the distributions in plots (A,B). Because the curves do not cross, sugar water has first-order stochastic dominance over status quo for both objectives.

Each axis label and quadrant text in figure:

ΔCost (Sugar-Nectar, 1000 Rp)

Change dominated
(costlier and worse)

Change trade-off
(costlier but better)

Change trade-off
(cheaper but worse)

Change dominant
(cheaper and better)

ΔEffectiveness (Sugar-Nectar, $N_{t=4}$)

Figure 3. Cost-effectiveness assessment of changing from status quo of Aves© Nectar to sugar water feeding. Each point represents one simulation run; its coordinates indicate the difference in final population size (x-axis) and total cost over the four-year period (y-axis) between the two actions.

Discussion

The reintroduced population of olive white-eyes on Ile aux Aigrettes has undergone a strong positive growth, which managers believe is at least in part thanks to supplementary feeding. However, this success has resulted in a concomitant increase in the cost of supporting the growing population. The initial position of decision makers was to avoid changing the feeding regime if this reduced the chances of population recovery. However, the results of our experiment suggested that replacing an expensive and environmentally unstable food (Aves© Nectar) with a cheap and more

303   environmentally robust alternative (sugar water) had only a limited chance of reducing population

304   growth. Risk analysis then showed this marginal risk was more than compensated by substantial

305   economic savings.

306   Many species recovery programs might experience a similar struggle to maintain capacity to

307   support growing populations. A successful, but demanding management regime may lead to a

308   breakdown in the quality of work done as staff are unable to manage all tasks within resource

309   constraints; however, changing to a less demanding regime can be seen as an unacceptable risk

310   where uncertainty exists about its potential to maintain successful outcomes. Although such

311   uncertainty is unlikely to be eliminated completely from most conservation problems, we

312   effectively dealt with it by combining applied ecological research and formal methods for risk

313   analysis.

314   A formal definition of the problem was the first key step in our analysis. For supplementary

315   feeding, changing conservation actions that are believed to be effective might be considered in

316   order to reduce costs (Ewen *et al.*, 2014) or to move towards a more "natural" condition if this is

317   perceived as more desirable (e.g. Ewen *et al.*, 2018). Without such multiple objectives, there may

318   be no motivation to change and no decision to make. In the olive white-eye example, the status quo

319   was initially perceived as the best risk-averse decision and changing was only considered because

320   of costs. However, the results of the study suggested the switch to sugar water was likely to benefit

321   both population recovery and budget, rationally posing no risk. In different scenarios, other

322   conservation objectives such as reducing the "artificial" nature of management (Ewen *et al.*, 2018),

323   the potential for public interaction allowed by feeding stations (Walpole, 2001), or the impact on

324   other species (Cortés-Avizanda *et al.*, 2009) could be incorporated in the decision analysis methods

325   we used, for example using a weighted aggregate utility function (Keeney & Raiffa, 1993).

326   The following step was to make explicit predictions about the consequences of management

327   alternatives, to allow a rational risk assessment. In our case, prior to the experiment managers and

328   experts were uncertain about the differences between status quo and sugar water. The sugar water

329   option was considered because experience with other species suggested it might perform reasonably

330   well (Chauvenet *et al.*, 2012), but the exact outcome for olive white-eyes remained uncertain. This

331   uncertainty complicated a decision and encouraged an empirical comparison of the two actions.

332   Experimenting with critically endangered species is not straightforward. The small population

333   limited the sample size for the feeding experiment and made it impossible to test more than two

334   alternatives. A longer study could have provided more reliable estimates of survival and fecundity,

335   but implementing multiple feeding regimes was perceived as excessively risky and expensive. It is

336   probably common for managers to consider it too risky to manipulate small populations of

337   endangered species (Canessa *et al.*, 2019). However, experimental and adaptive management

338   approaches can account for this risk to ensure the benefits of learning outweigh risks in the long

339   term (Runge, 2011). Our example reinforces this potential and demonstrates the advantages of

340   accounting for uncertainty when making decisions, for example using the full range of uncertainty

341   rather than relying on mean estimates.

342   Finally, formal methods for risk analysis helped interpret the predicted consequences of actions.

343   Uncertainty meant that across all simulations, there was still a possibility that changing regimes

344   involved some negative outcomes (e.g., the population would end up growing less than if the status

345   quo had been maintained; bottom-right ICER quadrant, Fig. 3). However, the chances of negative

346   outcomes were greater if maintaining the status quo (for population growth, top and bottom right

347   ICER quadrants). Changing regimes was therefore both more cost-effective on average *and* a better

348   risk averse choice, precisely the condition we verified as first-order stochastic dominance.

349   Four years after feeding was changed in September 2015, the recovery of Mauritius olive white-

350   eyes continues successfully. The population census in June 2019 estimated between 55 and 71 adult

351   birds (S. Henshaw, Mauritian Wildlife Foundation, *pers. comm.*), very close to the predicted values.

352   Interestingly, cost was slightly overestimated in our models based on a belief that olive white-eye

territoriality and aggression would require providing supplementary feeding stations to each additional pair. In reality, behavioural changes have allowed more than one pair to share the same feeder. Therefore, the number of supplementary feeding stations had grown from 14 at the end of the experiment to 22 in June 2019. As a result, the calculated cost incurred in 2019 (~10,300 Rp) was lower than the mean prediction for sugar water feeding (~15,300 Rp), although exact costs and savings are difficult to quantify, since staff effort was simply re-allocated to other tasks.

It is intuitive, and tempting, to interpret the close match between our predictions and the realised outcomes as an indication that our decision to change management was correct. Indeed, monitoring data obviously reinforce our confidence in the initial decision. However, we must also caution against judging whether a decision under uncertainty was valid exclusively from its realised success or failure, a common mistake known as outcome bias (Baron & Hershey, 1988). Our decision was rational not because predictions matched outcomes, but because it relied on the best information available, recognized uncertainty and followed a transparent process with measurable attributes. Such a rational process should increase chances of success on average. However, for individual cases, a well-understood and well-predicted system can still give poor results, and vice versa, simply because of chance. For example, the realised population size could have fallen in the lower range of our predictions, without undermining their validity or that of our decision. Embracing this randomness is a vital, although possibly counterintuitive, step towards more evidence-based conservation decision-making.

The results of our study highlight three key lessons for conservation decision making. First, risk is a result of uncertainty, so uncertainty should be recognised explicitly when predicting the outcomes of possible management actions. Second, risk is a subjective reaction to uncertainty, and as such must reflect the subjective components of a decision (its objectives), which may go beyond the conservation outcome alone. Risks to different conservation and management objectives (e.g. losing populations against spending too much money) may of course be of different importance for

378   conservation decision makers, but ignoring fundamental objectives altogether hinders rational

379   decisions. Third, the attitude toward risk may change depending on the expected outcomes;

380   discarding change without explicitly stating and comparing those expectations may be irrational.

381   When these key principles are recognised, applied ecological studies and decision-support methods

382   can be combined to great effect to inform conservation decisions in the face of uncertainty and risk.

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