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1 **Articles**

2 **Extinction Risk and Conservation Options for Maui Parrotbill, an**  
3 **Endangered Hawaiian Honeycreeper**

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19

20 **ABSTRACT:** Extinction rates for island birds around the world have been historically high. For  
21 forest passerines, the Hawaiian archipelago has suffered some of the highest extinction rates and  
22 reintroduction is a conservation tool that can be used to prevent the extinction of some of the  
23 remaining endangered species. Population viability analyses can be used to assess risks to

24 vulnerable populations and evaluate the relative benefits of conservation strategies. Here we  
25 present a population viability analysis to assess the long-term viability for Maui parrotbill(s)  
26 (Kiwikiu) *Pseudonestor xanthophrys*, a federally endangered passerine on the Hawaiian island of  
27 Maui. Contrary to indications from population monitoring, our results indicate Maui parrotbills  
28 may be unlikely to persist beyond 25 years. Our modeling suggests female mortality as a primary  
29 factor driving this decline. To evaluate and compare management options involving captive  
30 rearing and translocation strategies we made a female-only stage-structured, meta-population  
31 simulation model. Due to the low reproductive potential of Maui parrotbills in captivity, the  
32 number of individuals (~ 20% of the global population) needed to source a reintroduction solely  
33 from captive reared birds is unrealistic. A reintroduction strategy that incorporates a minimal  
34 contribution from captivity and instead translocates mostly wild individuals was found to be the  
35 most feasible management option. Habitat is being restored on leeward east Maui, which may  
36 provide more favorable climate and habitat conditions and promote increased reproductive  
37 output. Our model provides managers with benchmarks for fecundity and survival needed to  
38 ensure reintroduction success, and highlights the importance of establishing a new population in  
39 potentially favorable habitat to ensure long-term persistence.

40

41 *Key words:* extinction risk, Hawai‘i, Maui parrotbill, population viability analysis, *Pseudonestor*  
42 *xanthophrys*, reintroduction, translocation

43

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63

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65 Running Head: Risk and Recovery options for Kiwikiu

66

67

## Introduction

68 Bird species across the world are in peril; one in eight species is globally threatened  
69 (BirdLife International 2014) and extinction rates are highest on islands (Gilpin and Soulé 1986;  
70 Steadman 2006). Extinction risk to small populations is explained by a broad suite of ecological  
71 characteristics; stochastic threats (e.g., environmental or catastrophic) and deterministic factors  
72 (e.g., demographic or genetic; Shaffer 1981, 1987). While each characteristic or threat alone may

73 lead a population to extinction, together they produce interacting effects that can increase  
74 extinction probabilities, the ‘extinction vortex’ (Gilpin and Soulé 1986; Soulé and Mills 1998;  
75 Mills 2007).

76 Population viability analysis (PVA) is an analytical tool used to measure the processes that  
77 can lead to extinction. Data can be applied to a suite of models that combine the effects of  
78 deterministic and stochastic factors to estimate a population’s probability of future persistence  
79 (Gilpin and Soulé 1986; Caughley 1994; Beissinger 2002). Historically, PVA was used to  
80 quantify absolute risk of extinction and assess population sensitivity to model parameters, but  
81 arguably its real value is in an applied context, to examine the relative benefits of alternative  
82 management actions and estimate relative probability of extinction under different strategies  
83 (Akçakaya and Sjögren-Gulve 2000; Ellner and Fieberg 2003). In recent years PVA-type models  
84 have been applied to support specific endangered species management decisions such as  
85 landscape planning and habitat acquisition decisions (Bonnott et al. 2011; Robinson et al. 2016),  
86 allowing for mitigating incidental killings or harassment (McGowan and Ryan 2009; McGowan  
87 et al. 2011a), reintroduction decisions (Converse et al. 2013; Converse and Armstrong 2016),  
88 recovery planning (McGowan et al. 2014; Evans et al. 2016) and deciding whether species  
89 warrant US Endangered Species Act (ESA 1973, as amended) protections (Regehr et al. 2015,  
90 Reference S1; McGowan et al. 2017). Well-crafted PVA models can be designed and used to  
91 assess future extinction risk and examine and inform specific management decisions for species  
92 at risk (Runge 2011). Ideally, a PVA should incorporate the essential aspects of a population’s  
93 biology, and when correctly parameterized it can provide insights into what factors constitute the  
94 greatest threats to the population’s survival (Mills 2007). The species-specific information  
95 needed to calculate a population’s absolute risk of extinction with precision and to compare

96 relative extinction risk under different management scenarios is rarely achievable for endangered  
97 species, particularly those that exist at low density or have cryptic behaviors. However, in those  
98 instances where endangered species have been sufficiently well studied, PVA is a useful tool for  
99 conservation managers (Ralls et al. 2002). Indeed, predicting time to extinction under multiple  
100 scenarios can inform conservation decisions, help guide management efforts and prioritize and  
101 evaluate different management options (Clark et al. 1991; Cook et al. 2012).

102 The Maui parrotbill (Kiwikiu) *Pseudonestor xanthophrys* is listed as endangered pursuant to  
103 the ESA and is of immediate conservation concern (USFWS 1967; IUCN 2012; Figure 1). The  
104 Maui parrotbill is a feeding specialist with a parrot-like beak for extracting insect prey from bark  
105 and decaying wood (Simon et al. 1997). Maui parrotbills are long-lived, strongly monogamous  
106 passerines that can reproduce for at least 15 years (Becker et al. 2010; Mounce et al. 2013,  
107 2014). Breeding pairs typically produce only one offspring per year, exhibit prolonged parental  
108 care (5-17 months) and occupy relatively large home ranges averaging ~12 ha (Mounce et al.  
109 2013; Simon et al. 2000; Warren et al. 2015). Adults typically show further delayed maturation  
110 and do not breed until their third year, although second-year females may breed more commonly  
111 than males (Maui Forest Bird Recovery Project, unpubl. data). These slow life-history traits  
112 likely indicate that the species relies heavily on adult survival. Similar life history traits are seen  
113 in 'Akiapola'au *Hemignathus wilsoni* the Maui parrotbill's closest living relative, but uncommon  
114 in the Drepanidini tribe in general (Pratt et al. 2001). Maui parrotbills were once abundant on the  
115 islands of Maui and Molokai (James & Olson 1991), but have undergone substantial declines  
116 since the arrival of humans ~ 800-1000 years ago (Mounce et al. 2015). Today the wild  
117 population comprises ~500 individuals, and occupies less than 50 km<sup>2</sup> on windward east Maui  
118 (502 ± 116 [SE] reported from Scott et al. 1986; 590 ± 208 reported from Camp et al. 2009).

119 Population-wide surveys have not revealed a trend in the population since range-wide surveys  
120 were begun in 1980, although abundance estimates have remained fairly consistent (Camp et al.  
121 2009).

122 Due to an apparent lack of resistance by the native forest birds to mosquito-borne diseases,  
123 such as avian malaria *Plasmodium relictum*, forests above 1500 m elevation provide the only  
124 existing refuge for most native Hawaiian honeycreepers (Scott et al. 1986; Mountainspring 1987;  
125 Simon et al. 1997) including the Maui parrotbill. Avian malaria is now moving into higher  
126 elevations, coincident with increasing average temperatures in Hawaii and gradually eroding  
127 available habitat for these species (Benning et al. 2002; Giambelluca et al. 2008; Harvell et al.  
128 2002). Moreover, these high elevation windward habitats are suspected to be suboptimal for  
129 Maui parrotbills. These habitats contain few koa *Acacia koa*, a historically observed Maui  
130 parrotbill-preferred foraging substrate (Perkins 1903), and the prevalence of nest failures in these  
131 areas are high, frequently attributable to severe weather (USFWS 2006, Reference S2; Becker et  
132 al. 2010; Mounce et al. 2013). The historically forested island of Maui once provided almost  
133 island-wide habitat for Maui parrotbills including lowland and leeward (southeast) forests (James  
134 and Olson 1991). Little apparent habitat exists beyond the species' current range with the  
135 exception of a few remnant forest tracts on leeward east Maui, such as those found in Nakula  
136 Natural Area Reserve (NAR; 20.6°N, 156.3°W; 1097 – 2804 m in elevation; Figure 2), which is  
137 currently being reforested specifically to provide habitat for Maui parrotbills and other native  
138 forest birds. In addition to the wild population, there is a small captive flock of Maui parrotbills  
139 (currently 15 individuals) that was established in 1997 and is managed by San Diego Zoo  
140 Institute for Conservation Research. Together, the captive flock and habitat restoration efforts  
141 have paved the way for several potential conservation strategies for this species.

142 We applied PVA models using detailed data from demographic, genetic and ecological  
143 studies recently completed for this species (Mounce et al. 2013, 2014, 2015; Warren et al. 2015),  
144 to assess long-term viability of Maui parrotbills and evaluate potential conservation strategies.  
145 We used a custom-made simulation model to understand key limiting factors for the current  
146 population by determining which demographic variable(s) were most influential for population  
147 growth and long-term viability. We also expanded upon this model to examine the effects of (1)  
148 different management strategies to improve productivity and survival in the species' current  
149 range, (2) removing individuals from the wild for reintroduction elsewhere, and (3) establishing  
150 an additional geographically-distinct population in the leeward forests, currently unoccupied by  
151 Maui parrotbills, which may provide a long-term refuge for the species. We use our findings for  
152 the Maui parrotbill to illustrate the broader value of using PVA models to help guide the  
153 decision-making needed to plan future conservation strategy for endangered species.

154

155

## **Methods**

### **Base models**

157 Our base model was parameterized using estimates of Maui parrotbill vital rates from  
158 Mounce et al. (2013, 2014, 2015; Table 1). This base model was designed to represent the Maui  
159 parrotbill population in its current state without incorporation of any change in threats (besides  
160 normal demographic and stochastic effects of small population size) and thus produces  
161 simulations of a probable population trajectory without additional management actions. The  
162 effect of environmental variation on the annual reproduction and survival probabilities was not  
163 separately included in the model as these parameters were derived from long-term data sets that  
164 already averaged temporal variation (Table 1).



165 To explore different viability scenarios on the current wild populations we created a female-  
166 only stage-structured population model in R 3.4.2 (R Core Team 2017). We designed this model  
167 to incorporate demographic values (including variance) from field studies and to predict  
168 population dynamics as discrete, sequential events that incorporate environmental and  
169 demographic stochasticity through random draws from probability distributions. The model was  
170 designed to run 1,000 simulations to generate a distribution of possible fates that a population  
171 might experience under a given set of parameters. We did not model parametric uncertainty  
172 directly into our simulations using Bayesian PVA or double loop structures to incorporate hyper-  
173 parameters for demographic rates (e.g., McGowan et al. 2011b) but rather explored the effects of  
174 parametric uncertainty on model predictions through specific simulations, similar to Goodman et  
175 al. (2003). Each simulation steps through a series of events that describes an annual cycle  
176 (reproduction, mortality, and dispersal among populations,).

177 Our model was a stage-structured model that included a young of the year age class (0-1 yrs  
178 old), an immature age class (2-3 yrs old) and a breeding adult age class to reflect known age at  
179 maturation ( $\geq 3$  yrs old). We incorporated a dispersal function between sub-populations and  
180 different survival rates for juvenile ( $0.3 \pm 0.05$  [Woodworth and Pratt 2009]) and adult birds  
181 ( $0.72 \pm 0.02$  [Mounce et al. 2014]; see equations below) modeled as beta distributed random  
182 variables. Fecundity rate was incorporated as the number of female offspring fledged per  
183 breeding female and was modeled as a log-normally distributed random variable. Mean annual  
184 fecundity was set at 0.2415 with a 0.15 standard error to incorporate environmental variability  
185 into the reproductive rate (Mounce et al. 2013; see equation below). For some parameters where  
186 the source data did not include an estimate of variance we used a CV of 15%, an accepted  
187 practice in PVA models when no estimate of variance is available (Morris and Doak 2002). In

188 our model we also included an estimate of carrying capacity, whereby if a specified abundance  
189 threshold (432 females) was exceeded, the fecundity rate for that year was set to 0. Carrying  
190 capacity (K) was calculated using the Maui parrotbill range used in Camp et al (2009), 51.07  
191 km<sup>2</sup>, and pair home range size from Warren et al. (2015), 0.118 km<sup>2</sup>. This approximates the  
192 number of pairs (females) given total saturation in the entire range. Carrying capacity was set at  
193 92 females for the future Leeward population based on 10.9 km<sup>2</sup> of habitat that is being restored.  
194 We set the population ceiling very high compared to current estimated abundance. Informal  
195 sensitivity analysis indicates that unless the population is currently very close to, or in excess of,  
196 carrying capacity selecting the population carrying capacity has little influence on population  
197 predictions since all of our simulations decline and are not limited by K. If model predictions  
198 were sensitive to carrying capacity, more sophisticated approaches to estimating carrying  
199 capacity could be implemented, such as estimating available habitat and dividing by estimated  
200 female home range. This is a simplistic and fairly severe effect of abundance on demographic  
201 rates, however the density dependent mechanisms for this species are not known and a ceiling  
202 type function allows us to prevent exponential population growth without speculating on the  
203 functional form of density dependence (Morris and Doak 2002; McGowan and Ryan 2009). We  
204 modeled these processes as population level stochastic processes, not as individual based  
205 processes, where binomial functions are more appropriate for survival and Poisson are more  
206 appropriate for fecundity. The initial population of 292 females was calculated using the most  
207 recent available density estimate, 11.41 Maui parrotbill per km<sup>2</sup> (Brinck et al. 2011), estimated  
208 within a subset of the species range and extrapolated to the entire 51.07 km<sup>2</sup> range.

209 Future juvenile bird abundance ( $N_t^J$ ) per year was modeled as the product of the number of  
210 breeding adults ( $N_t^A$ ), the fecundity rate ( $F_t^A$ ) and the survival rate of young of the year ( $S_t^Y$ ), as  
211 follows:

$$212 \quad N_{t+1}^J = N_t^A \times F_t^A \times S_t^Y.$$

213 The number of adults in future years was a product of the number of adults ( $N_t^A$ ) and their annual  
214 survival rate ( $S_t^A$ ), plus the product of the number of juvenile birds ( $N_t^J$ ) and their annual survival  
215 rate ( $S_t^J$ ), as follows:

$$216 \quad N_{t+1}^A = N_t^A \times S_t^A + N_t^J \times S_t^J.$$

217 Adult and juvenile survival rates in the simulations were environmentally stochastic and drawn  
218 from a beta distribution, where the alpha and beta shape parameters were derived from the  
219 survival estimates reported in Mounce et al. (2014) using the method of moments calculations  
220 (see Morris & Doak 2002).

221 We modified the base model in four ways to increase demographic rates above the estimates  
222 from field studies. We used these modifications to capture parametric uncertainty in our  
223 estimated demographic parameters, following the recommendations of Goodman (2002) who  
224 demonstrated that structured changes in demographic rates within models can allow researchers  
225 to explore the effects of parametric uncertainty. Because our base model predicted rapid and near  
226 certain extinction for the population in contrast to the observed patterns in density estimates over  
227 the last 20 years (Camp et al. 2009), there is the possibility that some of the parameter estimates  
228 were not accurate in either the PVA simulations or in the count data analyses. As such, we  
229 investigated four suspect parameter estimates in detail that may have been driving the projected  
230 decline in population size. First, the estimate of juvenile survival (0-1 yrs old) presented in  
231 Mounce et al. (2014; 0.17) was markedly lower than other Hawaiian passerines (average  $0.32 \pm$

232 0.03; Woodworth and Pratt 2009). The Mounce et al. (2014) estimate was generated from only  
233 10 individuals and had a large standard error ( $\pm 0.15$ ). Second, the estimate of annual  
234 reproductive success (ARS) presented in Mounce et al. (2013) was derived from the core-  
235 breeding season for Maui parrotbills (January-June). Breeding attempts for this species have  
236 been observed in 11 months of the year. Therefore, it is probable that while this ARS estimate  
237 may capture the majority of the success in a given year, it is likely to be an underestimate of the  
238 true ARS over the entire calendar year. Third, an annual decline in carrying capacity (K) in their  
239 current habitat is inevitable because of predicted climate change and the associated upslope  
240 movement of avian malaria and its vector (Benning et al. 2002; Giambelluca et al. 2008; Harvell  
241 et al. 2002). Fourth, genetic analysis has shown that the Maui parrotbill population is not  
242 contiguous across its range likely because of limited dispersal between two subpopulations  
243 (Mounce et al. 2015).

244 To address these issues, we modified our base model by (1) increasing juvenile survivorship  
245 from 17% to 32% to reflect values found in the other Hawaiian passerines (equal to 39-44% of  
246 adult survival), (2) increasing the percentage of breeding females each year by 10% to account  
247 for a reasonable estimate of less well-documented ARS in the months outside of January-June,  
248 (3) decreasing K by 1% per year to account for the influx of avian malaria resulting from climate  
249 changes (Giambelluca et al. 2008), and (4) dividing the population into two sub-populations with  
250 an associated K for each calculated using home range data from Warren et al. (2015). Further,  
251 we allowed for the possibility of dispersal between sub-populations with the number of  
252 dispersers each year ( $D$ ) modeled as Poisson distributed random variables with a mean  $I$  (set to  
253 2% of the size of each subpopulation; Modified Base Model in Table 1). We do not have enough  
254 data on movements to estimate these rates of movement empirically. We set mean  $I$  between the

255 two extant wild populations to be a low annual rate (2%), because we know that movement  
256 between the populations does occur but it appears to be very uncommon (H. Mounce,  
257 unpublished data). Thus the number of juveniles in each time period ( $t$ ), i.e., year, were added to  
258 each population was modeled as follows:

$$D_t^{i,j} \sim \text{Poisson}(I_t^i \times N_t^{A+I,i})$$
$$N_{t+1}^I = N_t^A \times F_t^A \times S_t^Y + \sum D_t^{j,i} - \sum D_t^{i,j}$$

259 where  $i$  represents the current sub-population being projected, and  $j$  indicates the other sub-  
260 populations to or from which individuals can be translocated.

261 For all individual simulations in R we used 1,000 iterations spanning 25 years. Although  
262 longer time frames are more appropriate for assessing the predicted longevity of a species, for  
263 this exercise our focus was on the immediate viability risk and the effects of conservation actions  
264 that can be implemented to prevent imminent extinction. Due to persistent problems associated  
265 with introduced predators, continued loss of habitat, invasive species and the inherent risks of a  
266 critically endangered organism, modeling population dynamics for this species on a longer  
267 timeframe would not provide any additional insight for critical management needs.

268

## 269 **Sensitivity Analyses**

### 270 **Measures of viability**

271 Population viability analyses are limited by the quality of the input parameters available for a  
272 given species under each given scenario and do not identify absolute probabilities of extinction  
273 in a given time frame (Akçakaya and Sjogren-Gulve 2000; Reed et al. 2002). Viability measures  
274 most commonly presented in PVA studies include extinction probability, population size and  
275 estimates of time to extinction. However, it is important to evaluate the full suite of quantitative

276 measures that PVAs produce to evaluate population viability across all models rather than  
277 relying solely on these most common measures (Pe'er et al. 2013). Consequently, for each model  
278 we present mean finite rate of growth ( $\lambda$ ), probability of quasi-extinction ( $N < 10$ ; PE), median  
279 population size from all iterations ( $N$ -all) and median population size from extant populations ( $N$   
280  $\geq 10$ ;  $N$ -extant). Quasi-extinction is somewhat arbitrary in nature but here we used  $N \geq 10$  as our  
281 threshold because we expect that when the population falls to single digit abundance drastic  
282 changes in management approach would be enacted and demographic stochasticity, rather than  
283 environmental stochasticity, would become the predominate force driving population annual  
284 changes in abundance.

285

### 286 **Testing demographic sensitivity**

287 Demographic sensitivity and elasticity are common metrics to report in PVA analyses  
288 (Morris and Doak 2002; Reed et al. 2002). To test the demographic sensitivity and elasticity of  
289 the current wild populations we used the ‘popbio’ package in R (Stubben and Milligan 2007).  
290 We applied the sensitivity and elasticity functions in the ‘popbio’ package to the deterministic  
291 projection matrix for the Base Model. The sensitivity and elasticity analyses use the Modified  
292 Base Model parameter as a deterministic matrix (Table 1).

293

### 294 **Population viability with management**

295 A working group of researchers and managers (Maui Parrotbill Reintroduction Working  
296 Group) has developed a plan to reintroduce Maui parrotbills to Nakula NAR over a three-year  
297 period, a strategy designed to balance probability of success with efficient use of resources.  
298 Based on Maui parrotbill home range size (Warren et al. 2015), Nakula NAR may be able to

299 support ~12 Maui parrotbill pairs in the first few years of a reintroduction program. Considering  
300 these restrictions in the total numbers of individuals the area can immediately support, we  
301 therefore tested a variety of reintroduction scenarios whereby six pairs were released each year.  
302 While there are many possible scenarios that could be tested, we selected six that we thought  
303 were realistic given current management opportunities:

- 304 *i.* Release only the captive birds currently available to establish a second population;
- 305 *ii.* Augment the captive flock with wild birds such that the captive flock alone would  
306 source a second population;
- 307 *iii.* Augment the captive flock with wild birds such that the captive flock would provide  
308 half the individuals needed for reintroduction with the other half from translocated  
309 wild individuals;
- 310 *iv.* Augment the captive flock with wild birds such that the captive flock would provide 1  
311 female per year in combination with translocated wild individuals;
- 312 *v.* Release only the captive birds currently available in combination with wild  
313 translocations to establish a second population; and,
- 314 *vi.* Release wild translocated individuals to establish a second population with no input  
315 from the captive population.

316 We modified the female-only stage-structured, meta-population simulation model in R to  
317 evaluate and compare management options involving captive rearing and translocation strategies  
318 tailored to the recovery requirements for this species. This model was based on a spatially  
319 implicit meta-population structure of four separate sub-populations in the simulations. Two  
320 populations represent the existing east (Hanawi NAR) and west (TNC Waikamoi Preserve)  
321 populations (Mounce et al. 2015) on the windward slopes of Haleakala (Figure 2). Another

322 population represents the proposed third population that will be established on the leeward slopes  
323 of Haleakala (Nakula NAR; Figure 2), and a fourth population represents a captive breeding  
324 population that may serve as a source of individuals for release into the wild populations. The  
325 two wild populations were modeled with the demographic parameters described above for the  
326 Modified Base PVA Model. For the third (not yet established) reintroduced population we tested  
327 the effects of increased survival and fecundity rates on the probability of successfully  
328 establishing a wild self-sustaining population and on overall species extinction probability. To  
329 account for the potential that leeward habitats may be higher quality due to fewer storms and  
330 overall less precipitation than windward habitats, we used demographic rates that were 5%, 10%  
331 and 20% greater than those documented for Maui parrotbills to model the new population. In  
332 many cases these increased demographic values are more aligned with those of other Hawaiian  
333 honeycreepers (Woodworth and Pratt 2009). For example, a 20% increase in annual adult Maui  
334 parrotbill survivorship is 0.92 and annual adult survivorship of other Maui species have been  
335 estimated as high as 0.95. Although, a 20% increase in juvenile survivorship from the modified  
336 base model may be optimistic given that this parameter was already taken from the average from  
337 all honeycreeper species (0.32) and this parameter has not been found to be quite this high (20%  
338 increase in juvenile Maui parrotbill survivorship = 0.52) in other species.

339 We used the most recent density estimate from Brinck et al. (2011), 11.41 Maui  
340 parrotbill/km<sup>2</sup>, and extrapolated to generate initial abundance in the east and west populations  
341 using range sizes of 41.8 km<sup>2</sup> and 9.3 km<sup>2</sup>, respectively. These range sizes are based on the 51.07  
342 km<sup>2</sup> Maui parrotbill range map used in Camp et al. (2009) and divided at the Ko‘olau Gap, a  
343 large topographic feature thought to limit gene flow between the populations (Mounce et al.  
344 2015). Based on an initial abundance of 583 and these spatial parameters, we set the initial



345 abundance in the east wild population at 239 females, the west wild population at 53 females,  
346 and the third wild, yet to be established population at 0. With these populations combined, we set  
347 initial abundance for the entire current range at 292 female Maui parrotbills.

348 The captive population was modeled differently from the wild populations since in captivity  
349 the birds are not subject to the same ecological processes. We modeled the captive populations as  
350 an individual based model, which is common for captive populations (Lacy and Pollak 2014)  
351 because the conditions are more controlled. Instead, once established, the future abundance in  
352 captivity ( $N_{t+1}^C$ ) is the current number of individuals ( $N_t^C$ ), plus the number successfully reared  
353 ( $N_t^B$ ), minus the number that died ( $N_t^D$ ), which were modeled as Poisson distributed random  
354 variables with a mean of 2.0 and incorporated into the projection as follows:

$$N_t^B \sim \text{Poisson}(2)$$

$$N_t^D \sim \text{Poisson}(2)$$

355 
$$N_{t+1}^C = N_t^C + N_t^B - N_t^D.$$

356 We set initial abundance in captive population at 7 females to reflect current conditions of the  
357 captive flock. We set the captive population to be approximately stable with no increase or  
358 decrease on average (without inputs from the wild or outputs to the wild) with equal mean  
359 number of births and deaths each year (2). The captive breeding program thus far is very small  
360 and has limited production (i.e., births each year) so our rates of two births and deaths reflect the  
361 production capacity and limited space for the captive population.

362 Movements between the sub-populations were restricted to translocations in captive and the  
363 future leeward populations. Movements involving the current wild populations included  
364 translocations amongst all populations and natural dispersal between the east and west  
365 populations only. The projected abundance in a subpopulation was a function of natural

366 population dynamics (as described above), and the number of individuals added to and  
367 subtracted from the population as follows:

$$N_{t+1}^{A,i} = (N_t^{A,i} \times S_t^A) + (N_t^{I,i} \times S_t^I) + \sum T_t^{j,i} - \sum T_t^{i,j} + \sum D_t^{j,i} - \sum D_t^{i,j}$$

368 where  $T$  indicates the number of birds moved by management intervention and  $D = 0$  in the  
369 leeward and captive populations. The model was written in a generalized form so that birds could  
370 be moved from any sub-population to another (Text S1), but in our simulations management  
371 actions were limited to establishing a new sub-population and/or contributing to the small  
372 captive population. Translocations of birds between sub-populations was specified for a limited  
373 number of years such that if abundance in the west and east sub-populations fell below 25% of  
374 their starting population size, removing individuals from that sub-population was prohibited.  
375 Lastly, individuals introduced to the wild from captivity are typically less successful (Fischer and  
376 Lindenmayer 2000). The fact that Maui parrotbills will be re-established in a different habitat  
377 type increases the uncertainty regarding their survival. To reflect that uncertainty rather than use  
378 the estimated survival rates from the empirical studies on the windward populations (Mounce et  
379 al. 2014) in our model, we made first-year survival of captive-released birds an annually varying  
380 uniformly distributed random number bounded between 0.3 and 0.9. There is no data available to  
381 characterize the form and shape of the post release survival function so using a uniform  
382 distribution is appropriate in this case. The lower-bound value is based on success of Palila  
383 (*Loxioides bailleui*) translocated to the north slope of Mauna Kea on Hawaii Island (Banko et al.  
384 2009).

385 The reintroduction scenarios differed mainly based on the source of birds (i.e., the east and  
386 west wild populations, and the captive population). The goal of the captive breeding program  
387 from its onset has been to develop a sustainable breeding program for the species in the event of

388 a collapse of the wild populations and/or to act as a source for reintroduction as new habitat  
389 became available. However, the captive program has only been moderately successful. As of  
390 2015, the captive population consisted of seven females and eight males, which together produce  
391 an average of one bird each year. Given a sex ratio of 50:50 this represents a rate of 0.07 females  
392 produced per female per year. Realistic options for sourcing birds for reintroduction from the  
393 captive population include:

- 394 a) Releasing a large proportion of the existing captive birds (e.g., 7 females and 7 males) in  
395 a single year;
- 396 b) Releasing a minimal subset of the captive population (e.g., 1 female and 1 male) over the  
397 course of a few years; and
- 398 c) Augmenting the captive flock with wild birds, allowing for the release of a larger number  
399 of captive birds over the course of a few years.

400

401

## Results

402 All population viability models predicted a negative population growth rate (Table 2) in the  
403 wild populations, with none of the trajectories for the east and west subpopulations as well as the  
404 meta-population persisting beyond 25 years (Figure 3, Figure 3A and 3B). As expected the  
405 sensitivity analyses identified female mortality, followed by fecundity and juvenile mortality as  
406 the main contributors to the overall population trajectory (Table 3). There are a number of  
407 constraints to carrying out the proposed reintroduction scenarios given demographic variables in  
408 the captive and wild populations. Assuming no changes to the fecundity among the captive flock,  
409 sourcing the reintroduction using only captive birds would require either using six available  
410 females (leaving one female in captivity; *i*), or moving 68 females from the wild into captivity to

411 increase the captive populations' productivity (*ii*). The third option (*iii*) would still require 30  
412 additional females to be brought into captivity to supplement the captive flock in order to source  
413 50% of the translocations (Table 4). The fourth option (*iv*) requires that eight additional birds are  
414 brought into captivity so that the captive flock could consistently supply one female per year for  
415 reintroduction efforts. Population trajectories among the last three scenarios (*iv*, *v* and *vi*) are  
416 similar as they use the same input parameters, but the scenarios differed with regards to the  
417 origin of individuals (Table 4; Figure 4).

418 The increase of demographic parameters based on the assumption that leeward forests will  
419 provide higher quality habitat for Maui parrotbills resulted in different population trajectories  
420 (under reintroduction scenario *iv*) after the initial three-year reintroduction timeframe (Figure 5).  
421 All reintroduction scenario models show that a 10% increase in key demographic rates is not  
422 expected to be sufficient to maintain the reintroduced population ( $\lambda = 0.96$ ). However, a 15%  
423 increase results in a likely stable population ( $\lambda = 1.0$ ) and a 20% increase results in a growing  
424 population ( $\lambda = 1.1$ ; Figure 5).

425

## 426 Discussion

427 Our PVA models provide a tool to evaluate management scenarios and generate demographic  
428 benchmarks necessary for a sustainable Maui parrotbill population. The rapid decline projected  
429 by these models highlights the fact that certain aspects of the species' biology, ecology and life  
430 history traits (e.g., mainly single egg clutches, prolonged parental investment) make this species  
431 in its current state (e.g., small, contracting range, occupying potentially suboptimal habitat)  
432 highly vulnerable to extinction. Our models allowed us to identify the demographic rates most  
433 limiting the species, to explore potential management solutions, and identify the most promising  
434 scenarios for reintroducing the species to previously occupied leeward mesic forests. Given the

435 assumptions in our model and current restraints in captive Maui parrotbill productivity, we found  
436 that a reintroduction scenario that incorporates a minimal contribution from captivity and instead  
437 translocates mostly wild individuals to be the most practicable strategy.

438 Our population models highlight the strength and weakness of several conservation strategies  
439 that managers could implement given existing resources and capabilities in attempts to recover  
440 the wild population. Given that the population model does not reach carrying capacity, simply  
441 increasing available habitat in the current Maui parrotbill range may not increase the total  
442 population. Alternatively, if managers are able to augment the current habitat (e.g., through  
443 threat management, such as predator control) to increase quality, and thus increase some of the  
444 more sensitive parameters (i.e., female survival) within the current populations, they may be able  
445 to increase the population viability. Unfortunately, these options appear to be quite limited for  
446 Maui parrotbills for several reasons.

447 Weather has been identified as a key, limiting factor to reproductive success of Maui  
448 parrotbills, with high incidence of nest failure in heavy rain events (Mounce et al. 2013).  
449 Although weather cannot be manipulated, there have been numerous other unsuccessful attempts  
450 to manage Maui parrotbills within their current forest habitat. These efforts have included trying  
451 to increase productivity, survival, or both by providing supplemental food to wild individuals,  
452 decreasing predation risk through control of invasive mammalian predators, and decreasing nest  
453 predation by protecting nest trees from mammalian predation (suspected rat depredation on Maui  
454 parrotbill nests; HL Mounce, personal observation). The forest currently occupied by Maui  
455 parrotbills is native Hawaiian rainforest that is already protected (i.e., fenced and free of  
456 ungulates) and actively managed by the National Park Service, the State of Hawaii, and The  
457 Nature Conservancy. Thus, the vegetation community is in prime condition, and there are limited

458 options for other management interventions that can improve vital rates in Kiwikiu. It is  
459 particularly concerning that the models presented here predict a rapid decline in the species given  
460 that the species primarily occupies areas with such a high degree of protection. This leaves few  
461 options beyond landscape-scale management actions (e.g., aerial broadcast rodenticide) in the  
462 current range. Therefore, establishing an additional population that may appreciate increased  
463 vital rates represents a measure that may ensure long-term persistence of the species.

464 Our model simulations and predicted extinction probabilities are limited by the precision of  
465 the demographic parameters estimates. While this study has used the most comprehensive data  
466 available on the Maui parrotbill, there is still uncertainty in several critical parameter estimates.  
467 The variables with the most uncertainty included initial population size and annual habitat loss as  
468 predicted through climate change models. However, our sensitivity analysis identified female  
469 mortality at all ages as the parameter most responsible for driving the observed population  
470 changes, with juvenile survivorship and fecundity playing a lesser role – yet (with the exception  
471 of juvenile survivorship) these are all parameter estimates that we have high confidence in from  
472 empirical studies (e.g., Mounce et al. 2015). Our model predictions contrast with the estimated  
473 population estimates based on point counts over the last 20 years (Camp et al. 2009). These  
474 disparities could be the result of un-modeled observation error in the count data, inaccurate  
475 parameter estimates in the PVA model, or a variety of other analytic or sampling issues. It is  
476 possible that the PVA is predicting a decline that has not yet been observed in the count data but  
477 something we may see in the near future. All abundance estimates for this species are associated  
478 with extremely large confidence intervals reflecting the low number of detections typically  
479 recorded for the species on these counts. As a result, significant trends have not been found and,  
480 although it has been tempting to say that the population is stable given similar abundance

481 estimates between years, we do not know whether the wild population is stable with any  
482 certainty. It would be a mistake to ignore model predictions based solely on how well the output  
483 conforms to count estimates for such a cryptic species. We suggest that a productive path  
484 forward would be to implement an integrated population model that incorporates both  
485 demographic data and count data, applies observation error models to both data sets and  
486 integrates the analysis to estimate key demographic parameters using all available data (e.g.,  
487 Schaub and Abadi 2011; Rushing et al. 2017).

488       Given the lack of management actions for increasing population viability in currently  
489 occupied habitat, an alternative is to establish new populations, particularly in areas with high-  
490 quality habitat. Historically, Maui parrotbills were observed to prefer koa as a foraging substrate  
491 (Perkins 1903) and it stands to reason that habitats containing a higher proportion of koa, which  
492 tend to occur in drier, leeward areas on Maui, were important areas for the species. Furthermore,  
493 sub-fossil records show a distribution of this species across the island, not restricted to the high  
494 elevation wet windward forests where they are found currently (James and Olson 1991). Today,  
495 there are no Maui parrotbills in koa-dominated forests on Maui. We do not know whether Maui  
496 parrotbills were historically distributed at higher densities in the wet windward forests, but it  
497 may be that these areas were always marginal habitat. Regardless, if managers do not have the  
498 tools to successfully manage this species in currently occupied habitats, then increasing the range  
499 of occupied habitats may provide a viable long-term conservation strategy. Furthermore,  
500 establishing geographically disjunct populations is good conservation practice, as having an  
501 entire global population of a species within one 51 km<sup>2</sup> area (such as the Maui parrotbill) puts it  
502 at greater risk of extinction in the event of a severe hurricane or other weather event.

503        Given the apparent limitation of management options in currently occupied habitats, moving  
504 birds from existing populations to the leeward side of Haleakala, Nakula NAR, a drier, koa-  
505 dominated habitat, as modeled here, may be necessary. Furthermore, the birds in this new habitat  
506 may be able to benefit from increased survival and productivity, key to the species long-term  
507 success. Options for moving birds include moving birds from the wild, using captive-bred birds,  
508 and a combination of these alternatives. Using captive-bred individuals can have ecological  
509 consequences such as behavioral deficiencies, high susceptibility to starvation and disease, high  
510 post-release depredation rates and overall low reintroduction success rates that have been widely  
511 documented (Curio 1996; Fischer and Lindenmayer 2000; Jule et al. 2008; Rantanen et al. 2010).  
512 Captive Maui parrotbills have the additional disadvantage of reduced genetic variation and  
513 significant genetic differentiation compared to some wild individuals (pairwise  $F_{st}$  and  $R_{st}$   
514 between west and captive populations [ $F_{st} = 0.1$ ;  $R_{st} = 0.16$ ] Mounce et al. 2015). Furthermore,  
515 given the low reproduction rate of captive Maui parrotbills, using only captive-bred birds would  
516 1) render the current captive population ineffective, 2) establish a new population with genetic  
517 variation from just a few females (*i*) or 3) require that a large number of wild individuals be  
518 brought into captivity (*ii* and *iii* requiring 23% or 10%, respectively, of all wild females).  
519 Without considering potential effects on the wild populations, the resources necessary to capture  
520 and care for these high numbers of an endangered species in captivity is unrealistic with current  
521 conservation support available in Hawaii (Leonard 2008). Conversely, if the availability of  
522 resources for this type of hands-on management substantially increased, there may be some  
523 advantages, namely that captive birds may possibly anchor any wild birds to the release area,  
524 which would facilitate monitoring (Banko et al. 2009). A major obstacle in translocations of wild  
525 individuals is that they often reject the habitat close to release sites and travel long distances



526 before settling (Stamps and Swaisgood 2007), exhibiting preferences that captive individuals  
527 may not have.

528 Scenario *iv* models a reintroduction that incorporates a minimal contribution from captive  
529 individuals and has the advantage of potentially being among the least expensive scenarios. The  
530 ultimate monetary costs of many of the key steps involved in these scenarios remain unknown  
531 and in some cases are impossible to predict (e.g., the amount of field time required to capture 68  
532 females [scenario *ii*]). Without these figures, a cost comparison among all scenarios is  
533 impossible at this time. However, scenario *iv* calls for the least amount of effort devoted to  
534 capturing wild individuals to be added to the captive population, a benefit over *ii* and *iii*, while  
535 also making use of the investment already made toward maintaining the captive population, an  
536 advantage over *vi*. This scenario also does not deplete the already small captive population,  
537 unlike *i* and *v*, and minimizes the addition of new birds to captivity and thus the costs in  
538 maintaining the larger captive population. In order for the leeward population to be considered  
539 genetically viable (Foose 1993) birds from both the east and the west need to be incorporated  
540 into the releases, yet captive birds were sourced only from the east population. Therefore,  
541 scenario *iv* would also likely provide any new populations with the most comprehensive genetic  
542 foundation considering the genetic differentiation observed between the east (including captive)  
543 and west wild Maui parrotbill populations (Mounce et al. 2015).

544 We further explored scenario *iv* by looking at the reintroduced population's viability using  
545 parameter values from the current wild population (Figure 5; Mounce et al. 2013, Mounce et al.  
546 2014) as well as predicted trends in annual fecundity, female survivorship and juvenile  
547 survivorship increased by 5%, 10% and 20%. These changed demographics were examined  
548 based on potential benefits that the leeward mesic habitat may have for the species. These

549 potential increased Maui parrotbill vital rates are not outside what has been estimated for other  
550 honeycreeper species, including other Maui endemics (Woodworth and Pratt 2009). The exact  
551 limitations of the wetter windward habitats are unknown but Maui parrotbills in the mesic forest  
552 may have increased nest success, increased foraging success, or both in the drier habitat. Maui  
553 parrotbills may also have reduced predation pressure in a habitat with lower invasive mammal  
554 densities (HL Mounce, personal observation). No Maui parrotbills currently occupy koa-  
555 dominated habitats, thus it is impossible to predict if the demography of released birds and their  
556 offspring will differ from that of the windward population. Our results demonstrate that the  
557 persistence of the reintroduced population is largely predicated on there being an increase in key  
558 demographic parameters in the new and potentially favorable environment. Given the importance  
559 of higher demographic rates for a new leeward population, a reintroduction strategy that includes  
560 an adaptive management plan is likely the most successful approach, where elevated parameter  
561 values in the leeward release sites serve as an alternative management hypothesis that can be  
562 evaluated through management actions and system monitoring (Williams et al. 2007, Reference  
563 S3). Managers could use the demographic parameter values we present here as benchmarks to  
564 strive for in future populations to ensure that the populations are successful and viable.

565

566

### **Supplemental Materials**

567

568 **Text S1.** The code for the custom-made model in Program R for the Maui parrotbill  
569 557 (*Pseudonestor xanthophrys*) population viability analyses conducted. This is a female-only,  
570 558 stochastic model assuming multiple, isolated populations. Notes are indicated with the  
571 pound 559 symbol (#) and Program R will not read these as part of the code.

572 Found at DOI: <http://dx.doi.org/10.3996/072017-JFWM-059.S1> (11 KB DOCX).

573

574 **Reference S1.** Regehr EV, Wilson RR, Rode KD, Runge MC. 2015. Resilience and risk: a  
575 demographic model to inform conservation planning for polar bears. Reston, VA: U.S.  
576 Geological Survey, Open-File Report 2015-1029:1-56.

577 Found at DOI: <http://dx.doi.org/10.3996/072017-JFWM-059.S2> (2005 KB PDF).

578

579 **Reference S2.** US Fish and Wildlife Service. 2006. Revised recovery plan for the Hawaiian  
580 forest birds. Portland, OR: Region 1. Of particular pertinence is the specific recovery plans and  
581 actions for Maui Parrotbills; Part II, pages 77-85.

582 Found at DOI: <http://dx.doi.org/10.3996/072017-JFWM-059.S3> (6 KB PDF).

583

584 **Reference S3.** Williams BK, Szaro RC, Shapiro CD. 2007. Adaptive Management: the US  
585 Department of the Interior Technical Guide. Washington, D.C.: US Department of the Interior,  
586 Adaptive Management Working Group.

587 Found at DOI: <http://dx.doi.org/10.3996/072017-JFWM-059.S4> (38,192 KB PDF).

588

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603

604

### References

605 Akçakaya HR, Sjogren-Gulve P. 2000. Population viability analyses in conservation planning: an  
606 overview. Ecological Bulletins 48:9-21

607 Banko, PC, Brinck KW, Farmer C, Hess SC. 2009. Palila. Pages 513-532 in Pratt TK, Atkinson  
608 CT, Banko PC, Jacobi JD, Woodworth B, editors. Conservation biology of Hawaiian forest  
609 birds: implications for island avifauna. New Haven, CT: Yale University Press

610 Becker CD., Mounce HL, Rassmussen TA, Rauch-Sasseen A, Swinnerton KJ, Leonard DL.  
611 2010. Nest success and parental investment in endangered Maui parrotbill (*Pseudonestor*  
612 *xanthophrys*) with impacts for recovery. Endangered Species Research 278:189-194

613 Beissinger SR. 2002. Population viability analysis: past, present, and future. Pages 5-17 in  
614 Beissinger SR, McCullough DR, editors. Population viability analysis. Chicago: University  
615 of Chicago Press

616 Benning TL, LaPointe D, Atkinson CT, Vitousek PM. 2002. Interactions of climate change with  
617 biological invasions and land use in the Hawaiian Islands: modeling the fate of endemic birds  
618 using a geographical information system. Proceedings of the National Academy of Sciences  
619 of the United States of America 99:14246-14249

620 Birdlife International. 2014. IUCN red list for birds. Available: <http://www.birdlife.org> (March  
621 2018)

622 Bonnot TW, Thompson FR, Millspaugh JJ, Jones-Farrand DT. 2013. Landscape-based  
623 population viability models demonstrate importance of strategic conservation planning for  
624 birds. Biological Conservation 165:104-114

625 Camp RJ, Gorresen PM, Pratt TK, Woodworth BL. 2009. Population trends of native Hawaiian  
626 forest birds: 1976-2008: the data and statistical analyses. Hawaii Cooperative Studies Unit  
627 Technical Report HCSU-012. Available: <http://dspace.lib.hawaii.edu/handle/10790/2692>  
628 (March 2018)

629 Caughley G. 1994. Directions in conservation biology. Journal of Animal Ecology 63:215-244

630 Clark TW, Backhouse GN, Lacy RC. 1991. The population viability assessment workshop: A  
631 tool for threatened species management. Endangered Species Update 8:1-5

632 Converse SJ, Armstrong DP. 2016. Demographic modeling for reintroduction decision-making.  
633 Pages 123-395 in Jachowski DS, Millspaugh JJ, Angermeier PL, Slotow R, editors.  
634 Reintroduction of Fish and Wildlife Populations. Oakland: University of California Press

635 Converse SJ, Moore CT, Armstrong DP. 2013. Demographics of reintroduced populations:  
636 estimation, modeling, and decision analysis. *The Journal of Wildlife Management* 77:1081-  
637 1093

638 Cook CN, Carter RW, Fuller RA, Hockings M. 2012. Managers consider multiple lines of  
639 evidence important for biodiversity management decisions. *Journal of Environmental*  
640 *Management* 113:341-346

641 Curio E. 1996. Conservation needs ethology. *Trends in Ecology and Evolution* 11:260–263

642 Ellner SP, Fieberg J. 2003. Using PVA for management despite uncertainty: effects of habitat,  
643 hatcheries, and harvest on salmon. *Ecology* 84:1359-1369

644 [ESA] US Endangered Species Act of 1973, as amended, Pub. L. No. 93-205, 87 Stat. 884 (Dec.  
645 28, 1973). Available at: <http://www.fws.gov/endangered/esa-library/pdf/ESAall.pdf> (March  
646 2018)

647 Evans DM, Che-Castaldo JP, Crouse D, Davis FW, Epanchin-Niell R, Flather CH, Frohlich RK,  
648 Goble DD, Li YW, Male TD, Master LL, Moskwik MP, Neel MC, Noon BR, Parmesan C,  
649 Schwartz MW, Scott JM, Williams BK. 2016. Species recovery in the United States:  
650 Increasing the effectiveness of the endangered species act. *Issues in Ecology* 20:1-29

651 Fischer J, Lindenmayer DB. 2000. An assessment of the published results of animal relocations.  
652 *Biological Conservation* 96:1-11

653 Foose TJ. 1993. Riders of the last ark: the role of captive breeding in conservation strategies.  
654 Pages 149-178 in Kaufman L, Mallory K, editors. *The last extinction*. Cambridge, MA: MIT  
655 Press and New England Aquarium.

656 Giambelluca TW, Diaz HF, Luke MSA. 2008. Secular temperature changes in Hawaii.  
657 *Geophysical Research Letters* 35:L12702

658 Gilpin ME., Soulé ME. 1986. Minimum viable populations: processes of species  
659 Extinction. Pages 19-34 in Soulé ME, editor. Conservation biology: the science of scarcity  
660 and diversity. Sunderland, MA: Sinauer Associates

661 Harvell CD, Mitchell CE, Ward JR, Altizer S, Dobson AP, Ostfeld RS, Samuel MD. 2002.  
662 Climate warming and disease risks for terrestrial and marine biota. *Science* 296:2156-2162

663 [IUCN]. International Union for Conservation of Nature. 2012. IUCN Red List of Threatened  
664 Species. Available: <http://www.iucnredlist.org> (March 2018)

665 James HF, Olson SL. 1991. Descriptions of thirty-two new species of birds from the Hawaiian  
666 Islands: part II. Passeriformes. *Ornithological Monographs* 46:1-88

667 Jule KR, Leaver LA, Lea SEG. 2008. The effects of captive experience in carnivores: a review  
668 and analysis. *Biological Conservation* 141:355–363

669 Lacy RC, Pollak JP. 2014. Vortex: A stochastic simulation of the extinction process. Version  
670 10.0. Brookfield, IL: Chicago Zoological Society

671 Leonard DL. 2008. Recovery expenditures for birds listed under the U.S. Endangered Species  
672 Act: the disparity between mainland and Hawaiian taxa. *Biological Conservation* 141:2054-  
673 2061

674 McGowan CP, Allan N, Servoss J, Hedwall S, Wooldridge B. 2017. Incorporating population  
675 viability models into species status assessment and listing decisions under the US  
676 Endangered Species Act. *Global Ecology and Conservation* 12:119-30.

677 McGowan CP, Catlin DH, Shaffer TL, Gratto-Trevor CL, Aron C. 2014. Establishing  
678 endangered species recovery criteria using predictive simulation modeling. *Biological*  
679 *Conservation* 177:220-229

680 McGowan CP, Ryan MR. 2009. A quantitative framework to evaluate incidental take and  
681 endangered species population viability. *Biological Conservation* 142:3128-3136

682 McGowan CP, Ryan MR, Runge MC, Millsbaugh JJ, Cochrane JF. 2011a. The role of  
683 demographic compensation theory in incidental take assessments for endangered species.  
684 *Biological Conservation* 144:730-737

685 McGowan CP, Runge MC, Larson MA. 2011b. Incorporating parametric uncertainty into  
686 population viability analysis models. *Biological Conservation* 144:1400-8.

687 Mills LS. 2007. Conservation of wildlife populations: demographics, genetics, and management.  
688 2<sup>nd</sup> edition. Malden, MA: Blackwell Publishing

689 Morris W, Doak D. 2002. Quantitative conservation biology: theory and practice of population  
690 viability analysis. Sunderland, MA: Sinauer Associates

691 Mounce HL, Raisin C, Leonard DL, Wickenden H, Swinnerton KJ, Groombridge JJ. 2015.  
692 Spatial genetic architecture of the critically-endangered Maui parrotbill (*Pseudonestor*  
693 *xanthophrys*): management considerations for reintroduction strategies. *Conservation*  
694 *Genetics* 16:71-84

695 Mounce HL, Iknayan KJ, Leonard DL, Swinnerton KJ, Groombridge JJ. 2014. Management  
696 implications derived from long term re-sight data: annual survival of the Maui parrotbill  
697 *Pseudonestor xanthophrys*. *Bird Conservation International* 24:316-326

698 Mounce HL, Leonard DL, Swinnerton KJ, Becker CD, Berthold LK, Iknayan KJ, Groombridge  
699 JJ. 2013. Determining productivity of Maui parrotbills, an endangered Hawaiian  
700 honeycreeper. *Journal of Field Ornithology* 84:32-39

701 Mountainspring S. 1987. Ecology, behavior, and conservation of the Maui parrotbill. *The*  
702 *Condor* 89:24-39



703 Pe'er G, Matsinos YG, Johst K, Franz KW, Turlure C, Radchuk V, Malinowska AH, Curtis JMR,  
704 Naujokaitis-Lewis I, Wintle BA, Henle K. 2013. A protocol for better design, application,  
705 and communication of population viability analyses. *Conservation Biology* 27:644-656

706 Perkins RCL. 1903. Vertebrata. Pages 365-466 in Sharp D, editor. *Fauna Hawaiiensis*.  
707 Cambridge, UK: Cambridge University Press

708 Pratt, TK, Fancy SG, Ralph CJ. 2001. Akiapolaau (*Hemignathus wilsoni*), version 2.0. In *The*  
709 *Birds of North America* (P. G. Rodewald, editor). Cornell Lab of Ornithology, Ithaca, New  
710 York, USA. Accessed from the Birds of North America: [https://birdsna.org/Species-](https://birdsna.org/Species-Account/bna/species/akiapo)  
711 [Account/bna/species/akiapo](https://birdsna.org/Species-Account/bna/species/akiapo) (March 2018)

712 R Core Team. 2017. R: A language and environment for statistical computing. Vienna, Austria:  
713 R Foundation for Statistical Computing

714 Ralls K, Beissinger SR, Cochrane JF. 2002. Guidelines for using PVA in endangered-species  
715 management. Pages 521-550 in Beissinger SR, McCullough DR, editors. *Population viability*  
716 *analysis*. Chicago: University of Chicago Press

717 Rantanen EM, Buner F, Riordan P, Sotherton N, Macdonald DW. 2010. Habitat preferences and  
718 survival in wildlife reintroductions: an ecological trap in reintroduced grey partridges.  
719 *Journal of Applied Ecology* 47:1357-1364.

720 Reed JM, Mills LS, Dunning JB, Menges ES, McKelvey KS, Frye R, Beissinger SR, Anstett  
721 MC, Miller P. 2002. Emerging issues in population viability analysis. *Conservation Biology*  
722 16:7-19

723 Regehr EV, Wilson RR, Rode KD, Runge MC. 2015. Resilience and risk: a demographic model  
724 to inform conservation planning for polar bears. Reston, VA: U.S. Geological Survey, Open-

725 File Report 2015-1029:1-56. Available <https://pubs.usgs.gov/of/2015/1029/pdf/ofr2015->  
726 1029.pdf (March 2018). Reference S1

727 Robinson OJ, McGowan CP, Apodaca JJ. 2016. Decision analysis for habitat conservation of an  
728 endangered, range-limited salamander. *Animal Conservation* 19:561-569

729 Runge MC. 2011. Adaptive management for threatened and endangered species. *Journal of Fish*  
730 *and Wildlife Management* 2:220–233

731 Rushing CS, Hostetler JA, Sillett TS, Marra PP, Rotenberg JA, Ryder TB. 2017. Spatial and  
732 temporal drivers of avian population dynamics across the annual cycle. *Ecology* 98:2837-  
733 2850

734 Scott JM, Mountainspring S, Ramsey FL, Kepler CB. 1986. Forest bird communities of the  
735 Hawaiian Islands: their dynamics, ecology, and conservation. *Studies in Avian Biology* 9:i-  
736 xii

737 Shaffer ML. 1981. Minimum population sizes for species conservation. *BioScience* 31:131-134

738 Shaffer ML. 1987. Minimum viable populations: coping with uncertainty. Pages 69-86 in Soulé  
739 ME, editor. *Viable populations for conservation*. New York: Cambridge University Press

740 Schaub M, Abadi F. 2011. Integrated population models: a novel analysis framework for deeper  
741 insights into population dynamics. *Journal of Ornithology* 152: 227-237

742 Simon JC, Pratt TK, Berlin KE, Kowalsky JR. 2000. Reproductive ecology of the Maui  
743 parrotbill. *Wilson Bulletin* 112:482-490

744 Simon JC, Baker PE, Baker H. 1997. Maui parrotbill (*Pseudonestor xanthophrys*). Pages 1-16 in  
745 Poole A, Gill F, editors. *The birds of North America*, number 311. Philadelphia and  
746 Washington, D.C.: Academy of Natural Sciences and American Ornithologists Union

747 Soulé ME, Mills LS. 1998. No need to isolate genetics. *Science* 282:1658-1659

748 Stamps JA, Swaisgood RR. 2007. Someplace like home: experience, habitat  
749 selection and conservation biology. *Applied Animal Behaviour Science* 102:392–409

750 Steadman DW. 2006. *Extinction and biogeography of tropical Pacific birds*. Chicago: University  
751 of Chicago Press

752 Stubben CJ, Milligan BG. 2007. Estimating and analyzing demographic models using the popbio  
753 package in R. *Journal of Statistical Software* 22:11

754 [USFWS] US Fish and Wildlife Service. 1967. Office of the Secretary; Native Fish and Wildlife;  
755 Endangered Species. *Federal Register* 32:4001

756 [USFWS] US Fish and Wildlife Service. 2006. Revised recovery plan for the Hawaiian forest  
757 birds. Portland, OR: Region 1. Available:  
758 <https://www.fws.gov/pacificislands/recoveryplans.html> (March 2018). Reference S2

759 Warren CC, Motyka PJ, Mounce HL. 2015. Home-range patterns of two Hawaiian  
760 Honeycreepers: implications for proposed translocation efforts. *Journal of Field Ornithology*  
761 86:305-316

762 Williams BK, Szaro RC, Shapiro CD. 2007. *Adaptive Management: the US Department of the*  
763 *Interior Technical Guide*. Washington, D.C.: US Department of the Interior, Adaptive  
764 Management Working Group. Available: [https://www2.usgs.gov/sdc/doc/DOI-](https://www2.usgs.gov/sdc/doc/DOI-%20Adaptive%20ManagementTechGuide.pdf)  
765 [%20Adaptive%20ManagementTechGuide.pdf](https://www2.usgs.gov/sdc/doc/DOI-%20Adaptive%20ManagementTechGuide.pdf) (March 2018). Reference S3

766 Woodworth BL, Pratt TK. 2009. Life history and demography. Pages 194-233 in Pratt TK,  
767 Atkinson CT, Banko PC, Jacobi JD, Woodworth B, editors. *Conservation biology of*  
768 *Hawaiian forest birds: implications for island avifauna*. New Haven: Yale University Press.

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770

771 **FIGURE LEGENDS**

772

773 **Figure 1.** Female Maui parrotbill *Pseudonestor xanthophrys*. Photo taken on 6 May 2017 in The Nature  
774 Conservancy’s Waikamoi Preserve, Maui, HI by Zach Pezzillo used with permission by Maui Forest Bird  
775 Recovery Project.

776

777 **Figure 2.** Map of land protections that benefit native forest birds in east Maui, HI (Haleakala  
778 Volcano) and the Maui parrotbill (*Pseudonestor xanthophrys*) range. The Maui parrotbill range  
779 overlays the windward (northeast) reserves, Hanawi Natural Area Reserve and The Nature  
780 Conservancy’s Waikamoi Preserve. The reserve where Maui parrotbills will be reintroduced,  
781 Nakula NAR, is shown on the leeward (southern) slope.

782

783 **Figure 3.** Projected mean final female population sizes ( $N$ -all) for Maui parrotbill (*Pseudonestor*  
784 *xanthophrys*) under base and modified base models in R 3.4.2. Solid black line represents the  
785 “Base Model Population Metapopulation”, solid black line with dots represents the “Modified  
786 Base Model Metapopulation”, and broken gray line with box represents the “Modified Base  
787 Model East Population”, and dotted gray line with diamond represents the “Modified Base  
788 Model West Population”. Population projections are presented for the East population (i.e.,  
789 Hanawi Natural Area Reserve), West (i.e., The Nature Conservancy’s Waikamoi Preserve), and  
790 the Metapopulation (i.e., East and West combined).

791

792 **Figure 4.** Female Maui parrotbill (*Pseudonestor xanthophrys*) population trajectories for the  
793 three existing populations (A- East [Hanawi Natural Area Reserve], B- West [The Nature  
794 Conservancy’s Waikamoi Preserve], and D- Captive [San Diego Zoo Global facilities]) and the

795 proposed reintroduced leeward population (C- Leeward [Nakula Natural Area Reserve]).  
796 Population estimates for (A), (B), (C), and (D) are based on a proposed three-year reintroduction  
797 scenario wherein the captive flock is augmented to source 1 female/year in combination with  
798 translocations from existing wild populations (scenario *iv*). Demographic parameters for wild  
799 populations are set to values from Mounce et al. (2013, 2014). Solid lines indicate mean number  
800 of adult females in the population from 1000 model runs. Dashed lines indicate 95% CI around  
801 mean values.

802  
803 **Figure 5.** Maui parrotbill (*Pseudonestor xanthophrys*) population trajectories for the proposed  
804 future reintroduced leeward population (Nakula Natural Area Reserve) based on a realistic  
805 proposed reintroduction scenario (scenario *iv*, wherein the captive flock sources 1 female/year  
806 and additional translocations from wild populations [Hanawi Natural Area Reserve and The  
807 Nature Conservancy's Waikamoi Preserve]). Panel A demonstrates a population trajectory  
808 predicting trends in the reintroduced population with annual fecundity, female survivorship, and  
809 young of the year set as in Mounce et al. (2013, 2014; A). The other panels demonstrate  
810 trajectories for the same population with parameters increased by 10% (B), 15% (C), and 20%  
811 (D) based on potential benefits of the leeward mesic habitat. Solid lines indicate mean number of  
812 adult females in the population from 1000 model runs. Dashed lines indicate 95% CI around  
813 mean values.

814 **Table 1.** Parameter input values for the base and modified base PVA model used for Maui  
815 parrotbills (*Pseudonestor xanthophrys*). Input parameters derived from Mounce et al. (2013,  
816 2014, 2015), Warren et al. (2015), and unpublished data from Maui Forest Bird Recovery  
817 Project. Values in bold highlight changes between the Base and Modified Base models.  
818 Percentage of breeding females and survival rates are presented with estimates of environmental  
819 variation (EV;  $\pm$  SD).

	<b>Base Model</b>	<b>Modified Base Model</b>	
		East Pop (1)	West Pop (2)
Dispersal	No	<b>Yes</b>	<b>Yes</b>
Age range of dispersers	.	<b>0-1</b>	<b>0-1</b>
% survival of dispersers	.	<b>40-90</b>	<b>40-90</b>
Mean % dispersing between pops	.	<b>2</b>	<b>2</b>
Age of 1st breeding	2	2	2
% adult females breeding (EV)	46 ( $\pm$ 0.25)	<b>56 (<math>\pm</math> 0.25)</b>	<b>56 (<math>\pm</math> 0.25)</b>
% 1 offspring (% 2 offspring)	95 (5)	95 (5)	95 (5)
% survival rates 0-1, $S_y$	17 ( $\pm$ 0.15)	<b>32 (<math>\pm</math> 0.02)</b>	<b>32 (<math>\pm</math> 0.02)</b>
% survival rates after age 1, $S_a$	72 ( $\pm$ 0.02)	72 ( $\pm$ 0.02)	72 ( $\pm$ 0.02)
Initial population size	292	239	53
Carrying capacity (K)	432	354	78
Future change in K?	No	<b>Yes</b>	<b>Yes</b>
% annual increase	.	<b>-1</b>	<b>-1</b>

820

821

822 **Table 2.** Population viability analysis model results for the base and modified base model for the  
823 Maui parrotbill (*Pseudonestor xanthophrys*) population(s) with the viability measures of  $\lambda$   
824 (median rate of population change), PQE (probability of quasi-extinction [ $N < 10$ ] at 25 years), N-  
825 all (median population size from all iterations at year 25), N-extant (median population size from  
826 extant populations at year 25). \* N-extant is defined as  $N > 10$ , thus in the base model with N-  
827 extant = 10, no SD can be calculated as the model considers the population extinct.

	Base model	Modified base model
$\lambda$	0.784	0.866
PQE	0.999	0.992
N-all	1	2
SD N-all	0.75	3.04
N-extant	10	12
SD N-extant	*	6.317

828

829 **Table 3.** Results of the sensitivity and elasticity analysis for the Maui parrotbill (*Pseudonestor*  
830 *xanthophrys*) meta-population based on parameter changes in the “modified base model.”  
831 Demographic parameters included were juvenile survival (% survival rates 0-1;  $S_y$ ), adult  
832 survivorship (% survival rates after age 1;  $S_a$ ), and fecundity (F). Lambda ( $\lambda$ ) for this model was  
833 0.866

	$S_y$	$S_a$	F
Sensitivity .	0.316	0.856	0.39
Elasticity .	0.144	0.712	0.144

834

835



836 **Table 4.** Reintroduction scenarios indicating the total number female Maui parrotbill (*Pseudonestor xanthophrys*) would be needed to  
837 move between the East (Hanawai Nat  
838 ural Area Reserve), West (The Nature Conservancy’s Waikamoi Preserve), Leeward (Nakula Natural Area Reserve), and Captive (San  
839 Diego Zoo Global) populations over three years. For each scenario are population viability analysis model results for the modified  
840 base model with the viability measures of PQE (probability of quasi-extinction [ $N < 10$ ] at 25 years), N-all (median population size  
841 from all iterations at year 25), N-extant (median population size from extant populations at year 25). (-) indicates N/A.

842

	Scenario <i>i</i>					Scenario <i>ii</i>			
	East	West	Leeward	Captive		East	West	Leeward	Captive
# to captivity	0	0	-	-	# to captivity	51	17	-	-
# to Leeward	0	0	-	7	# to Leeward	0	0	-	15
PQE	0.990	1.000	1.000	0.089	PQE	0.966	1.000	1.000	0.000
N-all	1	1	0	9	N-all	1	0	1	66
SD N-all	2.157	0.599	0.422	4.469	SD N-all	1.701	0.528	0.475	4.127
N-extant	14	-	-	12	N-extant	11	-	-	66
SD N-extant	1.805	-	-	2.36	SD N-extant	3.559	-	-	4.127

	Scenario <i>iii</i>					Scenario <i>iv</i>			
	East	West	Leeward	Captive		East	West	Leeward	Captive
# to captivity	23	8	-	-	# to captivity	6	2	-	-
# to Leeward	4.5	3	-	7.5	# to Leeward	7.5	4.5	-	3
PQE	0.991	1.000	1.000	0.000	PQE	0.988	1.000	1.000	0.000
N-all	1	1	0	32	N-all	1	1	0	18
SD N-all	1.936	0.555	0.483	4.263	SD N-all	1.884	0.556	0.477	4.230
N-extant	11	-	-	32	N-extant	10	-	-	18

SD N-extant	1.59	-	-	4.263
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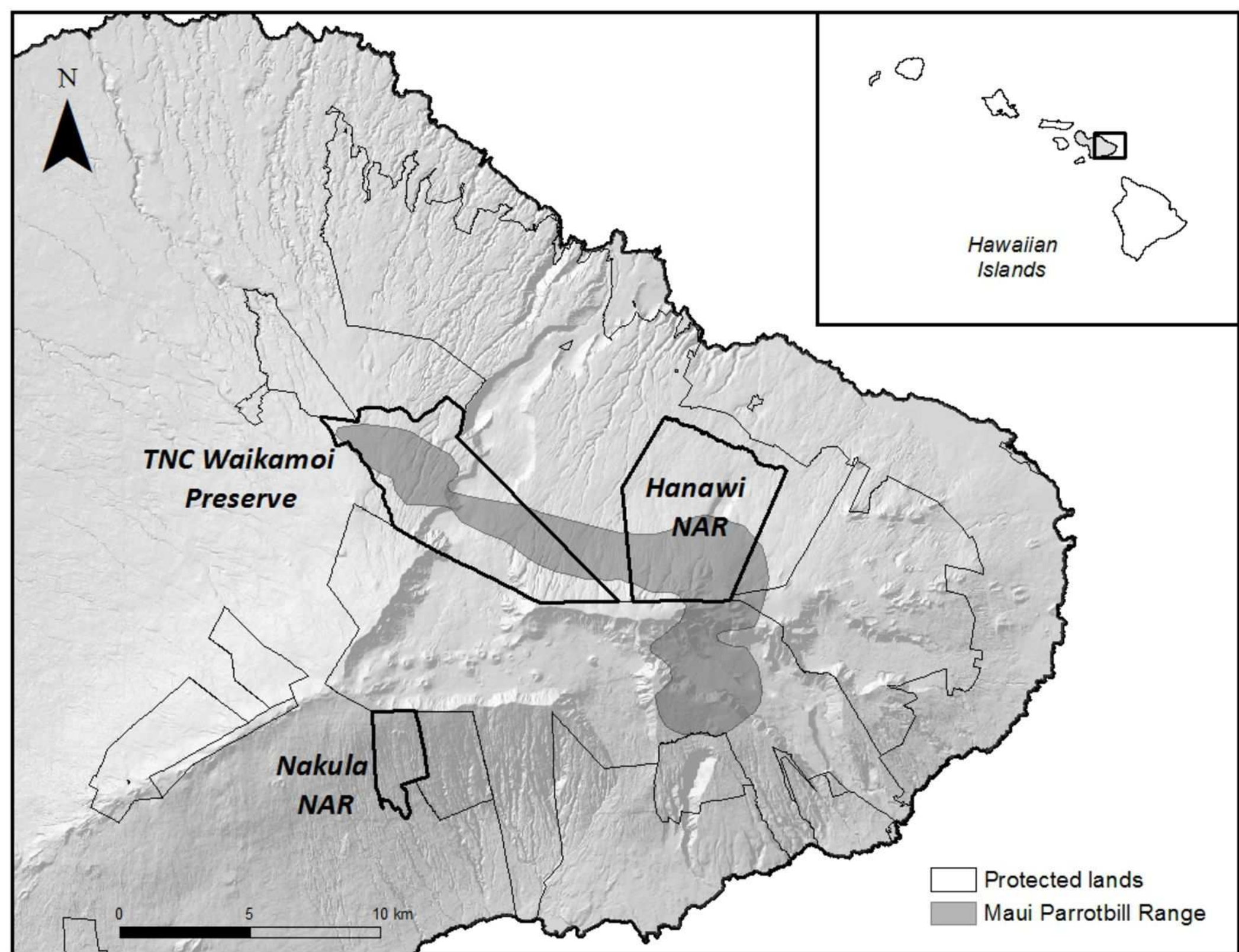
SD N-extant	2.348	-	-	3.972
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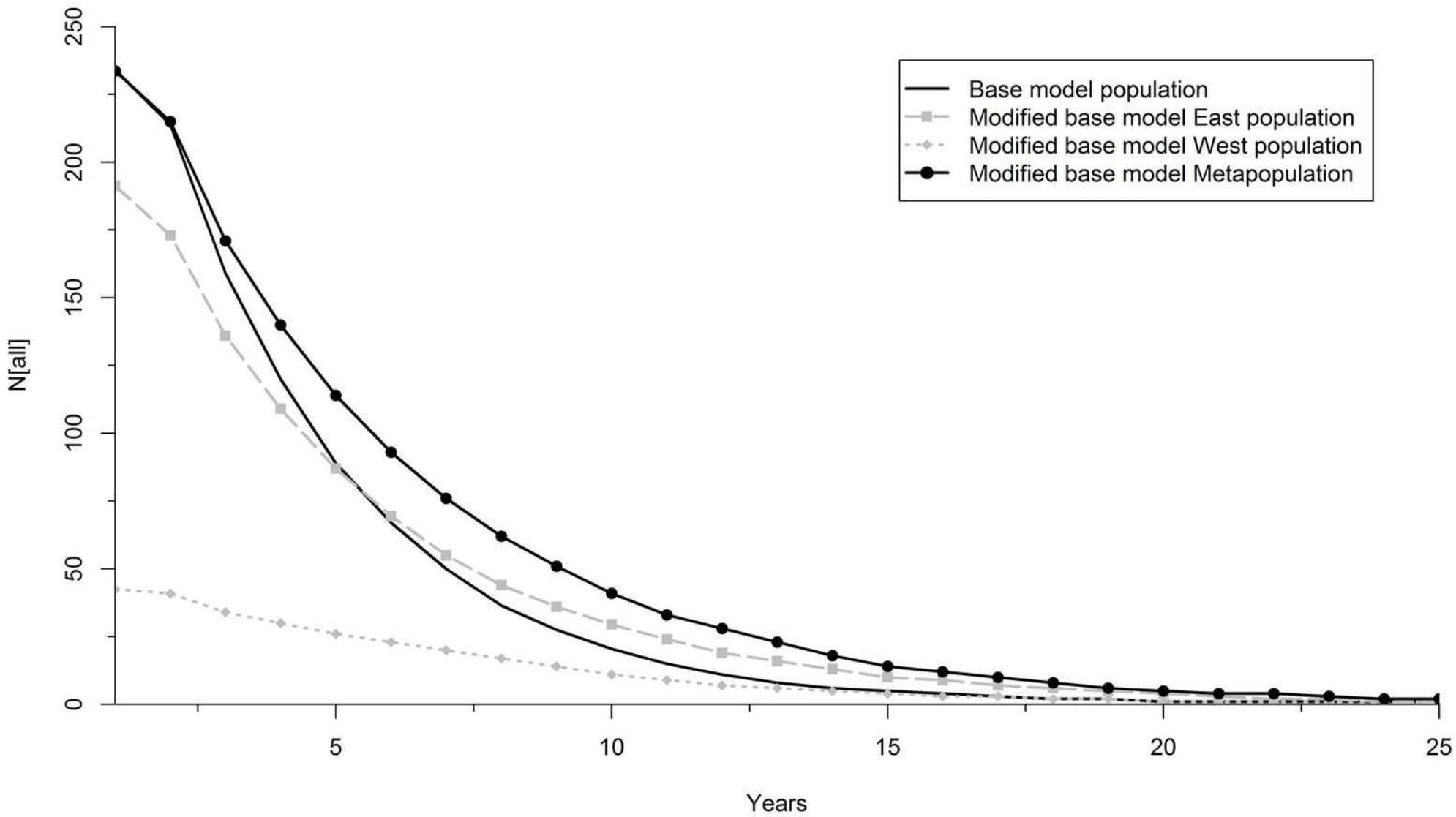
Scenario <i>v</i>				
	East	West	Leeward	Captive
# to captivity	0	0	-	-
# to Leeward	4.5	3	-	7.5
PQE	0.989	1.000	1.000	0.045
N-all	1	1	0	10
SD N-all	2.018	0.563	0.455	4.443
N-extant	12	-	-	12
SD N-extant	1.859	-	-	2.762

Scenario <i>vi</i>				
	East	West	Leeward	Captive
# to captivity	0	0	-	-
# to Leeward	9	6	-	0
PQE	0.980	1.000	1.000	0.006
N-all	1	1	0	13
SD N-all	2.367	0.640	0.465	4.241
N-extant	12	-	-	14
SD N-extant	3.832	-	-	3.311

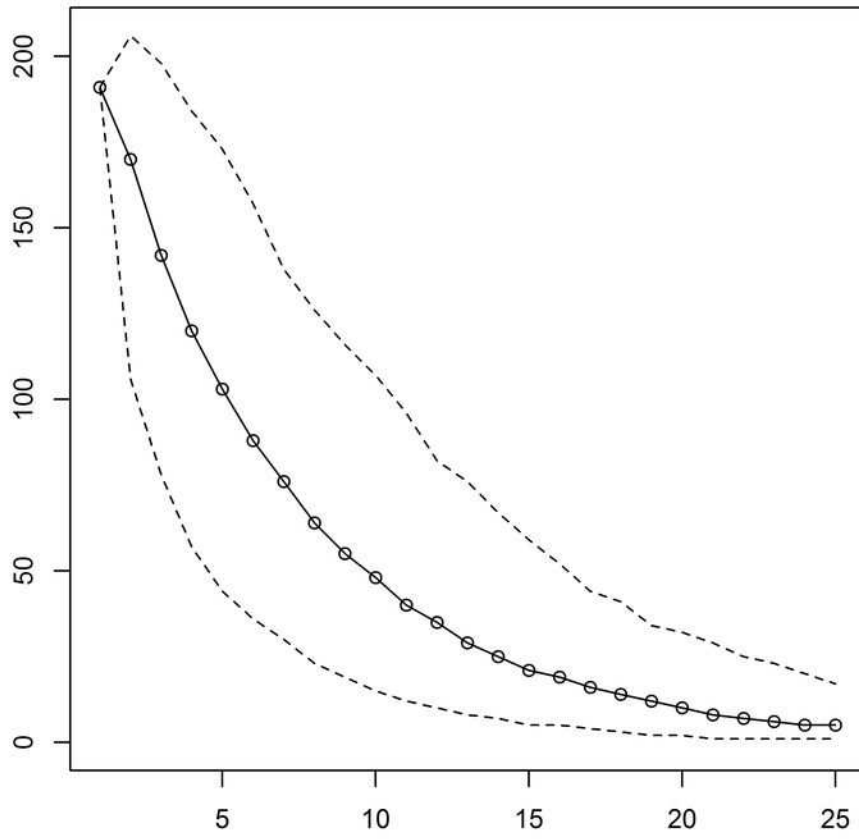
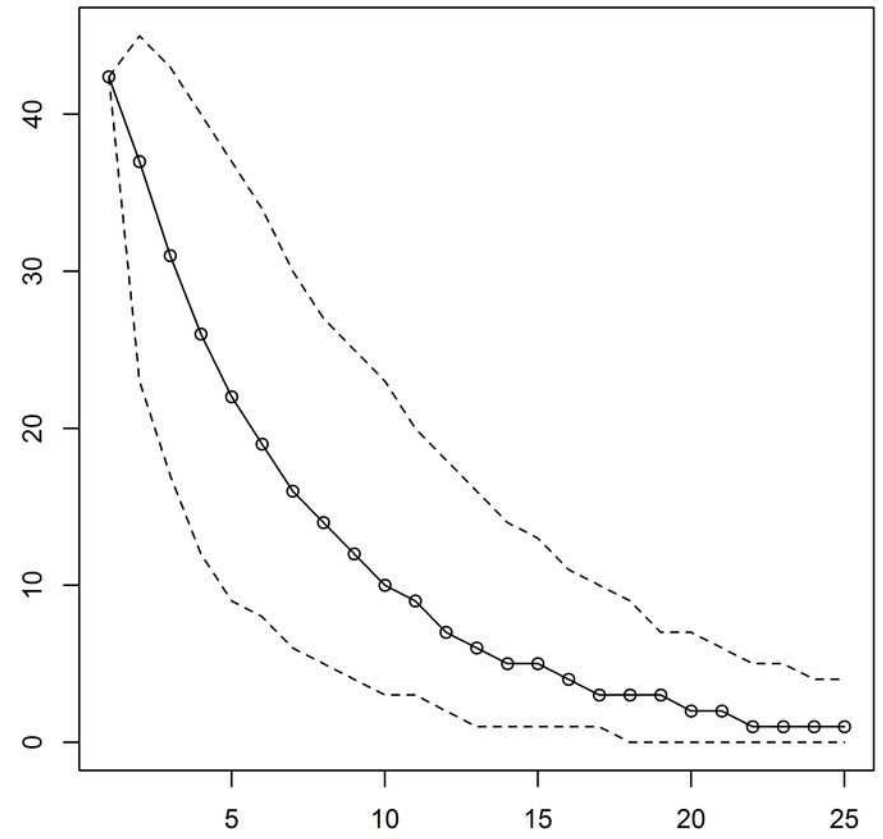
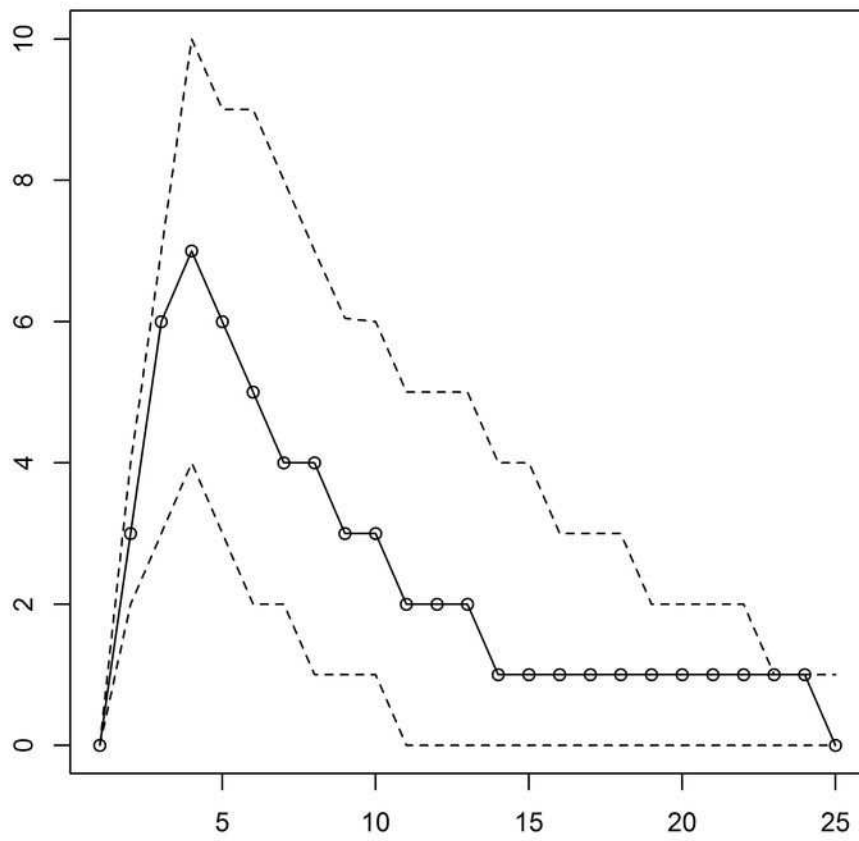
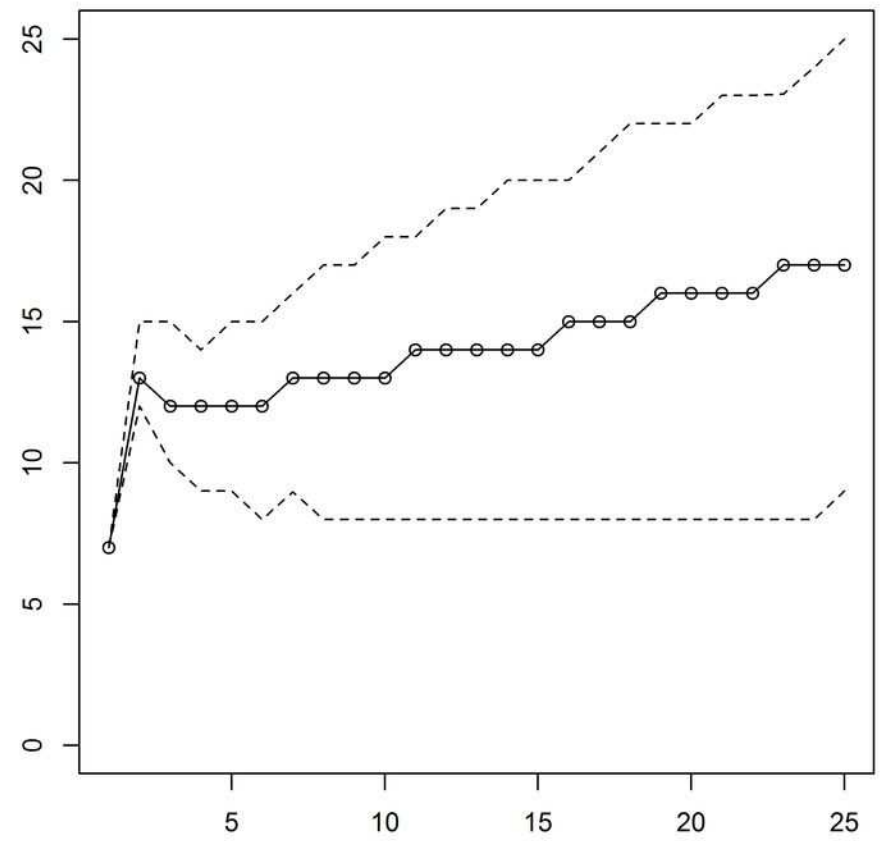
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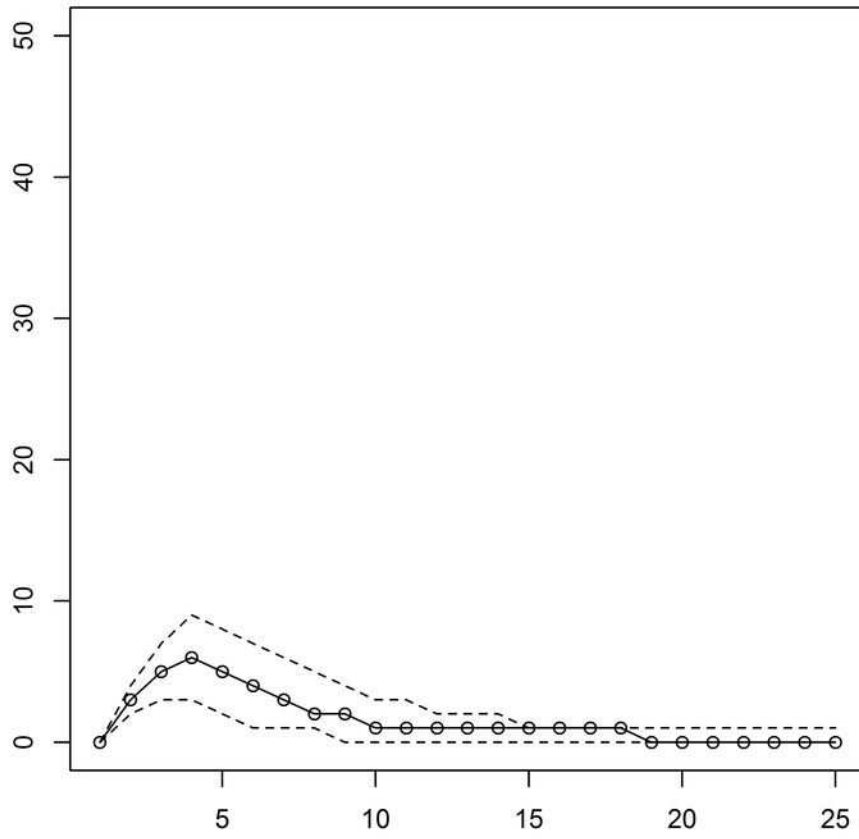
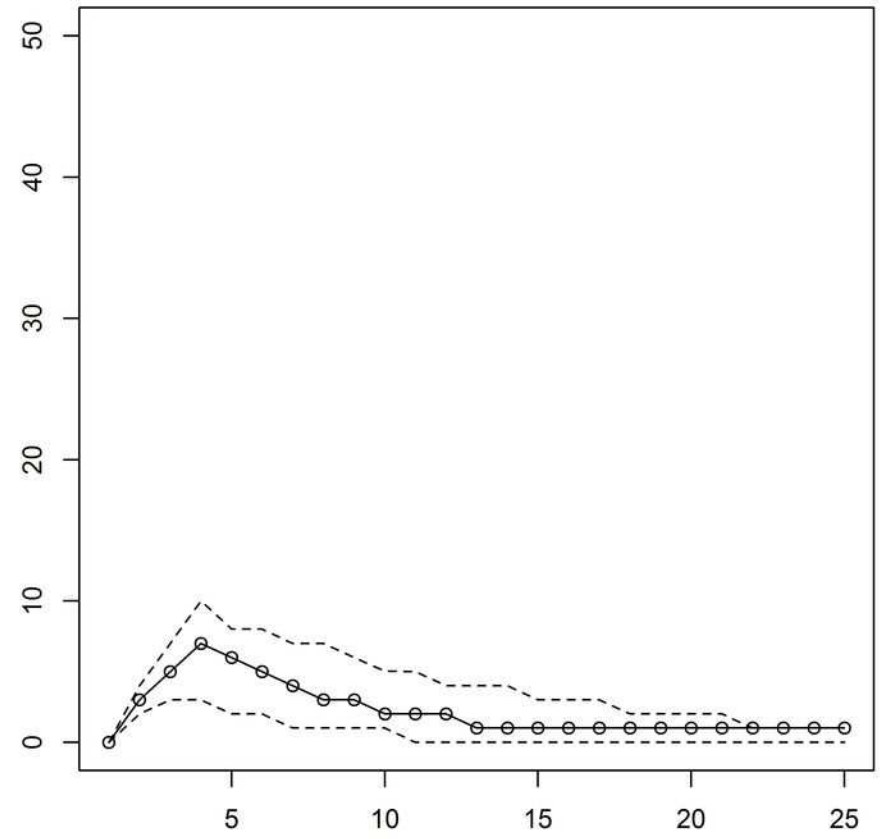
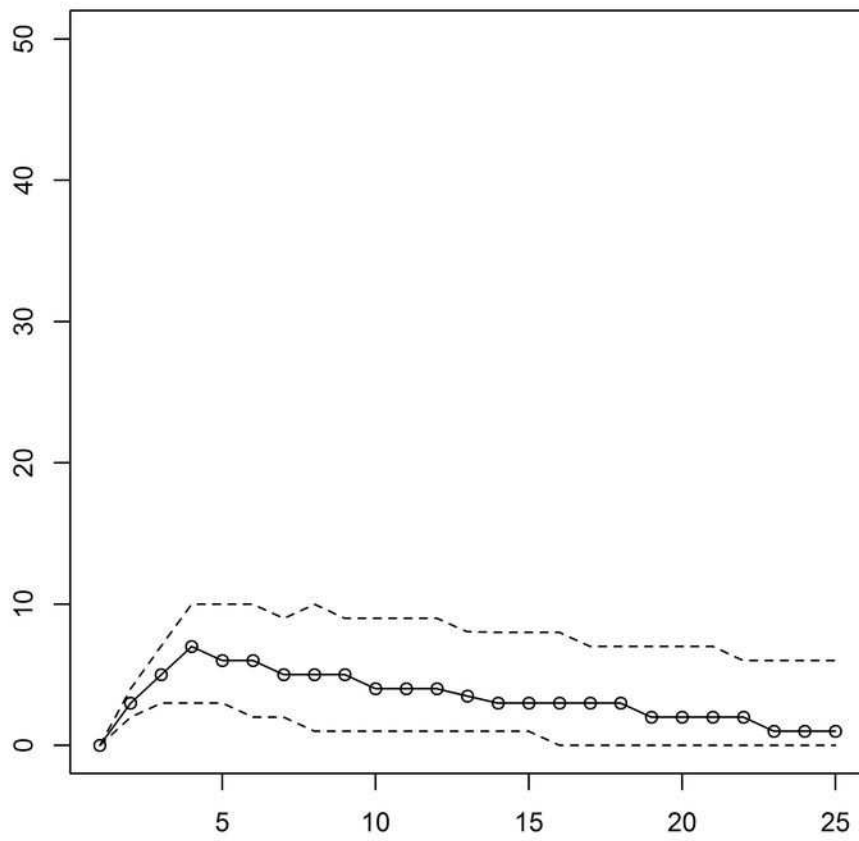
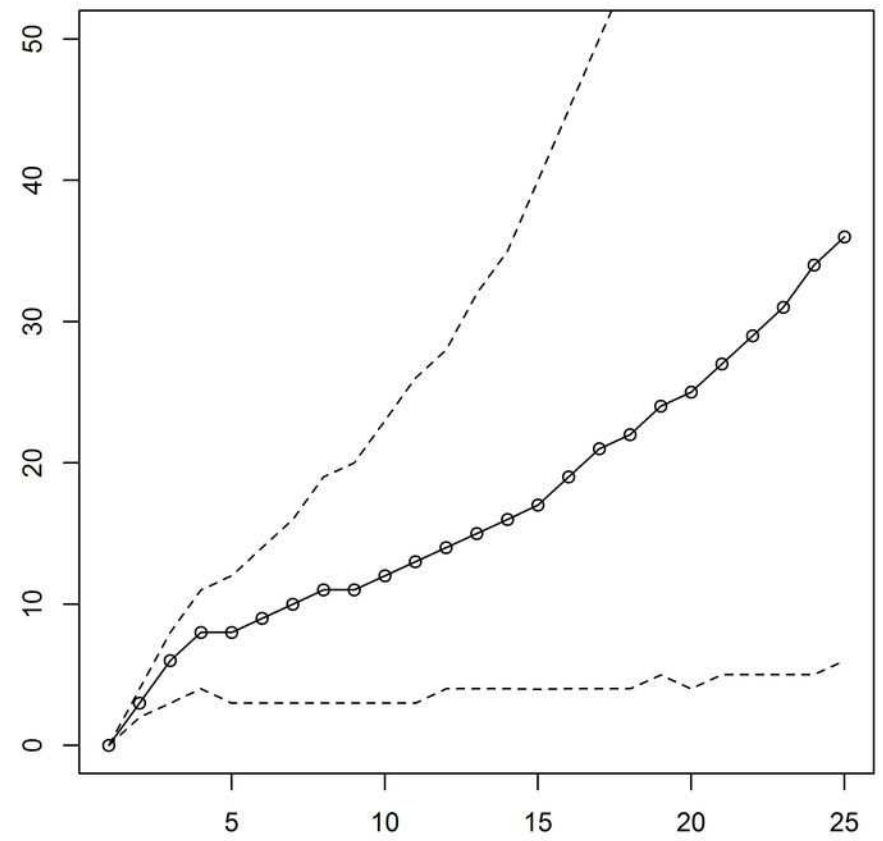








**A) East****B) West****C) Leeward****D) Captive****Year**

**A) +0%****B) +5%****C) +10%****C) +20%****Year**