

1 **REVIEW**

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3 **Conservation Translocations in Aotearoa New Zealand in the Predator-Free era**

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20 **Running head:** Conservation translocations in Aotearoa NZ

21

22 **Abstract**

23

24 The biological changes that have occurred in Aotearoa New Zealand following human settlement are
25 well documented with almost all ecosystems and taxa having been negatively impacted. Against this
26 background of loss there have been remarkable advances in conservation management, particularly in
27 the large-scale eradication and control of exotic mammalian pests. In 2016, the New Zealand
28 Government announced Predator Free 2050, an ambitious project to eradicate introduced predators in
29 Aotearoa New Zealand by 2050. Here, we discuss conservation translocations in the context of
30 Predator Free 2050 aspirations. Our review draws together knowledge from Aotearoa New Zealand's
31 rich history of translocations and outlines a framework to support translocation decision making in the
32 predator-free era. Predator Free 2050 aspirations encompass an ongoing question in conservation
33 management; should we focus on maintaining small protected populations, because this seems
34 generally easier and currently achievable, or on reversing declines in the large mainland areas that
35 contain most of our biodiversity, a much harder challenge largely reliant on the continued use of
36 aerially applied toxins? We focus on successfully establishing small translocated populations because
37 they will provide the source populations for colonisation of a predator-free landscape. We define a
38 successful translocation as one that meets a clear set of fundamental objectives defined a priori. If
39 translocation objectives are clearly defined all subsequent decisions about factors that influence
40 conservation translocation outcomes (e.g. the cultural and social setting, pest thresholds, habitat
41 quality, genetic management) will be easier. Therefore, we encourage careful thinking in formulating
42 conservation translocation objectives that align with aspirations for a predator-free Aotearoa NZ. We
43 discourage a focus on any single element of planning and rather encourage all people involved in
44 conservation translocations, particularly decision makers, to explicitly recognise the multiple values-
45 based objectives associated with conservation translocations.

46

47 **Keywords:** Conservation translocation, reintroduction, restoration, Predator Free 2050

48

49 Introduction

50

51 The biological changes that have occurred in Aotearoa New Zealand (NZ) following two waves of
52 human settlement are well documented, with almost all ecosystems and taxa having been negatively
53 impacted (Caughley 1989; Holdaway 1989). For example c. 50% of all native bird species have
54 become extinct since first human contact (Caughley 1989; Holdaway 1989), and the remaining
55 species show varying levels of vulnerability to exotic predators (Innes et al. 2010). This history of
56 extinction and drastic reduction in population size and range is neatly captured in Māori whakataukī
57 (proverbs) including “*Ko te huna i te moa- destroyed like the moa*”, (Wehi et al. 2018) or by Diamond
58 (1984) who stated that “*New Zealand doesn’t have an avifauna, just the wreckage of one*”.

59

60 Against this background of loss there have been remarkable advances in conservation management,
61 particularly in large-scale pest eradication and control (pest, as used here, primarily refers to exotic
62 mammalian predators and competitors but also includes other unwanted harmful vertebrates,
63 invertebrates, plants and pathogens). Multi-species eradications have been completed on several large
64 islands (Towns & Broome 2003) and there are plans in place to tackle ever larger islands (e.g.
65 Auckland Island at 51 000ha). Many fenced mainland reserves offer island-like conditions on the
66 mainland in that they are often isolated from other indigenous habitats and most significant pests are
67 absent most of the time (Innes et al. 2019). The number of unfenced mainland sites under varying
68 forms of protection is also increasing every year (Innes et al. 2019). There was considerable
69 excitement - and scepticism - around the NZ Government’s 2016 announcement of Predator Free
70 2050. Regardless of whether this is an achievable goal in the next three decades it is likely to lead to
71 an increase in control of some pests (especially rats (*Rattus* spp.), stoats (*Mustela erminea*) and
72 possums (*Trichosurus vulpecula*)) and a pest landscape ranging from areas with complete eradication
73 through to areas with lower density pest levels than are currently present. Surprisingly, there has been
74 little detail about what a predator-free Aotearoa NZ might look like, but implicit is the goal of
75 exchanging pest biomass for native and endemic biomass. In short, a predator-free Aotearoa NZ has a
76 mix of fundamental objectives (what we really want) and means objectives (how we get what we
77 really want) including more native and endemic wildlife and fewer pests.

78

79 The first and most urgent means by which we can achieve this is to maintain and increase the
80 biodiversity we still have. We are very good at doing this on islands. However, we are also making
81 gains, at least for some forest birds, close to many urban areas, where growing community
82 conservation initiatives have led to the establishment of mainland ecological restoration projects
83 involving varying levels of pest control, planting, and conservation translocations. Many such projects
84 have been successful in achieving high-density populations of native and endemic wildlife, again with
85 an emphasis on forest birds. A critical limitation is that most of these restored sites are small (c.100-
86 1000ha), and mice (*Mus musculus*) have rarely been eradicated, or even sufficiently controlled, with
87 important implications for the recovery of endemic lizards, amphibians, invertebrates, bats, and
88 threatened plants. In contrast, the bulk of our biodiversity is contained within vast areas (1000s of
89 hectares) of back country conservation estate which are much harder to protect and harder for the
90 general public to engage with. The Department of Conservation (DOC) “Tiakina Ngā Manu/Battle for
91 our Birds” programme is achieving impressive pest control over huge areas of Aotearoa NZ forests (c.
92 1 million ha in 2019), operating in parallel with species-focussed mainland recovery programmes (e.g.
93 kakī/black stilt (*Himantopus novaeseelandiae*) and orange fronted kākārīki (*Cyanoramphus malherbi*)).
94 Nevertheless, vast tracts of land, especially non-forested habitats, remain unprotected, and
95 biodiversity continues to decline. This is reflected in the most recent NZ threat classification for birds
96 (Robertson et al. 2017) that has seen some species previously ranked “non-threatened” move to “at

97 risk, declining”, including popokatea/whiteheads (*Mohoua albigilla*), North Island (NI) and South
98 Island (SI) toutouwai/robins (*Petroica longipes* and *P. australis*), and NI and SI mātātā/fernbirds
99 (*Bowdleria punctata vealeae* and *B.p. punctata*).

100

101 The current situation on mainland Aotearoa NZ is neatly captured by Caughley’s (1994) small
102 population and declining population paradigms. Our small protected populations are subject to the
103 many risks of being small, for example pest incursions, dispersal, extreme weather events,
104 unpredictable stochastic events, novel pathogens, and loss of genetic diversity. In contrast, many of
105 our large mainland populations are declining because of the ongoing pervasive impacts of pests. The
106 current tension in NZ conservation management is one of maintaining small populations, because this
107 seems generally easier and currently achievable, versus securing the large mainland areas that contain
108 the bulk of our biodiversity, a much harder challenge largely reliant on the continued use of aerially
109 applied toxins. A predator-free Aotearoa NZ necessitates both of these approaches.

110

111 Small intensively protected populations provide insurance against further declines, and can serve as
112 source populations for colonisation of, or translocation to, the surrounding habitats when these come
113 under some form of pest control. The sites such populations occupy also provide a glimpse of what a
114 predator-free Aotearoa NZ might look like, and thus are critical tools for engaging the general public
115 in conservation management, whether as active participants or passive supporters (Parker 2008). In
116 contrast, ongoing pest control in large mainland areas is essential for protecting biodiversity not able
117 to be protected on islands, or in small intensively protected areas. When these large mainland areas
118 are released from the pervasive effects of pests (primarily a question of social license and technical
119 advance) they will further buffer threatened species against the challenges of being small.

120

121 In this paper we focus on small population management in Aotearoa NZ, particularly small
122 translocated populations that have been established at a site to compensate for local extinction,
123 although many of the same principles will apply also to small recovering relict populations. These
124 populations will be critical if a predator-free Aotearoa NZ becomes reality because they will be the
125 source populations that recolonise the pest-free landscape. It was recently suggested by the
126 Parliamentary Commissioner for the Environment that there is “*An urgent need for translocation
127 policy based on clear principles*” in Aotearoa NZ (Parliamentary Commissioner for the Environment
128 2017). This is an odd statement because our collective experience across many translocations is that
129 they are guided by very clear principles, although we agree that these principles are not currently
130 captured in DOC policy, which compromises the ability of DOC to assess the value of individual
131 translocation proposals. However, the DOC approval process, via the translocation proposal
132 document, captures many of the principles of sound conservation translocation practice, including
133 those described in the IUCN “*Guidelines for reintroductions and other conservation translocations*”
134 (IUCN 2013). Immediately following the call for a translocation policy, but still under the
135 translocation section of the Parliamentary Commissioner report, it was suggested that a recent work
136 on genetic management of fragmented populations by Frankham et al (2017) might provide a basis for
137 rethinking genetic management in NZ conservation. This seems to confuse genetic management with
138 translocation management. While Frankham et al (2017) is an excellent text providing valuable
139 information for translocation planning and management, genetic management is only one component
140 of a successful translocation and it is unhelpful to focus on just one aspect of a translocation when
141 there are so many other ways that translocations can fail, often long before genetic issues can become
142 problematic.

143

144 Here, we discuss conservation translocations in the context of the Predator Free 2050 aspirations. We
145 spend little time on the fraught task of predicting the future success of actually freeing Aotearoa NZ
146 of the target pests. Rather, we focus on what can be achieved today, with the available resources and
147 technologies, and how contemporary translocation decisions will contribute to meeting predator-free
148 fundamental and means objectives (specifically, more native and endemic species, less pests) if a
149 predator-free nation becomes reality. In particular we are interested in “successful” translocations, the
150 definition of which is also fraught. However, here we define a successful translocation as one that
151 meets a clear set of measurable a priori fundamental objectives (Ewen et al. 2014). A predator-free
152 Aotearoa NZ will encompass a range of values-based objectives. For the authors, more native and
153 endemic wildlife typically translates to the creation of large populations (100s-1000s of individuals)
154 with a high probability of persisting in the long term (100s of years). The distinction between reaching
155 a long-term state of persistence versus any single point in time at which success is measured is
156 important (Seddon 1999; Armstrong & Seddon 2008). Achieving this objective requires critical,
157 careful, and measurable evaluation of all of the factors that might contribute to translocation success,
158 and an understanding of the species-specific time scales over which such factors might act, rather than
159 focussing on single factors and arbitrary timeframes. We also note the increasing demand for
160 conservation translocations and that some might proceed with very different objectives to those
161 posited above, especially where there is a high level of uncertainty about translocation of a particular
162 species and/or particular release sites.

163

164 This review draws together knowledge that has been gained from Aotearoa NZ’s rich history of
165 translocations and outlines a framework to support future translocation decision-making as we work
166 towards predator-free Aotearoa NZ. First, we discuss the need to set clearly defined objectives for
167 each conservation translocation, measure outcomes against those objectives, and test our predictions
168 that our management actions will achieve these objectives (Box 1 and Figure 1). Objectives are
169 always based on collective and individual values so the most critical question is what goal or problem
170 we are trying to resolve through translocation and what are the underlying cultural, social, political
171 and management objectives? We then address 1) the extirpation and management history of the
172 translocation candidate species (e.g. what has been the outcome of previous translocations of the
173 species to the chosen release site and/or to similar release sites?), and 2) the biological and physical
174 aspects of the release site, i.e. habitat, and its ability to support the translocated species, including
175 pests and dispersal opportunities. This is followed by a discussion about 3) suitable source
176 populations and how they can be matched to release sites, including issues around health screening,
177 founder size, population growth, and whether ongoing post-release management, including genetic
178 management, is required or even feasible. Finally, 4) we briefly discuss the future of conservation
179 translocations in Aotearoa NZ, including emerging genomic tools.

180

181 Little of what we present is especially novel. However, there seems to be a perception in the general
182 Aotearoa NZ conservation community that translocations are relatively easy and success is assured,
183 something not demonstrated by data on success rates either in Aotearoa NZ (Miskelly & Powlesland
184 2013), or internationally (Griffith et al. 1989; Wolf et al. 1996; Fischer & Lindenmayer 2000). The
185 frequency of conservation translocations is also increasing (Cromarty & Alderson 2013), including
186 calls for urban translocations (van Heezik & Seddon 2018), and translocations are likely to increase
187 with predator-free Aotearoa NZ successes. Furthermore, the quality of translocation proposals
188 processed by DOC is highly variable, with some poorly written, poorly thought out, or just a bad idea
189 for the candidate species. The DOC approval process itself also produces variable outcomes.
190 Therefore, our goal is to encourage careful thinking in the formulation of conservation translocation
191 objectives, and the derivation of appropriate performance measures for these objectives, that align

192 with aspirations for a predator-free Aotearoa NZ. We discourage a focus on any single element of
193 planning and rather encourage all people involved in conservation translocations, particularly decision
194 makers, to explicitly recognise the multiple values-based objectives associated with conservation
195 translocations. The feasibility and timeframes over which predator-free objectives can be met are
196 uncertain. Regardless, we want more native and endemic wildlife and fewer pests in Aotearoa NZ. To
197 this end we anticipate this review being of utility to conservation scientists, managers, treaty partners,
198 decision makers, community based practitioners, and all others interested in these lofty objectives.
199

200 **The cultural and social setting of translocations**

201

202 Conservation translocations are most frequently conducted on public land administered by national or
203 local government and they usually involve the use of public money for at least some aspect of the
204 project. Accountability for the appropriate management of translocated species is also vested in
205 government, i.e. DOC, which is in turn bound by obligations under Te Tiriti o Waitangi/The Treaty of
206 Waitangi. Therefore, there is an immediate requirement to consult with Treaty Partners, and other
207 stakeholders, on the intention to conduct a translocation, along with what that means for ongoing
208 management of the source population, the translocated population, and the release site. However, this
209 obligation is not purely economic and legal because Treaty Partners, and often other stakeholders,
210 have deeper connections to, and interests in, the source population, the translocated species, and the
211 release site (Bioethics Panel 2019). Therefore, a translocation is usually more than just an opportunity
212 to establish a new population as it includes broader cultural and societal desires, aspirations and
213 objectives (Parker 2008).

214

215 The objectives of any particular conservation translocation are often seen as blatantly obvious to the
216 project instigators, managers and decision makers. However, these objectives are often rooted in
217 modern science and management which risks missing key fundamental objectives of Treaty Partners
218 and other stakeholders. For example, a manager trained in modern sciences might see a translocation
219 as an opportunity to restore a component of an ecosystem. In contrast, a Treaty Partner might see it as
220 an expression of kaitiakitanga (guardianship) and the restoration of mauri (not easily defined but often
221 translated as life essence), whereas a community conservation practitioner or private landowner might
222 simply want a particular species living in their area. These objectives might seem very similar but this
223 should not be assumed, nor will they necessarily be measured in the same way. Furthermore, a recent
224 review by Ewen et al (2014) showed that the setting, reporting and, critically, the measurement of
225 objectives is highly variable among reintroduction programmes, most of which are rooted in modern
226 science. Fundamental objectives are often mixed with means objectives or are not measured in an
227 appropriate way, nor even explicitly stated (Ewen et al. 2014). For example, what does predator-free
228 NZ really mean? Is this all that we want? Predator Free NZ (www.predatorfreenz.org), and the
229 authors, think not but rather see it as a means to something much more ambitious, i.e. a landscape
230 dominated by native and endemic species. However, in many cases native and endemic species will
231 not just reappear if we remove pests from the Aotearoa NZ landscape. So what do we need to do to
232 ensure we get more native and endemic species?

233

234 All stakeholders should be directly involved in setting fundamental and means objectives for any
235 particular translocation project, and then deciding between management alternatives as to how we
236 might achieve them. For example, while support for the establishment of a new population is usually
237 forthcoming, because people just want more native and endemic biodiversity, many also want to be
238 involved in the capture, handling and monitoring of translocated animals, especially kaimahi
239 (workers) eager to gain new skills. Clearly, it is critical to determine the level of involvement Treaty

240 Partners and other stakeholders might want to have at the very outset of any translocation project.
241 This is especially important as many iwi, hapū, and community conservation groups often feel that
242 they hear about translocations well after they have begun, rather being involved at the beginning.

243

244 Ultimately, meaningful engagement, consultation and decision sharing with Treaty Partners, and other
245 stakeholders, provides a means to deepen support, interest, and engagement in local, national and
246 even international conservation. This will be crucial for Predator Free 2050 aspirations to be realised
247 and is particularly important where translocated species might disperse from the release site into the
248 surrounding area (e.g. NI kākā (*Nestor meridionalis*) in Wellington), or if site management can impact
249 local communities (e.g. cat control).

250

251 **Setting objectives**

252

253 Ewen et al (2014) characterised a conservation translocation as a sequence of decisions, and argued
254 that poor planning, implementation, and monitoring is a consequence of not approaching the decision-
255 making process in a deliberate and rational manner. They, along with several other authors, advocate
256 a more structured approach to decision making (Maguire 1986; McCarthy et al. 2012; Converse et al.
257 2014; Ewen et al. 2014). Structured decision making is an iterative process whereby uncertainty is
258 addressed by 1) defining clear objectives and how they will be measured; 2) identifying a range of
259 possible management alternatives; 3) predicting the outcomes of the chosen management alternatives
260 relative to the stated objectives; 4) evaluating trade-offs and uncertainty; 5) implementing the optimal
261 management alternative and monitoring its results (Figure 1) (Gregory et al. 2012; Ewen et al. 2014).

262 This approach to decision making has been characterised as “*a formalisation of common sense for*
263 *decision problems which are too complex for informal use of common sense*” (Keeny 1982).

264 Conservation translocations seem deceptively simple, but as noted usually consist of a mix of
265 biological and non-biological values that are not necessarily equal, and in some cases might be
266 competing with each other. Therefore, careful formulation of measurable objectives provides an
267 effective and transparent way to make choices and signal success (Ewen et al. 2014). This approach is
268 especially valuable in pursuing the aspirations of a predator-free Aotearoa NZ because, while one
269 objective might seem simple, i.e. reduce or remove pests, this desire is actually deeply entwined with
270 governmental, Treaty partner, community, and individual objectives that for many, the authors
271 included, translate to a landscape dominated by native and endemic species. Therefore, a means
272 objective (remove or reduce pest populations to low density) is being confused with the fundamental
273 objective expressing what we really want (more native and endemic species). This directly relates to
274 setting objectives for conservation translocations as we move beyond translocations to typical sites
275 (islands and relatively small protected mainland areas), towards release sites with much more
276 uncertainty, e.g. very large contiguous areas of habitat (1000s of hectares), and urban (van Heezik &
277 Seddon 2018) and rural landscapes.

278

279 Understanding the extirpation history and the outcomes of previous translocations of a particular
280 species, to a given release site, and/or sites with similar characteristics, along with relevant non-
281 translocation work and theory (e.g. on dispersal) is an obvious start point for addressing uncertainty
282 and setting informative performance measures for achieving the objectives we have for any particular
283 translocation. As an example some species, such as NI robins, have persisted on the mainland,
284 including at sites with no predator management whereas others, such as NI tīeke, have been extinct on
285 the mainland for >120 years. Therefore, these two species clearly show very different levels of
286 vulnerability to pests and will require different performance measures for pest control (a means
287 objective) for a translocated population to establish and persist (a fundamental objective). In assessing

288 the outcomes of previous translocations we recommend examination of factors likely to have
289 influenced project outcomes (e.g. predation, dispersal pathways, vegetation associations, pathogens)
290 in setting performance measures but note that it can be extremely difficult to determine why a
291 translocation fails. One way is to model vital rates from another species to model the focal species
292 vulnerability to pests. For example Parlato and Armstrong (2018) used NI robin data to predict rat
293 tracking indices that might correlate with NI tīeke translocation success. Alternatively, factors other
294 than pests might lead to translocation failure. For instance, of nine korimako/bellbird translocations
295 (Miskelly & Powlesland 2013) only one (to Mana Island) appears to have successfully established a
296 breeding population. While several factors might have contributed to these failures it is unequivocal
297 that dispersal from the release site has been a critical factor, even at sites where some breeding has
298 occurred (for example, Zealandia). Given such low success it is questionable whether any further
299 translocations of bellbirds are justified, particularly given the low threat ranking of bellbirds (Miskelly
300 et al. 2008), and their ability to naturally recolonise protected sites (Brunton et al. 2008), unless there
301 is a significant change in methods or understanding. Clearly, if a species has been translocated only a
302 few times, or not at all, then the outcomes of previous translocations are not useful indicators of future
303 outcomes. In these cases, the translocation of other species, along with the ecology and conservation
304 history of the target species, will have to be assessed against vulnerability to pests, dispersal abilities
305 and other habitat requirements. However, there will naturally be a higher degree of uncertainty
306 regarding establishment and persistence of the translocated population.

307

308 **The release site**

309

310 Conservation translocations are typically, but not always, carried out within the former range of a
311 species, i.e. reintroductions (IUCN 2013), following local extirpation and where natural recolonisation
312 is unlikely on a time scale acceptable to site managers. Clearly, the conditions that we predict animals
313 need to persist must be present in the release area, although these might also be provided through
314 supportive management, for example supplemental feeding of translocated hihi.

315

316 Unfortunately, the concept of habitat is often poorly used and poorly defined in conservation
317 translocation planning (Stadtmann & Seddon 2018). Here, we use the definition of Hall et al. (1997),
318 in describing habitat “...as the resources and conditions in an area that produce occupancy –
319 including survival and reproduction – by a given organism.” This includes all physical (e.g. climate,
320 aspect, altitude, soil type) and biological (e.g. predators, competitors, vegetation associations, prey
321 species, parasites, landscape connectivity) aspects of an area where a species lives. Habitat quality
322 refers to “...the ability of the environment to provide conditions appropriate for individual and
323 population persistence” (Hall et al. 1997), specifically survival, reproduction and population growth.
324 Habitat quality is a continuous variable ranging from low quality to high quality habitats, and can be
325 very difficult to define explicitly, although there are useful proxies (Hall et al. 1997). Lambda (annual
326 population growth rate), is the most useful proxy for measuring translocation success as it needs to >1
327 for population growth to occur, until density dependence, or other limiting effects, regulate population
328 growth. High quality habitat is typically perceived as places where animals formerly occurred.
329 However, habitat conditions need not replicate past states so long as they provide the critical habitat
330 characteristics that a translocated species requires. Moving animals out of range is sometimes
331 controversial but is relevant in the highly modified ecosystems of Aotearoa NZ and under climate
332 change predictions (Chauvenet et al. 2013). Therefore, careful, but flexible, thinking might realise
333 new opportunities for more native and endemic wildlife.

334

335 **Pest control**

336

337 Pests are rarely explicitly considered as a habitat variable in Aotearoa NZ, where discussions of
338 habitat quality have focussed on vegetation associations. However, any discussion on habitat quality
339 in Aotearoa NZ must begin by defining the presence and density of pests because they have such a
340 critical impact on the survival of so many native and endemic species (Innes et al. 2010; Richardson
341 et al. 2014). While other biological and physical habitat variables will clearly be important for
342 translocation success, the role of pests is so pervasive that suitable pest control is almost always a
343 prerequisite for translocated populations to establish and persist, although the target pests, and the
344 level of control required, will vary depending on the translocated species (Table 1). In Aotearoa NZ,
345 current (2020) pest management, at least for mammalian pests, comprises three broad categories of
346 control; 1) total eradication on offshore islands, 2) maintenance of pests at “zero density” within
347 fenced mainland sites, and 3) ongoing maintenance of pests at low population densities in unfenced
348 mainland areas. These definitions are not mutually exclusive and there is often some overlap between
349 them. For example, peninsula fences, such as Tāwharanui and Shakespear Open Sanctuaries, are leaky
350 and both have extensive buffer zones on the outside of the fences. This hopefully reduces incursions
351 while also potentially providing some protection for animals that disperse outside the fence.

352

353 The key point that must be addressed early in the translocation approval process is that translocated
354 species have widely varying thresholds for coping with pests. Therefore, the pest densities maintained
355 at the release site must be within the tolerance of the translocated species (Table 1) because this will
356 directly influence which species can establish and persist at different sites following translocation. For
357 example, NI robins can persist with moderate levels of ship rats (*Rattus rattus*), but will have highest
358 survival and reproduction rates if rats are reduced to low levels ($\leq 5\%$ tracking tunnel indices) before
359 each breeding season, with mustelid control also likely to be beneficial. NI robins will actually persist
360 at ship rat tracking indices of at least 25% at some sites, but female survival, reproductive output and
361 ultimately population growth will be reduced (Parlato & Armstrong 2012, 2013). As well as
362 potentially putting population persistence at risk, slow population growth and loss of unique founders
363 will increase loss of genetic diversity and potentially lead to inbreeding depression. In stark contrast,
364 the mainland extinction history, and current distribution, strongly suggests that species such as NZ
365 tīeke, hihi, and red-crowned kākāriki (*Cyanoramphus novaezelandiae*) are much more vulnerable to
366 pests as they currently persist only in sites where pests have either been eradicated or reduced to zero
367 density. A particular challenge is that it is difficult to test vulnerability to particular pests although
368 both extinction history and modelling data from other species can be useful (see Parlato and
369 Armstrong (2018)).

370

371 A further challenge when making translocation decisions is that the impact of varying densities of
372 pests is well understood for a few bird species, poorly predicted for many others and virtually
373 unknown for most invertebrates, lizards, amphibians and threatened plants (Table 1). For example, on
374 the mainland pest thresholds and population growth in response to pest control have only been
375 demonstrated for Otago (*Oligosoma otagense*) and grand (*Oligosoma grande*) skinks (Reardon et al.
376 2012), just two of 106 endemic lizard species. Ultimately, if pests cannot be reduced to the levels
377 required for a translocated species to establish and persist, then the translocation is likely to fail.

378

379 ***Other biological and physical habitat variables***

380

381 Assuming that pests can be controlled at potential release sites, consideration must then be given to
382 other biological and physical habitat variables. In assessing these other habitat factors in Aotearoa
383 NZ, the focus is typically on the vegetation associations that the translocation candidate is known or

384 assumed to have inhabited, and which provide feeding, nesting and refuge opportunities that support
385 establishment and persistence. However, other habitat variables are equally important. The physical
386 size of the release site, often defined by the extent of pest control, is a critical consideration simply
387 because big well-protected sites can support large populations. In contrast, small populations at small
388 sites are more vulnerable to extinction for a range of reasons, e.g. pest incursions, extreme weather
389 and stochastic events. In the medium to long term, small populations are also more vulnerable to the
390 negative impacts of loss of genetic diversity (see discussion below) (Jamieson & Lacy 2012; Keller et
391 al. 2012; Weiser et al. 2013; Frankham et al. 2017). This can be managed through ongoing expansion
392 of protected sites, the creation of natural corridors to other protected sites and supplemental
393 translocations (Weiser et al. 2013; Frankham et al. 2017). However, all of these options require
394 ongoing commitment and resources. This does not mean that conservation translocations to small sites
395 should not happen but rather that uncertainty and management challenges must be implicitly
396 recognised by all decision makers at the outset of any translocation (Box 1).

397

398 Other habitat variables, including climate, altitude, aspect, and soil type will also clearly be associated
399 with different vegetation associations and might shift habitat quality from high to low, i.e. decrease
400 the probability of establishment and persistence, depending on the needs of the translocated species
401 and their ability to adapt to variable conditions. This might be especially difficult at highly variable
402 sites, especially those that experience climatic extremes relative to those with more benign conditions.
403 In addition, predicted climate change might mean high quality habitat will become low quality in the
404 future. Furthermore, the impact of these variables is not consistent across species. For example, some
405 species, such as NI robins and mātātā/NI fernbirds, appear to be flexible in their habitat requirements
406 and have been translocated successfully to very contrasting habitats, although productivity and
407 population growth has varied between sites suggesting that some are better than others (Parlato &
408 Armstrong 2012, 2013). In stark contrast, species such as hihi need protection from mammalian pests
409 but also seem to have other unknown habitat needs (Ewen et al. 2013), i.e. pest control alone is not
410 currently enough for a large population of hihi to establish without additional intensive management
411 via supplemental feeding.

412

413 ***Habitat connectivity and dispersal***

414

415 Habitat connectivity, and the concomitant ability for species to disperse between habitats, is typically
416 seen as a highly positive landscape feature and a desirable management objective. However, habitat
417 connectivity and dispersal opportunities from managed release areas into adjacent unmanaged areas
418 appear to be key determinants of success in many translocations (Richardson et al. 2014). Dispersal
419 generally affects population growth at two levels. First, post-release dispersal following the initial
420 release can cause the loss of individuals from the founding population, thereby reducing the
421 probability of establishment and persistence. For example, in an analysis of 14 reintroduced
422 toutouwai/NI robin populations Parlato and Armstrong (2013) showed that habitat connectivity was a
423 key factor in determining individual establishment following translocation, with individuals released
424 at highly connected sites having a lower establishment probability than those at less connected sites,
425 such as an island or isolated mainland reserve. Second, natal dispersal, i.e. the loss of juveniles raised
426 at the release site can also reduce establishment and persistence if juveniles move from managed to
427 unmanaged sites (Richardson et al. 2014). Critically, the interaction of post-release dispersal and natal
428 dispersal can limit population growth, erode genetic diversity and reduce the likelihood of the long-
429 term persistence of a translocated population.

430

431 The propensity and abilities of translocated species to disperse from release sites is highly variable
432 and sometimes difficult to predict (Table 1) (Richardson et al. 2014). For instance, some translocated
433 species, especially birds, are very strong dispersers regardless of habitat connectivity. This includes
434 korimako/bellbird, miromiro/tomtit (*Petroica macrocephala*), and red crowned kākārīki (Parker et al.
435 2004; Brunton et al. 2008; Ortiz-Catedral 2010) whereas others, such as NI toutouwai/robin and NI
436 tīeke, are less likely to disperse (Newman 1980; Richard & Armstrong 2010). However, the inherent
437 dispersal abilities of a translocated species directly interact with the landscape features of the release
438 site, specifically the degree to which it is connected to surrounding unprotected habitats, although the
439 shape of this relationship remains unknown for all species, and connectivity is sometimes difficult to
440 characterise (Figure 2). Many species, including some with relatively strong dispersal abilities, rarely
441 leave isolated sites such as islands or forest patches surrounded by pasture. In contrast, species with
442 poor dispersal abilities can move out of protected areas if connected to habitat that the species will
443 willingly move through (Richard & Armstrong 2010), although this is likely to be a greater problem
444 for birds and bats relative to reptiles, amphibians, invertebrates, and plants.

445

446 The best way to manage dispersal in contiguous landscapes is to manage as large an area as possible,
447 including potential dispersal routes, through an integrated landscape management approach
448 (Richardson et al. 2014). However, beyond protecting everything it is not currently known how big a
449 site needs to be to accommodate post-release and natal dispersal in most species, and in many cases it
450 will be difficult, too expensive, or simply not feasible to protect very large sites. Therefore, this
451 currently limits our ability to translocate some species to the large sites that will increasingly be the
452 target of Predator Free 2050 operations. A variety of alternative approaches have been used to try to
453 reduce dispersal, albeit with variable and limited results. Holding animals in captivity at the release
454 site (delayed release) has been tried with many taxa, and many sites, but the results have been
455 extremely variable, i.e. generally ineffective for wild to wild releases, but sometimes useful when
456 releasing captive-reared animals (Parker et al. 2012b; Richardson et al. 2013; Smuts-Kennedy &
457 Parker 2013; Richardson et al. 2014; Parker et al. 2015). Supplementary feeding has also been used
458 with success for some species at some release sites (e.g. kākā, pāteke (*Anas chlorotis*) (Rickett et al.
459 2013)), but has been less useful for others (e.g. hihi) (Richardson et al. 2014). Acoustic anchoring
460 (playback of pre-recorded calls) has also been used on NI kōkako, NI toutouwai/robins, and
461 popokatea/whiteheads in NZ, but does not appear to be effective in reducing dispersal (Leuschner
462 2007; Molles et al. 2008; Bradley et al. 2011).

463

464 Another option for mitigating the impact of dispersal in the early stages of establishment is the release
465 of large numbers of individuals, either in one big release or over several years. This is intuitively
466 appealing but is rarely effective because if initial post release dispersal is a problem then dispersal
467 will likely remain a problem via natal dispersal (Richardson et al. 2014). In addition, there are many
468 examples where relatively large numbers of animals have been released but the translocations have
469 failed (Miskelly & Powlesland 2013), despite release into habitats that should enable persistence once
470 established (Armstrong & Seddon 2008). For instance, popokatea/whitehead translocations have been
471 successful to many sites with founding populations of 40-100 birds. However, at one large (c.17 000
472 ha) contiguous site in the Waitakere Ranges with a protected block of 2450 ha, 653 birds were
473 released over 12 years in an effort to compensate for post-release dispersal. In stark contrast to
474 isolated sites up to 3300 ha in size, the Waitakere translocation is showing few signs of success (K.A.
475 Parker, *unpublished data*). In addition, the true relationship between release group size and
476 establishment is unclear (Armstrong & Wittmer 2011). This is because high quality sites where
477 translocations are successful following the release of large numbers of animals could have been
478 equally successful if fewer animals were released (Armstrong & Wittmer 2011). In contrast, managers

479 typically release fewer animals when they have less confidence in a site, creating a reporting bias
480 towards success with larger releases (Armstrong & Seddon 2008; Armstrong & Wittmer 2011). There
481 are also significant welfare, ethical and relational risks around translocating large numbers of animals
482 with the expectation that many will die following translocation, especially where translocation is not
483 essential for species management. This uncertainty needs to be carefully and openly considered and
484 discussed at the policy level, so that decision makers can make good defensible decisions at a
485 national level, and with all Treaty Partners and stakeholders involved in any given translocation
486 project.

487
488 Ultimately, the best way to reduce dispersal is to release animals at isolated or relatively isolated sites.
489 However, the great challenge with managing dispersal, and in meeting Predator Free 2050 aspirations,
490 is that we want translocated species to establish populations within large contiguous sites, and we
491 want individuals to be able to freely disperse between sites. This will protect against the problems of
492 populations being small, and will largely remove the need for supplemental translocations for genetic
493 management, i.e. natural dispersal via safe dispersal corridors will essentially act as passive meta-
494 population management. It will also open up new opportunities for populations in smaller sites. The
495 critical requirement will be safe dispersal corridors. In the current environment this generally means
496 protection from pests but as pest control improves other habitat variables will become more important.
497 For example, what size, shape, and structure do corridors need to be to cater for as wide a range of
498 native and endemic species as possible? Perhaps the best way to measure the ability of animals to
499 safely disperse from intensively managed areas will be as a performance measure for Predator Free
500 2050 aspirations. Furthermore, dispersal pathways should be incorporated into decisions about which
501 landscapes to protect first.

502

503 **Matching source populations to the release site**

504

505 The choice of source population raises several important considerations. The first is simply whether
506 the source population can sustain a harvest with minimal negative impacts? (We acknowledge there
507 are exceptions to this, especially mitigation translocations whereby the habitat sustaining the source
508 population is destroyed). Most source populations are “black boxes” in that we know little about their
509 population dynamics and vital rates. However, data from closely monitored populations (Armstrong
510 & Ewen 2013), along with translocation records (Lovegrove 1996; Miskelly & Powlesland 2013;
511 Parker 2013), demonstrate that some populations can be harvested at surprisingly high rates over
512 extended periods.

513

514 *How similar are the source and release sites?*

515

516 Does the source site share similar habitat characteristics especially the presence or absence of pests,
517 vegetation associations and pathogens? This is not necessarily critical because, as noted above, some
518 species seem to be quite tolerant of contrasting habitats. However, even within these species,
519 translocation between similar habitats is likely an easier transition than translocation between
520 contrasting habitats. For instance Parlato and Armstrong (2012, 2013) showed that translocation of NI
521 robins between habitats with similar pest assemblages and vegetation associations had a small
522 advantage over those between contrasting habitats. The similarity of the source and release site, the
523 objectives of the translocation, and especially the risk profile or level of uncertainty associated with
524 the translocation will also influence decisions about health screening. For example, translocation
525 between two mainland sites and/or inshore islands that are relatively close together likely represents a
526 low pathogen risk because their pathogen communities are likely to be similar. In contrast,

527 translocation between distant sites with different habitats might prompt a more considered approach,
528 especially if the recipient site has resident populations of highly valued species that could be put at
529 risk through the introduction of novel pathogens. Ideally, there is also an understanding of potential
530 pathogen impacts on the translocated species, on conspecifics and heterospecifics at the release site,
531 and/or a documented history of health screening (Parker et al. 2006; Ewen et al. 2007; Ortiz-Catedral
532 et al. 2011; Ewen et al. 2012; Massaro et al. 2012) to inform decisions about health management.
533 Unfortunately this information is usually lacking or of poor quality.

534

535 *Managing genetic diversity*

536

537 Genetic diversity is critical for maintaining evolutionary potential by providing a population with the
538 long-term capacity to adapt to changing conditions (Frankham et al. 2012; Frankham et al. 2017). All
539 populations lose genetic diversity over time as a result of chance events through genetic drift but
540 small populations are especially vulnerable (Frankham et al. 2012; Frankham et al. 2017). Inbreeding
541 (mating between relatives) is also problematic in small populations because it can reduce survival and
542 reproductive success (inbreeding depression) which in turn threatens population persistence
543 (Frankham et al. 2012; Frankham et al. 2017). Translocations often impose a genetic bottleneck on
544 new populations because of the number of founders released. This is often further accentuated
545 because the number of founders that actually recruit and contribute to the new population is usually
546 smaller than the number released. In addition, some translocated populations will always be small, as
547 many managed sites are small. Translocated populations are thus susceptible to the negative genetic
548 consequences of genetic drift and inbreeding depression.

549

550 Therefore, careful thinking is required in setting genetic objectives to minimise the loss of genetic
551 diversity, both in selecting a source population, or populations, and in predicting the genetic diversity
552 of the translocated population (Weeks et al. 2015). It is also essential to clarify whether genetic
553 objectives are fundamental or means based. For example, we are rarely interested in maintaining
554 genetic diversity for its own sake, i.e. as a fundamental objective (although some, including several of
555 the authors, consider the maintenance of evolutionary potential as a fundamental objective). Rather
556 our interest in genetic diversity is usually as a means objective that contributes to the long-term
557 persistence of the translocated population by maintaining evolutionary potential. If this is the case
558 then a means objective might be releasing sufficient numbers of animals to maximise genetic diversity
559 in the founders and therefore the long-term adaptive potential of the new population.

560

561 Alternatively, there are many reasons why our values and objectives might mean a very small (≤ 100
562 individuals) translocated population is created, including because only small numbers of animals
563 exist, ease of management, advocacy, or simply that only small sites are available for release. In these
564 cases, genetic means objectives might include informed supplemental translocations to maintain
565 genetic diversity across a larger managed metapopulation. All management involves trade-offs. For
566 example, the best source populations are typically large and have no history of very tight (<40-100
567 individuals) and/or long-term bottlenecks (the effects of short-term bottlenecks are sometimes
568 acceptable if the bottleneck was small and of short duration). However, an inshore island might be an
569 easier and cheaper option as a source population, but have lower genetic diversity, than a more
570 expensive and logistically challenging offshore island population with higher genetic diversity.
571 Another option would be increasing the size of the release area through improved pest control thereby
572 enabling a larger population to establish and removing or reducing the need for supplemental
573 translocations. Alternatively, the cost of ongoing maintenance of a large release site, and translocation

574 of a large diverse founder population, might be greater than managing a much smaller site with
575 ongoing supplemental translocations, at least in the short to medium term.

576

577 Useful additional considerations in aligning genetic management with translocation objectives include
578 what is the genetic profile and history of the source population or populations and will it provide
579 genetically diverse individuals for the translocation? How many individuals are needed to capture
580 that diversity? And following release how many animals can the site eventually support? If it is small
581 and supplemental translocations are recommended how easy will this be to actually achieve? The
582 feasibility of follow-up translocations is often presented in a simplistic manner with little recognition
583 of the cost and difficulties in getting additional animals to recruit into an established population.
584 Often, very large numbers of individuals must be added to ensure that at least a few will be able to
585 recruit and breed in the established population (Weiser et al. 2013), as density dependence
586 (Armstrong et al. 2005) or behavioural barriers (Parker et al. 2010a; Parker et al. 2012a) are likely to
587 reduce recruitment of immigrants. As noted above, releasing large numbers of animals in the
588 expectation that few survive also has welfare, ethical and relational implications.

589

590 Regardless of the management alternative selected for maintaining genetic diversity it is important to
591 remember that not every translocated population has to represent maximal or ideal genetic diversity.
592 Overall genetic diversity can also be represented and conserved within a metapopulation connected
593 either via natural dispersal or management. This likely represents a more “natural” scenario (e.g.
594 genetic diversity will not be equal across all natural populations, especially when moving from the
595 core of a species range to the edges) whilst also increasing options for establishing and maintaining
596 translocated populations that cater to a wide range of values and objectives.

597

598 **The future of conservation translocations in Aotearoa New Zealand**

599

600 Conservation translocations are likely to play an increasing role in Aotearoa NZ conservation.
601 Ongoing practice and research will deepen our understanding of the values driving translocations
602 including, but not limited to, societal desires, cost, animal welfare, genetic, and pathogen
603 management, translocation techniques and dispersal. However, in Aotearoa NZ the biggest
604 opportunities will come about through improved control of pests over large unfenced areas of the
605 mainland, including forests, wetlands, dryland and braided river systems, and alpine zones. This will
606 provide a means to translocate species that are currently in higher threat categories, along with
607 providing further options for management and translocation of all species, especially habitat
608 specialists, such as whio, kāki/black stilt, and pīwauwau/rock wren (*Xenicus gilviventris*), and
609 neglected taxa, such as lizards, amphibians, invertebrates, and threatened plants. While opinion varies
610 on the feasibility of effective pest control over vast swathes of Aotearoa NZ (Urlich 2015) it will
611 clearly be a game changer if it can be achieved. We also expect to see an increasing shift away from
612 conservation translocations for single-species recovery toward those where the fundamental objective
613 is ecosystem restoration (Parker 2013). Pathogens and predators, such as weka, small rails (*Rallus*
614 spp.), crakes (*Porzana* spp.) and NZ karearea/falcons (*Falco novaeseelandiae*) are obvious
615 components of NZ ecosystems that are currently either actively avoided in restoration plans, or
616 relegated to some point in the distant future once their potential prey or host species are well
617 established. It seems logical to stage restoration sequences such that prey species are established
618 before predators, although it is important to distinguish between a pest, against which native and
619 endemic species have few defences, and a native or endemic predator that they have co-evolved with
620 over 10000s of years. For example, translocated Middle Island tusked wētā (*Motuweta isolata*) and
621 wētāpunga (*Deinacrida heteracantha*) have established in the presence of very high densities of a

622 natural predator, the NI tiēke, whereas pests caused the extinction of many large wētā populations
623 elsewhere. Therefore, conservation translocations of predators will require acceptance that there will
624 be ongoing predation, possibly a reduction in population size, and changes in the behaviour of prey
625 species. This will be difficult for some people to accept and could become problematic for very small
626 prey populations, but it is a logical objective for true ecosystem restoration. It might also require
627 flexible thinking in the management of predator species, and pathogens, especially where there is a
628 management need or perception that natural predators and pathogens have to be controlled.

629
630 There has also been considerable debate about the ongoing impacts of global climate change and how
631 conservation translocations can be used as a tool for species whose habitat will deteriorate under
632 current climate change predictions (Hoegh-Guldberg et al. 2008; Seddon et al. 2009; Seddon 2010). In
633 Aotearoa NZ this would likely mean moving animals across latitudinal gradients, e.g., between the
634 North and South Islands. For instance, climate modelling suggests that the northern South Island,
635 where hihi have never existed, might provide more suitable habitat at some time in the future than the
636 North Island, to which they are currently restricted (Chauvenet et al. 2013). Any decision to undertake
637 a translocation beyond a species' natural range will also clearly raise challenges in setting appropriate
638 objectives, especially if it would bring closely related species into contact.

639
640 Emerging genomic tools will further enhance translocation decisions (Luikart G. 2018; Santure et al.
641 2018; Funk et al. 2019). With advanced high-throughput sequencing technologies, combined with
642 rapidly dropping costs, increased capability and capacity in the conservation genetics community,
643 10s-100000s of markers from across the entire genome are readily available, even for non-model
644 species (Harrisson et al. 2014; Galla et al. 2019). These genome-wide markers can increase resolution
645 for translocation questions previously answered using just a handful of neutral genetic markers. For
646 example, genomic data can provide more robust estimates of relatedness to enhance pairing decisions
647 in conservation breeding programmes that include translocations (e.g., (Galla et al. 2020)). Further,
648 the promise of characterising adaptive variation has reignited debate over how we should source, or
649 mix, populations to enhance adaptive potential—that is, the ability of individuals, populations or
650 species to respond to environmental change (Ralls et al. 2017; Kardos et al. 2018; Kolodny et al.
651 2019; Kyriazis et al. 2019). Although there has been a surge of papers focused on characterising
652 adaptive variation (Funk et al. 2019; Hoelzel et al. 2019), there are relatively few empirical studies to
653 date and it remains difficult to translate theory into practise (Flanagan et al. 2017). Indeed, for many
654 threatened species it may prove challenging to characterise adaptive variation at all (Box 2).

655
656 More recently, a new era of conservation genomics has emerged that reintegrates the packaging and
657 function of DNA, and how these mediate the transfer of genomic information between parent and
658 offspring (Deakin et al. 2019; Liberles et al. 2020). For example, emerging chromosomal approaches
659 combine genomic data with cytogenetics (chromosome architecture), epigenomics (histone
660 modifications) and cell biology to reveal the mechanisms underpinning behavioural and phenotypic
661 traits under selection (Wellenreuther & Bernatchez 2018). Although these approaches certainly come
662 with their own caveats (Potter & Deakin 2018; Deakin et al. 2019), genomic and chromosomal
663 approaches are a valuable addition to the conservation translocation toolbox, particularly in the face of
664 novel challenges such as climate change (Bay 1999; Ruegg et al. 2018).

665
666 Another interesting proposition is the suitability of translocating close relatives of extinct species as
667 ecological replacements in ecosystem restoration (Atkinson 1988). For example, the Snares Island
668 snipe (*Coenocorypha huegeli*) was translocated to replace the extinct South Island snipe
669 (*Coenocorypha iredalei*), the North Island kōkako was translocated as a replacement for the presumed

670 extinct South Island kokako (*Callaeas cinerea*) and South Island takahē (*Porphyrio hochstetteri*) are
671 frequently translocated to the North Island (Jamieson & Ryan 2001; Parker et al. 2010b; Miskelly &
672 Powlesland 2013) (although takahe translocations are motivated by species recovery goals rather than
673 as a replacement for the extinct mōho, or North Island takahē (*Porphyrio mantelli*)). It has also been
674 suggested that the Australian brown quail (*Synoicus ypsilophorus*) is a suitable ecological replacement
675 for the extinct New Zealand quail (*Cotunix novaeseelandiae*) (Parker et al. 2010b). These species, and
676 others, might be useful for restoring ecosystem services, known or otherwise. In addition, genetic
677 techniques are advancing to the point where de-extinction, the resurrection of functional proxies of
678 extinct species, might become feasible (Seddon et al. 2014; Seddon 2017). This is a contentious issue
679 and the objectives of any such proposal will have to be very carefully considered, including the
680 opportunity cost of diverting funds from extant species to de-extinction proposals (Bennett et al.
681 2017).

682 683 **Conclusions**

684
685 Regardless of the specific purpose of future conservation translocations in NZ, we contend they
686 should be driven by carefully considered, constructed and communicated a priori objectives that
687 represent the values of all stakeholders and consider how the release site and the source
688 population/populations can be matched to maximise performance relative to these objectives (Box 1).
689 Haphazard conservation translocations can cause problems at the release site, for future
690 translocations, and in maintaining equitable relationships with Treaty Partners, other stakeholders, the
691 relevant agencies, and the general public. We disagree with the suggestion that conservation
692 translocations in Aotearoa NZ have not been guided by clear principles (Parliamentary Commissioner
693 for the Environment 2017). However, we do agree that these principles are not currently captured in
694 policy and that the fundamental objectives of many translocations have rarely been stated implicitly,
695 or can be dominated by singular means objectives. A clear and widely consulted translocation policy
696 framework would enable DOC decision makers to make better decisions about all translocations,
697 including those that might contribute to Predator Free 2050 aspirations. This policy should
698 specifically acknowledge that translocations are values based, should be driven by an understanding
699 of the problem at hand, require informed decisions between management alternatives (including
700 rejecting translocation as a management tool for some species/programmes), and should be measured
701 by implicitly stated objectives with appropriate performance indicators. Ultimately, being clear about
702 what all partners and stakeholders really want will set us on the right path towards the Aotearoa NZ
703 landscape being one that is once again dominated by indigenous biodiversity.

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706
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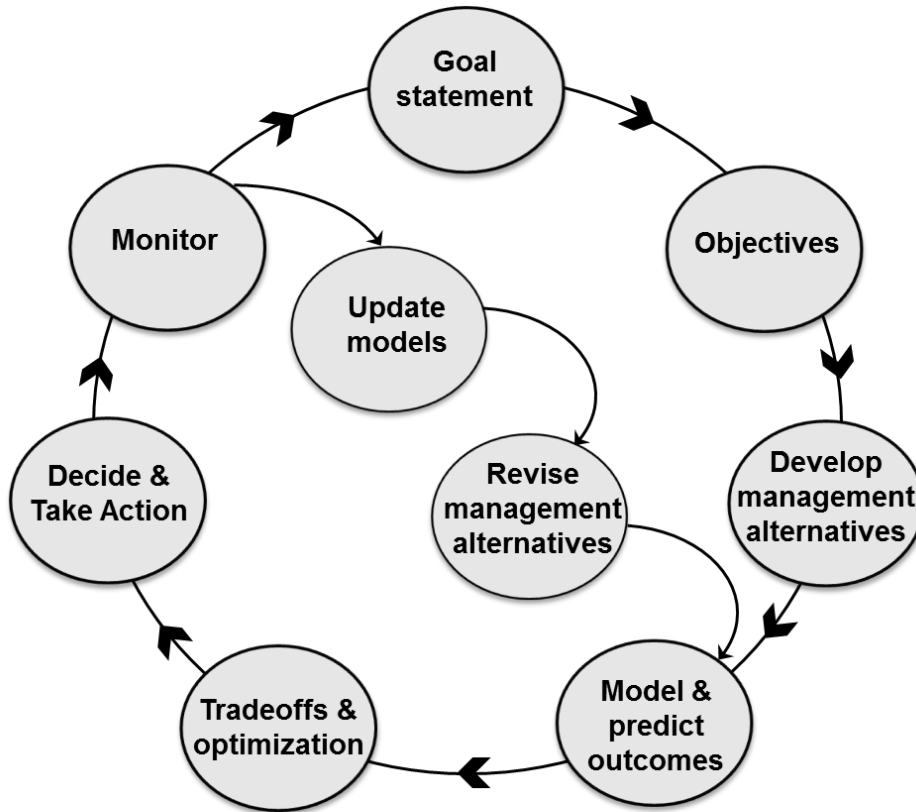
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971 **Box 1.** Some considerations for conservation translocations in Aotearoa New Zealand. Of these, the
972 first is the only critical step because, if done correctly, it will naturally envelop all other
973 considerations, both listed and unlisted.

1. All conservation translocation decisions are values based. Therefore, the cultural and social setting of a translocation is the single most critical factor in determining fundamental objectives (what we want) and means objectives (how we get what we want). If this is done correctly all other decisions will be better and easier.
2. What is the extirpation and management history of the translocation candidate and is natural recolonisation likely on an acceptable time scale?
3. Does the release site habitat (e.g. pests, vegetation associations, pathogens) match the proposed source population? If not, why is the release site considered appropriate? Can management ameliorate differences?
4. How connected is the release site and is dispersal a likely impediment to establishment and persistence?
5. How big is the release site and what is the maximum population size it can support?
6. Can the proposed source population/populations sustain harvest and what is its genetic history (e.g. size, bottlenecks)?
7. Will genetic management be required and how realistic is it that the management will be implemented (e.g. increase the number of founders, conduct supplemental translocations, increase the management area)?
8. Will future developments (e.g. improved pest control or emerging genomic tools) improve management of the translocation at hand?

975 **Figure 1.** Steps in the conservation translocation structured decision making process (adapted from
976 Gregory et al. 2012). Note the double loop learning whereby monitoring might lead to a revision of
977 management alternatives.



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980 **Figure 2.** A hypothetical relationship between expected population equilibrium density and habitat
981 connectivity mediated dispersal following translocation. The grey areas with solid black lines are
982 managed habitat. Those surrounded by dashed lines are unmanaged. The light stippled area
983 surrounding the first three managed areas represents habitat with a high resistance to dispersal (e.g.
984 open water or pasture). However, resistance to dispersal decreases as connectivity increases, i.e. when
985 managed areas are closer to unmanaged areas. The managed area on the right is within contiguous
986 habitat (grey stipple) that provides no resistance to dispersal (e.g. a managed forest patch within a
987 larger unmanaged forest). In this case dispersal/emigration is acting as mortality. A similar shaped
988 curve would be seen for other sources of mortality, e.g. increasing predator density. While it is
989 unequivocal that dispersal is problematic and directly related to connectivity the exact shape of the
990 curve is largely unknown for most species.
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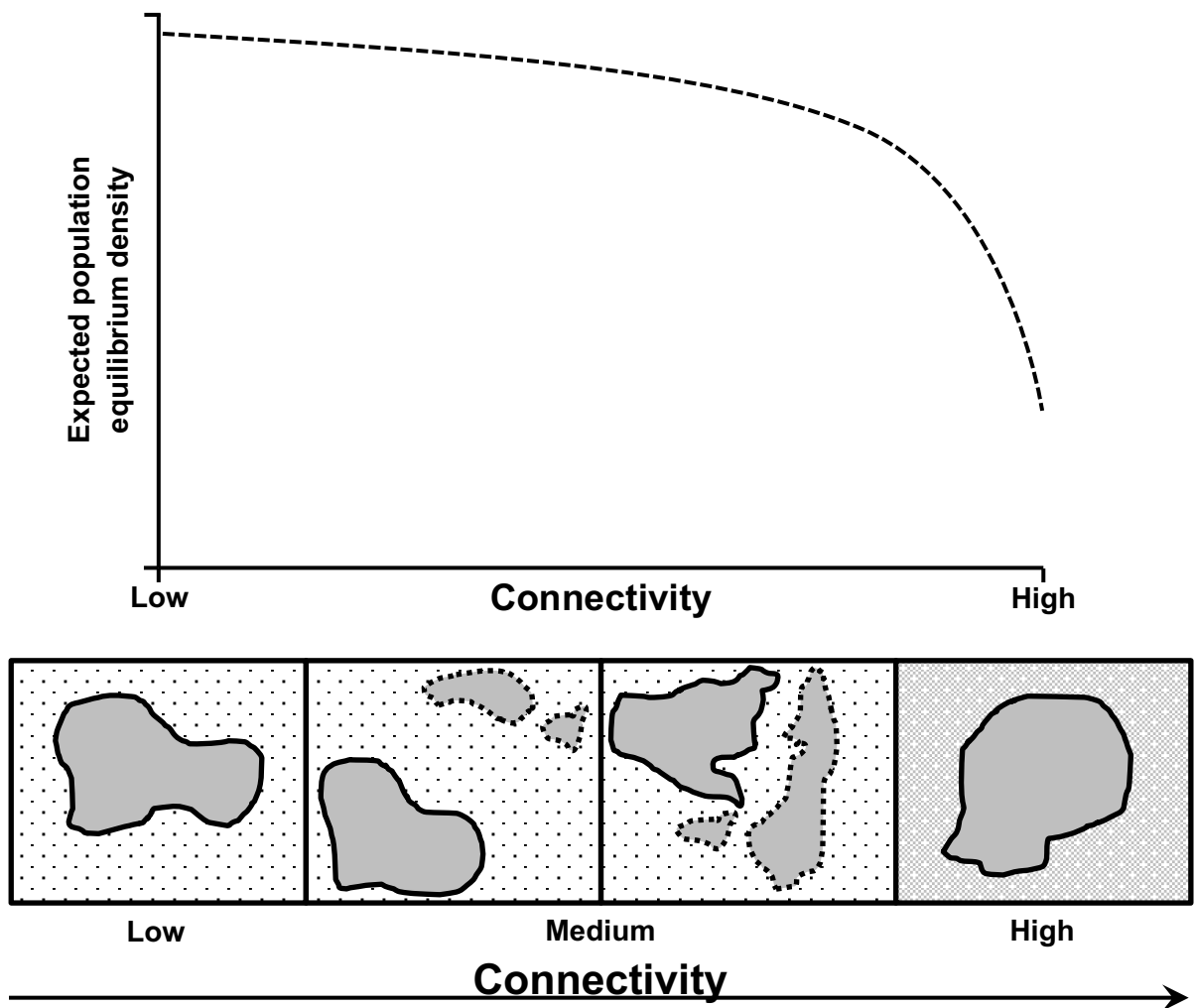


Table 1. Known or probable pest control thresholds, extinction history, and key uncertainties, for some terrestrial species that might be translocated in Aotearoa NZ. Knowledge is patchy, even for many bird species, and there is a lot of uncertainty to resolve, especially for herpetofauna and invertebrates. Other habitat variables, such as ideal vegetation associations, can be difficult to resolve until suitable pest control is in place.

Pest control delivery	Translocation candidates	Extinction history and current distribution	Ability to disperse when connectivity is:			Key uncertainties
			High	Medium	Low	
Key pest species controlled to low density, typically mustelids	Kiwi spp.	Extinct across most of their natural range Declining at unmanaged mainland sites Stable/increasing at managed sites	High	High	?	Availability of birds, i.e. balancing community desires with national recovery objectives
	Weka spp., particularly NI and buff weka	Extinct across most of their natural range Persisting at some unmanaged mainland sites Stable/increasing at managed sites	High	High	High	Weka are generally neglected and need managed sites, especially NI and buff weka Prone to population fluctuations in response to drought Possible undesirable impacts on reptiles and threatened invertebrates, although likely less of a problem at very large mainland sites Incompatible with burrowing seabirds at small sites and islands Weka often interfere with management devices such as bait stations and traps
	Whio	Extinct across most of their natural range Persisting at some unmanaged mainland sites Stable/increasing at managed sites	High	High	?	Habitat plasticity?

<p>Multi-species pest control to low density, typically including ship rats, mustelids, possums and cats, sometimes including ungulates and pigs.</p> <p>Mice usually present, sometimes at high density</p> <p>Control is sometimes delivered seasonally (e.g. over the bird breeding season)</p>	Robin spp.	<p>Extinct across most of their natural range</p> <p>Persisting at some unmanaged mainland sites</p> <p>Stable/increasing at managed sites</p>	High	?	Low	Density is highly variable at managed sites, likely due to climate and vegetation associations
	Yellow crowned kākārīki	<p>Extinct across most of their natural range</p> <p>Persisting at some unmanaged mainland sites</p> <p>Stable/increasing at managed sites</p>	High	High	?	Suitable source populations (logistically and genetically)
	Whiteheads	<p>Extinct across most of their natural range</p> <p>Persisting at some unmanaged mainland sites</p> <p>Stable/increasing at managed sites</p>	High	Moderate	Low	
	Mohua	<p>Extinct across most of their natural range</p> <p>Present at some unmanaged mainland sites</p> <p>Stable/increasing at managed sites</p>	High?	?	Low	
	Rifleman	<p>Extinct across most of their natural range</p> <p>Persisting at some unmanaged mainland sites</p> <p>Stable/increasing at managed sites</p>	?	?	Low	
	Kākā	<p>Extinct across most of their natural range</p> <p>Present at some unmanaged mainland sites</p>	High	High	High	Suitable source populations (logistically and genetically) Cost

		Stable/increasing at managed sites					
	North Island kōkako	Extinct across most of their natural range	High	?	Low	Availability of birds, i.e. balancing community desires with national recovery objectives	
		Stable/increasing at managed sites					
	Short-tailed bats	Extinct across most of their natural range	High	?	?	Successful translocation techniques have not been developed	
		Stable/increasing at managed sites					
	Mainland herpetofauna, e.g. Northern spotted skinks and the infrapunctatum complex, jewelled and forest geckos, Hochstetter's frog	Patchily distributed Persisting at unmanaged mainland sites but true status usually unknown Status at managed sites usually unknown	?	?	?	Successful translocation techniques have been developed for many species but usually overlooked in restoration projects The impacts of mice, especially at high densities, are poorly known but probably significant Often displaced by development thereby potentially providing a source of animals for translocation to appropriate sites Typically less likely to disperse c.f. birds, but much remains unknown.	
	Mainland invertebrates	Poorly known	?	?	?	With few exceptions (e.g. some land snails) there is little knowledge about the impacts of pest management and connectivity on most mainland invertebrates	
Multi-species pest control to eradication or zero density of all mammalian pests with the probable exception of mice (as is	Saddleback spp.	Extinct on the mainland late 1800s	High	Low	Low	Vulnerable to even very low densities of mustelids (individual animals) and rats (rat threshold currently unknown). NI saddlebacks persisted with kiore, SI saddlebacks did not, suggesting a greater degree of vulnerability	

typical of all mainland fenced sanctuaries).	Hihi	Extinct on the mainland late 1800s	High	Moderate?	Low	Likely similar vulnerability and pest thresholds as saddlebacks
	Kākāpō	Last males extinct on the mainland c. 1980s/1990s	High	?	?	Size and suitability of site and alignment with national recovery objectives
Multi-species pest control to eradication or zero density of all mammalian pests, including mice.	Highly threatened herpetofauna, e.g. McGregor's, robust, and Whitaker's skink, Duvaucel's gecko, tuatara	Extinct on the mainland	?	?	?	Vulnerability to mice and dispersal abilities unknown
	NZ snipe	Extinct on the mainland	?	?	?	Vulnerability to mice and dispersal abilities unknown
	Large native and endemic threatened invertebrates, e.g. giant wētā, weevils and beetles	Mostly extinct on the mainland	?	?	?	Vulnerability to mice unknown Dispersal abilities unknown but probably low

Box 2: Can we really characterise adaptive variation in threatened species?

With the emergence of next-generation sequencing in applied conservation has come the promise of characterising adaptive variation (Flanagan et al. 2017). For instance, approaches that incorporate information from the entire genome (e.g., whole-genome resequencing) or target putatively adaptive regions (e.g., SNP arrays) should dramatically increase our ability to identify adaptive genomic variants. There is growing interest in incorporating this additional information into conservation translocation decisions; but there are caveats. To date, successful characterisation of adaptive variants has largely been restricted to well-studied species, with a high-quality reference genome and comprehensive genomic and non-genomic data, such as informative fitness measures and environmental data (Attard et al. 2017; Flanagan et al. 2017; Harrisson et al. 2017). For these well-studied species, we are better able to explore a range of analytical approaches (e.g., outlier-detection based approaches, genotype-environment association studies and genome-wide association studies) (Rellstab et al. 2015). Further, new studies indicate that our chances of detecting locally-adaptive variants are highest in large, connected populations distributed across heterogenous habitats (e.g., Barrett et al. 2019). Thus—while genomic approaches are more likely to capture regions of the genome under selection compared to genetic approaches—characterising adaptive variation may still prove challenging for many threatened species (Fig. 3). Although characterising adaptive variation remains a promising conservation genomics tool, scientists and practitioners must be realistic around how readily it can be incorporated into translocation decisions.

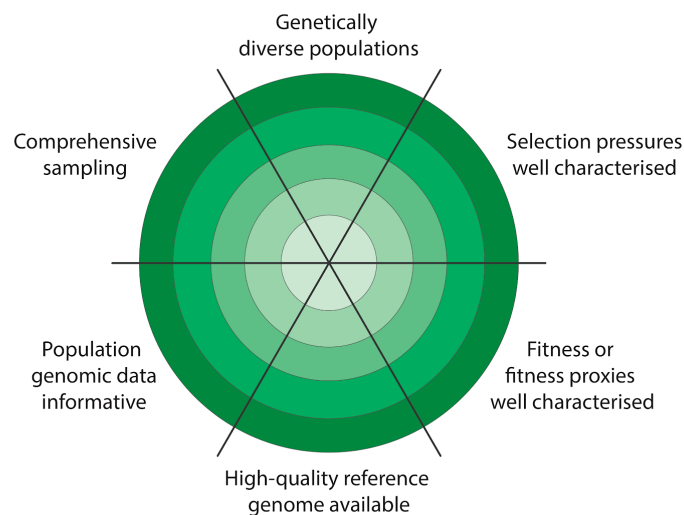


Figure 3. A novel framework for assessing key criteria for characterising adaptive variation in threatened species, including whether (i) populations are sufficiently large and genetically diverse to differentiate between selection and genetic drift; (ii) differential selection pressures are well characterised; (iii) fitness measures—or suitable proxies—are well characterised; (iv) a high-quality reference genome is available; (v) population genomic data adequately captures genome-wide diversity; (vi) comprehensive sampling is representative of relevant locally adapted populations. The further each coloured section extends toward the green circle edge reflects how well that consideration is met. Overall image design after Suding et al. (2015).