REVIEW

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

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Abstract

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

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

Introduction

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

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

The first and most urgent means by which we can achieve this is to maintain and increase the biodiversity we still have. We are very good at doing this on islands. However, we are also making gains, at least for some forest birds, close to many urban areas, where growing community conservation initiatives have led to the establishment of mainland ecological restoration projects involving varying levels of pest control, planting, and conservation translocations. Many such projects have been successful in achieving high-density populations of native and endemic wildlife, again with an emphasis on forest birds. A critical limitation is that most of these restored sites are small (c.100-1000ha), and mice (*Mus musculus*) have rarely been eradicated, or even sufficiently controlled, with important implications for the recovery of endemic lizards, amphibians, invertebrates, bats, and threatened plants. In contrast, the bulk of our biodiversity is contained within vast areas (1000s of hectares) of back country conservation estate which are much harder to protect and harder for the general public to engage with. The Department of Conservation (DOC) "Tiakina Ngā Manu/Battle for our Birds" programme is achieving impressive pest control over huge areas of Aotearoa NZ forests (c. 1 million ha in 2019), operating in parallel with species-focussed mainland recovery programmes (e.g. kakī/black stilt (*Himantopus novaezelandiae*) and orange fronted kākāriki (*Cyanoramphus malherbi*)). Nevertheless, vast tracts of land, especially non-forested habitats, remain unprotected, and biodiversity continues to decline. This is reflected in the most recent NZ threat classification for birds (Robertson et al. 2017) that has seen some species previously ranked "non-threatened" move to "at risk, declining", including popokatea/whiteheads (*Mohoua albicilla*), North Island (NI) and South Island (SI) toutouwai/robins (*Petroica longipes* and *P. australis*), and NI and SI mātātā/fernbirds (*Bowdleria punctata vealeae* and *B.p. punctata*).

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

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

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

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

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

Little of what we present is especially novel. However, there seems to be a perception in the general Aotearoa NZ conservation community that translocations are relatively easy and success is assured, something not demonstrated by data on success rates either in Aotearoa NZ (Miskelly & Powlesland 2013), or internationally (Griffith et al. 1989; Wolf et al. 1996; Fischer & Lindenmayer 2000). The frequency of conservation translocations is also increasing (Cromarty & Alderson 2013), including calls for urban translocations (van Heezik & Seddon 2018), and translocations are likely to increase with predator-free Aotearoa NZ successes. Furthermore, the quality of translocation proposals processed by DOC is highly variable, with some poorly written, poorly thought out, or just a bad idea for the candidate species. The DOC approval process itself also produces variable outcomes. Therefore, our goal is to encourage careful thinking in the formulation of conservation translocation objectives, and the derivation of appropriate performance measures for these objectives, that align with aspirations for a predator-free Aotearoa NZ. We discourage a focus on any single element of planning and rather encourage all people involved in conservation translocations, particularly decision makers, to explicitly recognise the multiple values-based objectives associated with conservation translocations. The feasibility and timeframes over which predator-free objectives can be met are uncertain. Regardless, we want more native and endemic wildlife and fewer pests in Aotearoa NZ. To this end we anticipate this review being of utility to conservation scientists, managers, treaty partners, decision makers, community based practitioners, and all others interested in these lofty objectives.

The cultural and social setting of translocations

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

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

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

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

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

Setting objectives

Ewen et al (2014) characterised a conservation translocation as a sequence of decisions, and argued that poor planning, implementation, and monitoring is a consequence of not approaching the decision-making process in a deliberate and rational manner. They, along with several other authors, advocate a more structured approach to decision making (Maguire 1986; McCarthy et al. 2012; Converse et al. 2014; Ewen et al. 2014). Structured decision making is an iterative process whereby uncertainty is addressed by 1) defining clear objectives and how they will be measured; 2) identifying a range of possible management alternatives; 3) predicting the outcomes of the chosen management alternatives relative to the stated objectives; 4) evaluating trade-offs and uncertainty; 5) implementing the optimal management alternative and monitoring its results (Figure 1) (Gregory et al. 2012; Ewen et al. 2014). This approach to decision making has been characterised as "*a formalisation of common sense for decision problems which are too complex for informal use of common sense*" (Keeny 1982). Conservation translocations seem deceptively simple, but as noted usually consist of a mix of biological and non-biological values that are not necessarily equal, and in some cases might be competing with each other. Therefore, careful formulation of measurable objectives provides an effective and transparent way to make choices and signal success (Ewen et al. 2014). This approach is especially valuable in pursuing the aspirations of a predator-free Aotearoa NZ because, while one objective might seem simple, i.e. reduce or remove pests, this desire is actually deeply entwined with governmental, Treaty partner, community, and individual objectives that for many, the authors included, translate to a landscape dominated by native and endemic species. Therefore, a means objective (remove or reduce pest populations to low density) is being confused with the fundamental objective expressing what we really want (more native and endemic species). This directly relates to setting objectives for conservation translocations as we move beyond translocations to typical sites (islands and relatively small protected mainland areas), towards release sites with much more uncertainty, e.g. very large contiguous areas of habitat (1000s of hectares), and urban (van Heezik & Seddon 2018) and rural landscapes.

Understanding the extirpation history and the outcomes of previous translocations of a particular species, to a given release site, and/or sites with similar characteristics, along with relevant non-translocation work and theory (e.g. on dispersal) is an obvious start point for addressing uncertainty and setting informative performance measures for achieving the objectives we have for any particular translocation. As an example some species, such an NI robins, have persisted on the mainland, including at sites with no predator management whereas others, such as NI tīeke, have been extinct on the mainland for >120 years. Therefore, these two species clearly show very different levels of vulnerability to pests and will require different performance measures for pest control (a means objective) for a translocated population to establish and persist (a fundamental objective). In assessing the outcomes of previous translocations we recommend examination of factors likely to have influenced project outcomes (e.g. predation, dispersal pathways, vegetation associations, pathogens) in setting performance measures but note that it can be extremely difficult to determine why a translocation fails. One way is to model vital rates from another species to model the focal species vulnerability to pests. For example Parlato and Armstrong (2018) used NI robin data to predict rat tracking indices that might correlate with NI tīeke translocation success. Alternatively, factors other than pests might lead to translocation failure. For instance, of nine korimako/bellbird translocations (Miskelly & Powlesland 2013) only one (to Mana Island) appears to have successfully established a breeding population. While several factors might have contributed to these failures it is unequivocal that dispersal from the release site has been a critical factor, even at sites where some breeding has occurred (for example, Zealandia). Given such low success it is questionable whether any further translocations of bellbirds are justified, particularly given the low threat ranking of bellbirds (Miskelly et al. 2008), and their ability to naturally recolonise protected sites (Brunton et al. 2008), unless there is a significant change in methods or understanding. Clearly, if a species has been translocated only a few times, or not at all, then the outcomes of previous translocations are not useful indicators of future outcomes. In these cases, the translocation of other species, along with the ecology and conservation history of the target species, will have to be assessed against vulnerability to pests, dispersal abilities and other habitat requirements. However, there will naturally be a higher degree of uncertainty regarding establishment and persistence of the translocated population.

The release site

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

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

Pest control

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

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

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

Other biological and physical habitat variables

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

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

Habitat connectivity and dispersal

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

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

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

Another option for mitigating the impact of dispersal in the early stages of establishment is the release of large numbers of individuals, either in one big release or over several years. This is intuitively appealing but is rarely effective because if initial post release dispersal is a problem then dispersal will likely remain a problem via natal dispersal (Richardson et al. 2014). In addition, there are many examples where relatively large numbers of animals have been released but the translocations have failed (Miskelly & Powlesland 2013), despite release into habitats that should enable persistence once established (Armstrong & Seddon 2008). For instance, popokatea/whitehead translocations have been successful to many sites with founding populations of 40-100 birds. However, at one large (c.17 000 ha) contiguous site in the Waitakere Ranges with a protected block of 2450 ha, 653 birds were released over 12 years in an effort to compensate for post-release dispersal. In stark contrast to isolated sites up to 3300 ha in size, the Waitakere translocation is showing few signs of success (K.A. Parker, *unpublished data*). In addition, the true relationship between release group size and establishment is unclear (Armstrong & Wittmer 2011). This is because high quality sites where translocations are successful following the release of large numbers of animals could have been equally successful if fewer animals were released (Armstrong & Wittmer 2011). In contrast, managers typically release fewer animals when they have less confidence in a site, creating a reporting bias towards success with larger releases (Armstrong & Seddon 2008; Armstrong & Wittmer 2011). There are also significant welfare, ethical and relational risks around translocating large numbers of animals with the expectation that many will die following translocation, especially where translocation is not essential for species management. This uncertainty needs to be carefully and openly considered and discussed at the policy level, so that decision makers can make good defendable decisions at a national level, and with all Treaty Partners and stakeholders involved in any given translocation project.

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

Matching source populations to the release site

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

How similar are the source and release sites?

Does the source site share similar habitat characteristics especially the presence or absence of pests, vegetation associations and pathogens? This is not necessarily critical because, as noted above, some species seem to be quite tolerant of contrasting habitats. However, even within these species, translocation between similar habitats is likely an easier transition than translocation between contrasting habitats. For instance Parlato and Armstrong (2012, 2013) showed that translocation of NI robins between habitats with similar pest assemblages and vegetation associations had a small advantage over those between contrasting habitats. The similarity of the source and release site, the objectives of the translocation, and especially the risk profile or level of uncertainty associated with the translocation will also influence decisions about health screening. For example, translocation between two mainland sites and/or inshore islands that are relatively close together likely represents a low pathogen risk because their pathogen communities are likely to be similar. In contrast, translocation between distant sites with different habitats might prompt a more considered approach, especially if the recipient site has resident populations of highly valued species that could be put at risk through the introduction of novel pathogens. Ideally, there is also an understanding of potential pathogen impacts on the translocated species, on conspecifics and heterospecifics at the release site, and/or a documented history of health screening (Parker et al. 2006; Ewen et al. 2007; Ortiz-Catedral et al. 2011; Ewen et al. 2012; Massaro et al. 2012) to inform decisions about health management. Unfortunately this information is usually lacking or of poor quality.

Managing genetic diversity

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

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

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

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

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

The future of conservation translocations in Aotearoa New Zealand

Conservation translocations are likely to play an increasing role in Aotearoa NZ conservation. Ongoing practice and research will deepen our understanding of the values driving translocations including, but not limited to, societal desires, cost, animal welfare, genetic, and pathogen management, translocation techniques and dispersal. However, in Aotearoa NZ the biggest opportunities will come about through improved control of pests over large unfenced areas of the mainland, including forests, wetlands, dryland and braided river systems, and alpine zones. This will provide a means to translocate species that are currently in higher threat categories, along with providing further options for management and translocation of all species, especially habitat specialists, such as whio, kāki/black stilt, and pīwauwau/rock wren (*Xenicus gilviventris*), and neglected taxa, such as lizards, amphibians, invertebrates, and threatened plants. While opinion varies on the feasibility of effective pest control over vast swathes of Aotearoa NZ (Urlich 2015) it will clearly be a game changer if it can be achieved. We also expect to see an increasing shift away from conservation translocations for single-species recovery toward those where the fundamental objective is ecosystem restoration (Parker 2013). Pathogens and predators, such as weka, small rails (*Rallus* spp.), crakes (*Porzana* spp.) and NZ karearea/falcons (*Falco novaeseelandiae*) are obvious components of NZ ecosystems that are currently either actively avoided in restoration plans, or relegated to some point in the distant future once their potential prey or host species are well established. It seems logical to stage restoration sequences such that prey species are established before predators, although it is important to distinguish between a pest, against which native and endemic species have few defences, and a native or endemic predator that they have co-evolved with over 10000s of years. For example, translocated Middle Island tusked wētā (*Motuweta isolata*) and wētāpunga (*Deinacrida heteracantha*) have established in the presence of very high densities of a natural predator, the NI tiēke, whereas pests caused the extinction of many large wētā populations elsewhere. Therefore, conservation translocations of predators will require acceptance that there will be ongoing predation, possibly a reduction in population size, and changes in the behaviour of prey species. This will be difficult for some people to accept and could become problematic for very small prey populations, but it is a logical objective for true ecosystem restoration. It might also require flexible thinking in the management of predator species, and pathogens, especially where there is a management need or perception that natural predators and pathogens have to be controlled.

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

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

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

Another interesting proposition is the suitability of translocating close relatives of extinct species as ecological replacements in ecosystem restoration (Atkinson 1988). For example, the Snares Island snipe (*Coenocorypha huegeli*) was translocated to replace the extinct South Island snipe (*Coenocorypha iredalei*), the North Island kōkako was translocated as a replacement for the presumed extinct South Island kokako (*Callaeas cinerea*) and South Island takahē (*Porphyrio hochstetteri*) are frequently translocated to the North Island (Jamieson & Ryan 2001; Parker et al. 2010b; Miskelly & Powlesland 2013) (although takahe translocations are motivated by species recovery goals rather than as a replacement for the extinct mōho, or North Island takahē (*Porphyrio mantelli*)). It has also been suggested that the Australian brown quail (*Synoicus ypsilophorus*) is a suitable ecological replacement for the extinct New Zealand quail (*Cotunix novaezelandiae*) (Parker et al. 2010b). These species, and others, might be useful for restoring ecosystem services, known or otherwise. In addition, genetic techniques are advancing to the point where de-extinction, the resurrection of functional proxies of extinct species, might become feasible (Seddon et al. 2014; Seddon 2017). This is a contentious issue and the objectives of any such proposal will have to be very carefully considered, including the opportunity cost of diverting funds from extant species to de-extinction proposals (Bennett et al. 2017).

Conclusions

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

Acknowledgments

We dedicate this paper to our colleague, mentor and friend, Ian G. Jamieson. We are grateful to the Biological Heritage National Science Challenge Project 1.4 team including Thomas Buckley, Levi Collier-Robinson, Rod Hitchmough, Michael Knapp, Roger Moraga and Anna Santure for robust dialogue on characterising adaptive variation for conservation translocations. We are also grateful to Jana Wold for her constructive feedback and text relating to chromosomics. Susan Walker provided helpful comments on earlier versions of this paper. KAP, ZS and DPA were partially supported in this project via the MBIE funded "More Birds in the Bush" project.

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

1. <u>All conservation translocation decisions are values based</u>. 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?

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

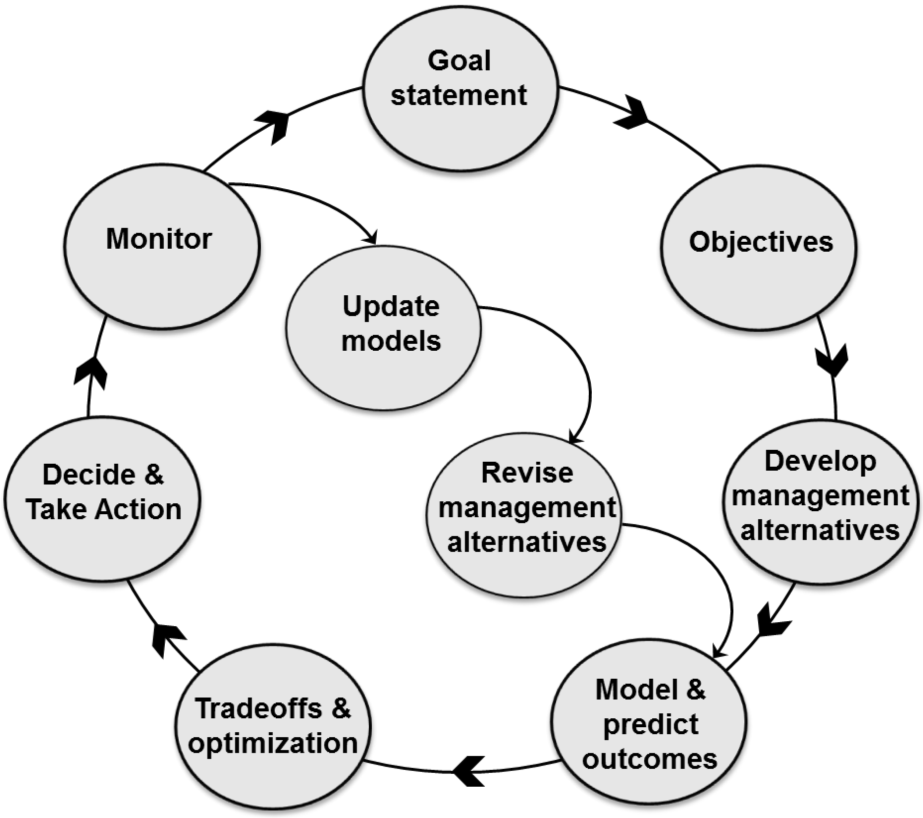

Figure 2. A hypothetical relationship between expected population equilibrium density and habitat connectivity mediated dispersal following translocation. The grey areas with solid black lines are managed habitat. Those surrounded by dashed lines are unmanaged. The light stippled area surrounding the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

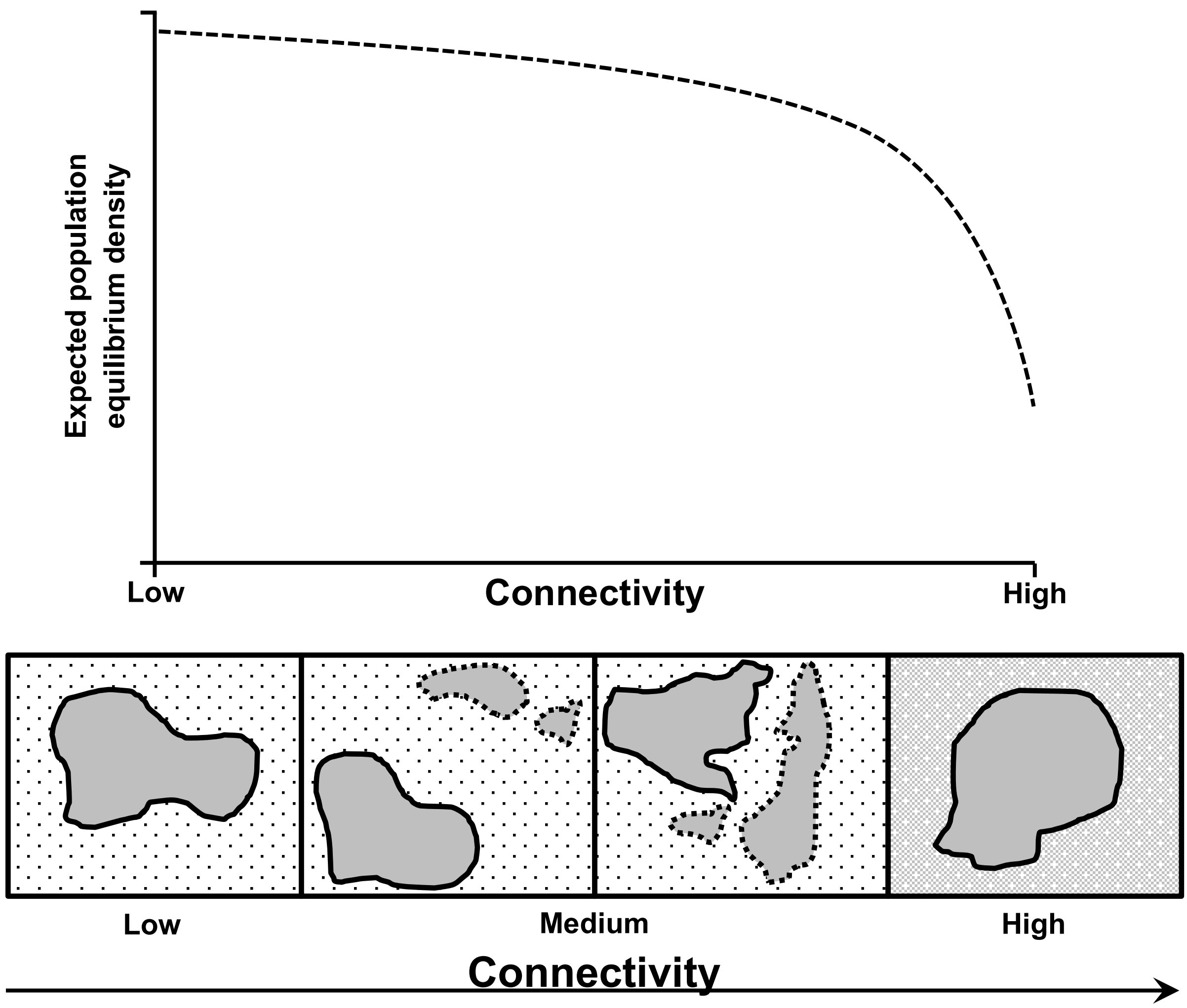

Table 1. Known or probable pest control thresholds, extinction history, and key uncertainties, for some terrestrial species that might be translocated in Aotearoa NZ. Knowledge is patchy, even for many bird species, and there is a lot of uncertainty to resolve, especially for herpetofauna and invertebrates. 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?

Multi-species pest control to low density, typically including ship rats, mustelids, possums and cats, sometimes including ungulates and pigs. Mice usually present, sometimes at high density Control is sometimes delivered seasonally (e.g. over the bird breeding season)	Robin spp.	Extinct across most of their natural range Persisting at some unmanaged mainland sites Stable/increasing at managed sites	High	?	Low	Density is highly variable at managed sites, likely due to climate and vegetation associations
	Yellow crowned kākāriki	Extinct across most of their natural range Persisting at some unmanaged mainland sites Stable/increasing at managed sites	High	High	?	Suitable source populations (logistically and genetically)
	Whiteheads	Extinct across most of their natural range Persisting at some unmanaged mainland sites Stable/increasing at managed sites	High	Moderate	Low	
	Mohua	Extinct across most of their natural range Present at some unmanaged mainland sites Stable/increasing at managed sites	High?	?	Low	
	Rifleman	Extinct across most of their natural range Persisting at some unmanaged mainland sites Stable/increasing at managed sites	?	?	Low	
	Kākā	Extinct across most of their natural range Present at some unmanaged mainland sites	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 Stable/increasing at managed sites	High	?	Low	Availability of birds, i.e. balancing community desires with national recovery objectives
	Short-tailed bats	Extinct across most of their natural range Stable/increasing at managed sites	High	?	?	Successful translocation techniques have not been developed
	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.

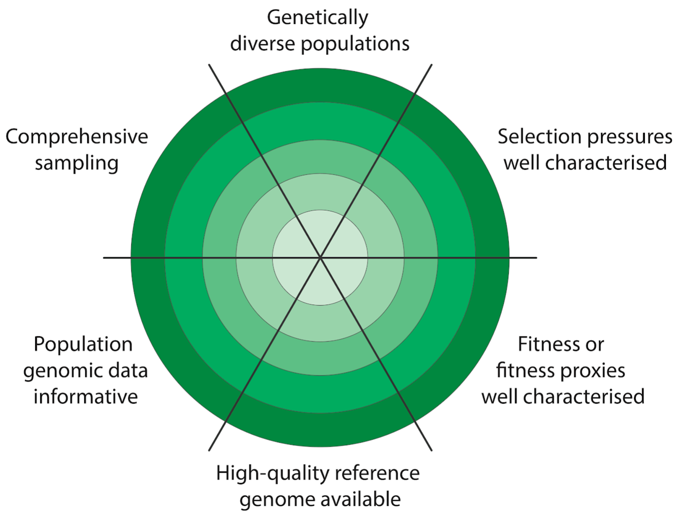

Figure 3. A novel framework for assessing key criteria for characterising adaptive variation in threatened species, including whether (i) populations are sufficiently large and genetically diverse to differentiate between selection and genetic drift; (ii) differential selection pressures are well characterised; (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).