



Considering Extinction of Dependent Species during Translocation, Ex Situ Conservation, and Assisted Migration of Threatened Hosts

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Abstract: *Translocation, introduction, reintroduction, and assisted migrations are species conservation strategies that are attracting increasing attention, especially in the face of climate change. However, preventing the extinction of the suite of dependent species whose host species are threatened is seldom considered, and the effects on dependent species of moving threatened hosts are unclear. There is no published guidance on how to decide whether to move species, given this uncertainty. We examined the dependent–host system of 4 disparate taxonomic groups: insects on the feather-leaf banksia (*Banksia brownii*), montane banksia (*B. montana*), and Stirling Range beard heath (*Leucopogon gnaphalioides*); parasites of wild cats; mites and ticks on Duvaucel's gecko (*Hoplodactylus duvaucelii*) and tuatara (*Sphenodon punctatus*); and internal coccidian parasites of Cirl Bunting (*Emberiza cirlus*) and Hibi (*Notiomystis cincta*). We used these case studies to demonstrate a simple process for use in species- and community-level assessments of efforts to conserve dependents with their hosts. The insects dependent on Stirling Range beard heath and parasites on tigers (*Panthera tigris*) appeared to represent assemblages that would not be conserved by ex situ host conservation. In contrast, for the cases of dependent species we examined involving a single dependent species (internal parasites of birds and the mite *Geckobia naultina* on Duvaucel's gecko), ex situ conservation of the host species would also conserve the dependent species. However, moving dependent species with their hosts may be insufficient to maintain viable populations of the dependent species, and additional conservation strategies such as supplementing populations may be needed.*

Keywords: assisted colonization, coextinction conservation planning, extinction risk, introductions, managed relocation, parasites, plant–insect interactions, translocation

Consideración de la Extinción de Especies Dependientes Durante la Conservación Ex Situ y Migración Asistida de Hospederos Amenazados

Resumen: *La translocación, introducción, reintroducción y migraciones asistidas son estrategias para la conservación de especies que cada vez atraen más atención, especialmente ante el cambio climático. Sin embargo, raramente se considera la prevención de la extinción de un conjunto de especies dependientes cuyas especies hospederas están amenazadas, y los efectos del traslado de hospederos amenazados no son*

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claros. Debido a esa incertidumbre no hay orientaciones publicadas sobre como decidir si se mueven especies. Estudiamos el sistema dependiente-hospedero de 4 grupos taxonómicos diferentes: insectos en *Banksia brownii*, *B. montana* y *Leucopogon gnaphalioides*; parásitos de felinos silvestres; ácaros y garrapatas en el gecko *Hoplodactylus duvaucelii* y la tuatara *Sphenodon punctatus* y parásitos coccidios internos de *Emberiza cirulus* y *Notiomystis cincta*. Utilizamos estos estudios de caso para demostrar un proceso simple para el uso de evaluaciones a nivel de especies y poblaciones de los esfuerzos para conservar dependientes y sus hospederos. Aparentemente, los insectos dependientes de *Leucopogon gnaphalioides* y los parásitos del tigre (*Panthera tigris*) representaron ensambles que no pudieron ser conservados por conservación *ex situ* de hospederos. En contraste, para los casos de especies dependientes que involucran una sola especie dependiente (parásitos internos de aves y el ácaro *Geckobia naultina* sobre *Hoplodactylus duvaucelii*), la conservación *ex situ* de la especie hospedera también conservaría la especie dependiente. Sin embargo, mover a las especies dependientes con sus hospederos puede ser insuficiente para mantener poblaciones viables de especies dependientes, y se pueden requerir estrategias de conservación adicionales como la suplementación de poblaciones.

Palabras Clave: coextinción, colonización asistida, introducciones, interacciones planta-animal, parásitos, planificación de la conservación, riesgo de extinción, translocación, traslado manejado

Introduction

The potential global loss of biological diversity through coextinction (i.e., extinction of parasites and symbionts when there is a change in the host population size or the host species becomes extinct) is high (Dunn et al. 2009; Moir et al. 2010). Unfortunately, the extinction of many dependent species may also occur as a result of conservation efforts directed at their hosts. Although many conservation programs aim to reduce probabilities of extinction of host species (Millennium Seed Bank Project, Global Strategy for Plant Conservation, World Zoo and Aquarium Conservation Strategy), few consider the conservation of the suite of symbionts and parasites of those hosts.

Some dependent species are highly specific to their hosts (Vesk et al. 2010; Moir et al. 2011) and may occur in only a few populations within the host's range (Miller et al. 2007; Taylor & Moir 2009). These dependent species may, in turn, act as hosts to other dependent species (e.g., at least one nematode species is dependent on each gall-forming fly on Myrtaceae hosts [Taylor & Davies 2008]).

Species dependent on threatened hosts may become extinct through either direct or indirect human action (Moir et al. 2010). Some conservation programs include direct efforts to rid the targeted host of dependent species (e.g., endo- and ectoparasites) that may be detrimental to its health, for example eradication of oestrid flies from mammals (Colwell et al. 2009), insect herbivores from endangered thistles (*Cirsium pitcheri*) (Bevill et al. 1999), and entire parasite assemblages from red wolves (*Canis rufus*) (Phillips & Scheck 1991).

In contrast, indirect extinction of dependent species may occur when *ex situ* conservation of host species at a given life stage excludes the dependent species. For example, storing seeds of threatened plants and eggs of threatened birds or fishes provides no means of conserving external phytophagous insects or parasites, re-

spectively. Indirect extinction also occurs when the host is maintained in microclimates or microhabitats that exclude the dependent species (e.g., plant tissue culture) or are beyond the dependent species' and hosts' historical ranges (e.g., in zoos, botanical gardens, and translocated populations) (Moir et al. 2010). Although possible indirect extinction of dependent species is rarely acknowledged, these extinctions are an implicit cost of preventing extinction of the host species. Failure to recognize the number of dependent species that may become extinct when their hosts are conserved reduces one's ability to evaluate trade-offs among conservation strategies. We aim to support the identification of conservation strategies that account for relations between host and dependent species.

Dependent Species as Conservation Targets

Conservation strategies for individual hosts can incorporate host assemblages. For example in Australia, any assemblage of species can be listed as a "threatened ecological community" (TEC) under the Environmental Protection and Biodiversity Conservation Act of 1999. Although most TECs listed under the act are plant communities (e.g., Barrett et al. 2008; Gibson et al. 2010), TECs also can be defined on the basis of animals when the habitat of a faunal assemblage is threatened (e.g., stygofauna in a particular aquifer or troglifauna in a particular underground void). We argue here that under some circumstances a similar designation could be given to fauna restricted to a particular threatened host species.

Host Conservation outside the Current Range

Movement of host species can occur naturally (e.g., in response to changes in climate) or through human

intervention (e.g., for conservation, horticulture, or agriculture). Here, we define *ex situ conservation* as any action outside a species' current range such as translocation, reintroduction to an area where a species was extirpated, assisted migration, relocation, captive breeding, and cultivation of botanical gardens. *Ex situ* conservation of threatened species is a well-established strategy to maintain or increase genetic diversity and to maintain or establish viable populations (Coates & Aktins 2001; van Winkel 2008; Volis & Blecher 2010). Much recent discussion has focused on the potential role of assisted migration in reducing the probability of extinction from climate change (e.g., Hoegh-Guldberg et al. 2008; Minter & Collins 2010). We aim to illustrate how *ex situ* conservation of threatened hosts may affect their dependent species and to provide a method to help identify possible management actions.

Ex situ conservation of hosts may increase the probability of persistence of a dependent species (e.g., an internal parasite) if it is moved with the host. Alternatively, the dependent species may be able to disperse naturally to the area newly occupied by the host if its geographic range is the same as the geographic range of its host (e.g., some insect species that feed on agricultural crops). *Ex situ* conservation of a threatened host species could cause extirpation of the dependent species through loss of access to the host in the wild. However, many *ex situ* conservation actions involve management of propagules or individuals with an objective of having minimal effect on the source populations. Maintaining dependent assemblages and species is as crucial as maintaining host species to preserve biological diversity and to avoid unforeseen undesirable effects on the host and other species (Dunn et al. 2009). For example, the loss of particular insects could result in cascades of extinctions if they perform ecological functions such as pollination (Menz et al. 2010). Furthermore, moving host species away from their native dependent species may increase the colonization ability of the host (via release from natural enemies), as is the case of tallow trees (*Triadica sebifera*) in Texas (U.S.A.) (Hartley et al. 2010) and cichlids (*Oreochromis niloticus*) in Panama (Roche et al. 2010).

Assessment of Actions to Conserve Dependent Species

To assess the effect of translocation and *ex situ* conservation actions for threatened hosts on their dependent species, with the ultimate goal of avoiding extinction of dependent species, we devised 2 assessment methods (Fig. 1). These methods have applications beyond dependent species of threatened hosts. For example, they could be used to assess the effects of host movement outside

the range of its native dependent species (e.g., Hartley et al. 2010; Roche et al. 2010).

The first step in the process of assessing the effect of *ex situ* conservation actions for threatened hosts on their dependent species is to ask whether a dependent assemblage is characteristic of a particular host species (Fig. 1a). Often these evaluations are made subjectively by experts, but a range of statistical methods exist for evaluating the composition of species assemblages where sufficient data are available (e.g., Clarke 1993; De Cáceres & Legendre 2009; De Cáceres et al. 2010). We did not consider the assemblage of dependent species further if it was not characteristic of a single host species. If the assemblage was characteristic of one host, then the composition of dependent assemblages of *in situ* and *ex situ* host populations was compared. A substantial difference in dependent species composition suggests the assemblages on the *in situ* host populations have not been reestablished by *ex situ* conservation of the host and their conservation may require additional actions (Fig. 1a).

Individual dependent species can be assessed with a similar process (Fig. 1b). Dependent species potentially restricted to one or more threatened host species are screened for presence on hosts being conserved *ex situ*. If the dependent species is absent, further actions may be necessary to conserve the dependent species (Fig. 1b).

We applied this assessment process to 4 case studies in which the taxonomic group, biogeographical range, host movement for conservation purposes, and completeness of inventory of dependent species differed. Our use of such widely disparate case studies suggests our methods may have broad application.

Case Studies

Insects and the Translocation of Threatened Plants in Southwestern Australia

The southwest of Western Australia has a high concentration of endemic plant (approximately 8000 species [Hopper 2009]) and invertebrate (e.g., Moir et al. 2009) species. Many plant species in this region have small, fragmented populations (Hopper 2009) and high probabilities of extinction (Coates & Atkins 2001). Historic threats were principally habitat fragmentation via clearing of vegetation for agriculture, but more recent stressors include salinization and waterlogging of habitat, colonization of non-native plants, the introduced fungal pathogen *Phytophthora cinnamomi*, changes in fire regimes (Barrett et al. 2008), and climate change (Hopper 2009). To decrease the immediate risk of extinction, translocations of 59 critically endangered and endangered plant species (all listed under national conservation legislation) in the region are underway. We considered 3

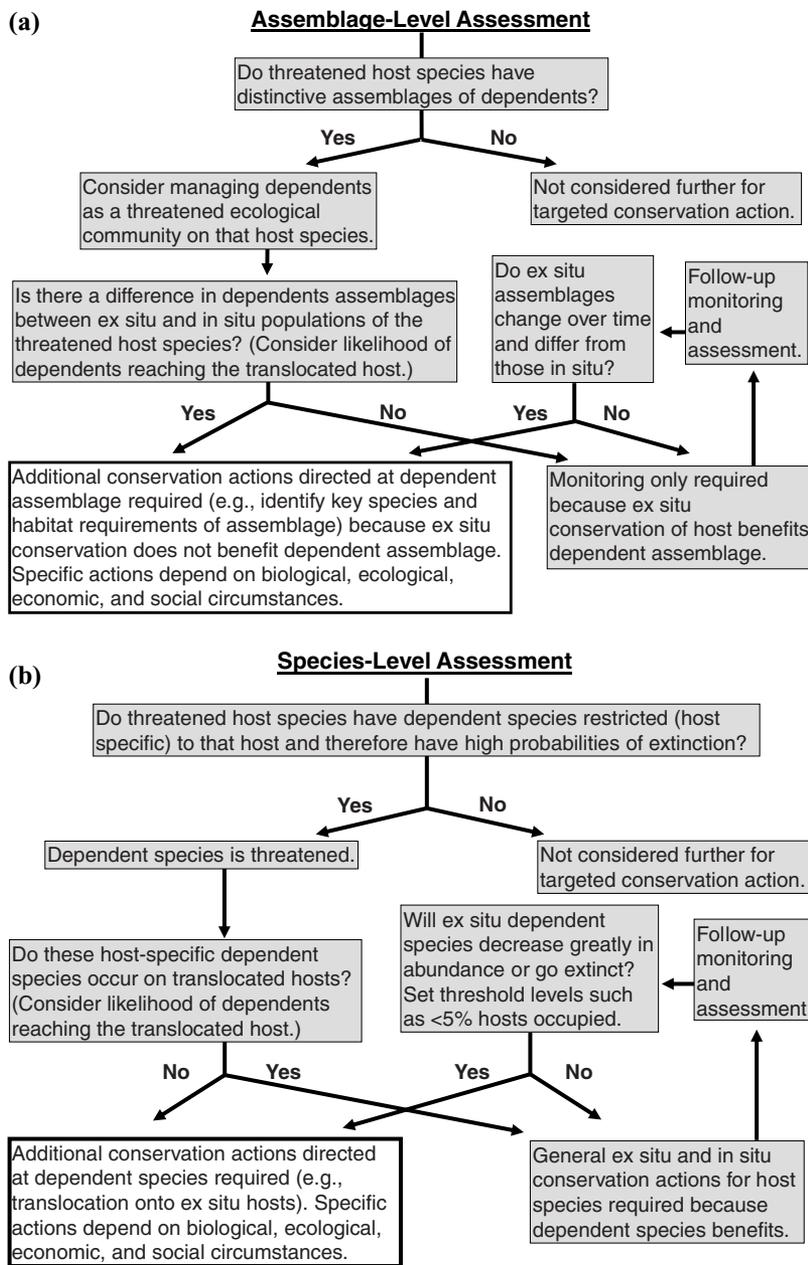


Figure 1. A method for assessing actions that may be necessary to maintain (a) dependent assemblages of species and (b) individual dependent species potentially threatened with extinction as a result of ex situ conservation of their host species. Distinctiveness of assemblages may be measured in various ways (e.g., by dissimilarity metrics, proportion of total population of most important indicator species in assemblage). In Australia any assemblage can be listed as a threatened ecological community under the Environmental Protection and Biodiversity Conservation Act of 1999. Sampling adequacy should also be considered because future sampling of other potential host species may reveal similar dependent assemblages or additional hosts for individual species.

translocated species: Stirling Range beard heath (*Leucopogon gnaphalioides* [Ericaceae]), feather-leaf banksia (*Banksia brownii* [Proteaceae]), and montane banksia (*B. montana* [Proteaceae]). We used a database of the insects that occur on these endangered plants, including in situ and translocated populations and insects on plants of related species, to assess the composition of their assemblages.

We applied the assemblage-level assessment (Fig. 1a) to the insects on in situ *B. brownii* and *B. montana*. Moir et al. (2012) found that the species composition of the assemblages was not significantly different from that on other closely related *Banksia* species (i.e., *B. quercifolia*, *B. concinna*, *B. plumosa*, *B. anatona*, *B. oreophila*);

thus, we did not consider these species further. In contrast, Moir et al. (2012) found that the composition of the insect assemblage on in situ beard heath differed significantly from that on other Ericaceae they studied. Thus, we proceeded to asking whether the dependent assemblages on in situ populations of the host differ from those on ex situ populations. The insect assemblage on in situ beard heath also differed significantly from assemblages on translocated plants (Moir et al. 2012). Translocation of beard heath, therefore, did not help the conservation of its in situ insect assemblage. We suggest this insect assemblage may warrant designation as a TEC and additional management actions (Fig. 1a & Supporting Information).

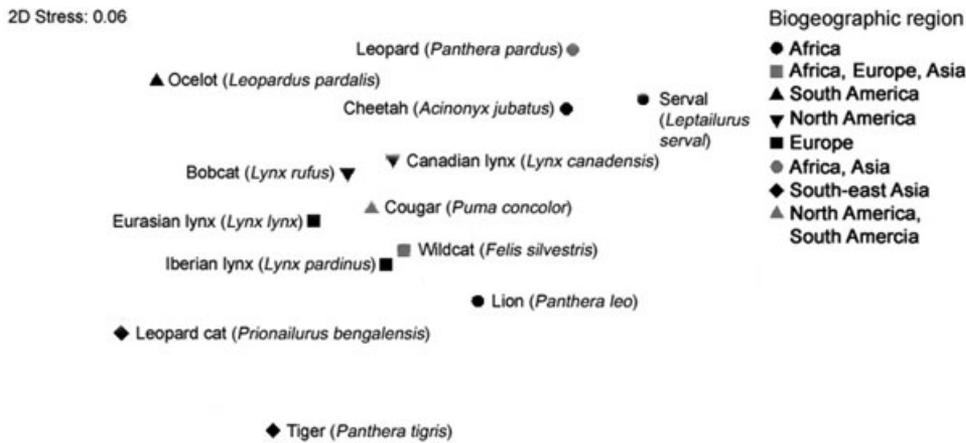


Figure 2. Multidimensional scaling (MDS) ordination of *Felidae* (cats) on the basis of presence or absence of their parasite species. The closer 2 points are together the greater the similarity in species composition of their parasite assemblages.

Despite Moir et al. (2012) finding that the insect assemblage of dependent species differed between in situ and translocated plants, no insect species were specific to beard heath; thus, we did not make further comparisons between in situ and translocated plants of this species (Fig. 1b). In contrast, one insect species on in situ *B. montana* and 3 insect species on in situ *B. brownii* appeared specific to these plant hosts after analysis with the protocols of Moir et al. (2011). These insects were not found on translocated individuals of *B. montana* or *B. brownii* (Moir et al. 2012). Our method suggests additional actions will be required to conserve these 4 insect species (Fig. 1b & Supporting Information). These conservation actions could include translocation with the host plants or captive breeding.

Parasites of Cats

We used data in the global mammal parasite database (Nunn & Altizer 2005) to examine parasite assemblages of cats (*Felidae*). The cats of the most conservation concern are the endangered tiger (*Panthera tigris*) and the critically endangered Iberian lynx (*Lynx pardinus*) (IUCN 2010). The latter has 2 remaining populations with approximately 200 individuals in total, and conservation strategies have included reintroduction into historic ranges in 2009 (Palomares et al. 2011).

We assessed parasite assemblages on cat hosts as we did for the insects on threatened plants. However, the database had numerous records of parasites identified only to genus, and it is unknown whether these are all distinct species. Thus, we assumed species identified to genus represented different undescribed species and included these in our analyses as separate species and conducted a second analysis with only described species.

Assemblages of parasites of cat species clustered biogeographically rather than taxonomically, although assemblages on *Lynx* species were loosely clustered within multidimensional scaling (MDS) ordinations (Fig. 2). The parasite assemblages of 3 cats in

Africa (leopard, serval, cheetah) formed a cluster, although hosts were members of 3 different genera (Fig. 2). The assignments of assemblages on different hosts to clusters were largely insensitive to uncertainty associated with undescribed parasites, except for assemblages on *Panthera bengalensis* and *P. tigris*. Data on described species are in the Supporting Information. We examined whether the parasite assemblage on Iberian lynx (Fig. 1a) was distinct from that on other cats. The results for all parasites, and only described parasites, showed that the parasite assemblage of Iberian lynx was similar to the parasite assemblage of the wildcat (*Felis silvestris*), a species that occurs in the same region (Bray-Curtis similarity for all parasite species = 0.61 and for only described species = 0.77) (Fig. 2). Therefore, the parasite assemblage may be conserved indirectly by efforts directed at the Iberian lynx, and need not be considered further (Fig. 1a & Supporting Information). The endangered *P. tigris* appeared to have a distinct assemblage of parasites (Bray-Curtis similarity <0.08 in all comparisons) (Fig. 2). This is, however, a preliminary assessment. Further studies are required to determine fully the consequences of host-based conservation efforts on any parasite assemblage of cat species for reasons outlined in the Supporting Information.

Ectoparasites on Geckos and Tuataras

We considered 2 species of reptiles and their ectoparasites that have been translocated together. Duvaucel's gecko (*Hoplodactylus duvaucelii*) is a lizard which was once widespread throughout New Zealand, but has declined greatly and is now restricted to only a few small islands offshore. It has been translocated to several other islands. The tuatara (*Sphenodon punctatus*) is a reptilian evolutionary relict (order Rhynchocephalia). This iconic and highly protected species was once distributed throughout New Zealand, but currently populations (4 of which were introduced) occur on 36 islands (Department of Conservation 2001). The tuatara is listed nation-

ally as requiring “urgent recovery work” (Brothers Island tuatara) or “work in the short term,” such as monitoring changes in population size to ensure that populations do not become locally extinct (Department of Conservation 2001).

DUVAUCEL'S GECKO

In November 2006, 39 geckos were translocated from Korapuki Island to Tiritiri Matangi (19) and Motuora (20) islands (van Winkel 2008). At the time of translocation, all geckos were hosts to mites (*Geckobia naultina*) that also occur on *Naultinus* geckos (McKenna 2003). However, *Naultinus* geckos do not occur on Tiritiri Matangi and Motuora islands. To determine the status of invertebrates dependent on Duvaucel's gecko, we followed the methods of Moir et al. (2012). The results of this protocol showed that further molecular and taxonomic analyses are required and may reveal the mite is restricted to Duvaucel's gecko. On the basis of a species-level assessment (Fig. 1b) in which we adopted the precautionary principle that the mite may be a new species that is specific to the gecko (Moir et al. 2012), the mite was assessed as threatened. In the next step of the assessment, we asked whether the mite occupies translocated geckos. Over the year following translocation, mites continued to occupy recaptured geckos, typically 5–10 mites per gecko (van Winkel 2008). Thus, the parasite was translocated with its host, and the mite may persist given conservation actions directed toward its gecko host (Fig. 1b & Supporting Information).

TUATARA

In 2003, 60 adult tuatara from Middle Island were introduced to Tiritiri Matangi Island (A. Cree & C. Smuts-Kennedy, personal communication). At the time of the translocation, the endangered, host-specific tick *Amblyomma sphenodonti*, occurred on many tuatara. Although occasionally reported on other species of reptiles, these occurrences appear to represent accidental associations because only single tick occurrences are detected among a large number of lizards (Heath 2006). Models that account for such incidental records (Vesk et al. 2010) suggest that the tick is host specific to the tuatara (Miller et al. 2007). In the species-level assessment we asked whether the tick persisted on translocated tuataras (Fig. 1b). After translocation to Tiritiri Matangi, ticks have persisted on the founder population, but only at very low abundances (A. Cree & C. Smuts-Kennedy, personal communication). Tuatara have also been translocated from Stephens Island to Karori Wildlife Sanctuary, a fenced reserve near Wellington. In late 2005, 70 adults were moved to the sanctuary, and they were recaptured every 5 weeks for 1 year after translocation (McKenzie 2007). At the time of translocation, ticks occurred on all 70 individuals (mean

of 26 ticks/host [SE 3.3]). One year later, ticks occurred on 22% of the tuataras (mean of 2/host [SE 1]) (McKenzie 2007).

In both these cases, tick populations apparently did not establish because tuatara densities were too low for successful transmission (see Miller et al. [2007] for correlations between tuatara densities and prevalence of ticks). Thus, although a species-level assessment at the time of translocation would not have suggested conservation actions for the tick, subsequent monitoring suggested that additional conservation actions directed at the tick may be necessary if tick abundance continues to decline (Supporting Information). On the basis of these translocations, we suggest that viable sizes of founder populations need to be determined for dependent species and that dependent-species populations need to be managed to ensure sustainable transmission rates between hosts.

Endoparasites in Birds

Coccidians (phylum Apicomplexa) are intracellular organisms that are obligate parasites of birds. At least 2 documented translocations of coccidians with their bird hosts exist. The first involves the Cirl Bunting (*Emberiza cirrus*), which is listed as a species of conservation priority in the United Kingdom (Natural England 2011). The abundance and range of this species has declined rapidly in the past century. The second example involves the Hiji (*Notiomystis cincta*), an endangered passerine with a few small populations in New Zealand.

CIRL BUNTING

In 2006, 75 Cirl Buntings were translocated from Devon to Cornwall. Juvenile birds were taken from their nests and reared in captivity prior to their release. Efforts were made during the captive period to prevent disease symptoms while maintaining natural infections of the coccidian *Isospora normanlevinei*, a common parasite of Cirl Buntings in other European populations (McGill et al. 2010). In the species-level assessment, we asked whether the coccidian persisted on translocated Cirl Bunting (Fig. 1b). The Cirl Bunting and its coccidian were both present at the translocation site in Cornwall. Six months after their release, 1 of 3 fecal samples from translocated birds contained *I. normanlevinei* oocysts (McGill et al. 2010). Thus, the parasite appeared to survive ex situ with its host and our assessment suggests that conservation action is not required for the coccidian (Fig. 1b & Supporting Information). Although *I. normanlevinei* benefited from conservation efforts directed toward the Cirl Bunting, the translocated population requires ongoing monitoring and reassessment (Fig. 1b) because the sample size was small (3 birds) and sampling occurred only 6 months after the initial translocation.

HIHI

In 1995–1996, 51 Hihi were translocated from Little Barrier Island to Tiritiri Matangi Island. Although only 21 of these birds survived to the following breeding season, the establishment of a new population was successful. In the species-level assessment, we asked whether the Hihi has any host-specific dependent species (Fig. 1b). Between 2004 and 2008, an undescribed coccidian, probably specific to Hihi, was recorded in both the source and translocated populations. Coccidians generally are host-specific parasites (Marquardt 1981). Consequently, the presence of coccidial oocysts in the feces of translocated birds is most likely due to translocation of the parasites with their hosts rather than post-translocation infections acquired from sympatric bird species. Therefore, in the species-level assessment, we asked whether the translocation of the Hihi benefited the coccidian (Fig. 1b). Fecal samples taken from 228 Hihi on Tiritiri Matangi on 6 occasions showed occurrence of the coccidian oocysts in up to 50% of samples, whereas in a single collection event of 27 fecal samples in 2004 on Little Barrier, 22% of samples contained coccidian oocysts (J. Ewen, personal communication). This suggests that the ex situ conservation of Hihi also benefited the coccidian. The benefit was confirmed through monitoring and assessment 12 years after translocation; Hihi fecal samples continued to contain coccidian oocysts (Supporting Information).

Managing pathogens is important in translocations of Hihi because Hihi previously have been threatened by disease (Ewen et al. 2007). However, parasites such as coccidians that do not negatively affect their hosts and that cannot easily be eliminated from the host are more likely to be translocated with their hosts than other types of parasites (J. Ewen, personal communication). Provided ongoing monitoring shows populations of birds and coccidians persist after translocation, both host and dependent species may be conserved simultaneously by the single conservation action (Fig. 1b).

Discussion

Effect of Global Ex Situ Conservation of Hosts on Dependent Species

Our assessment methods may help avoid extinction of dependent species and determine whether translocation and ex situ conservation of threatened host species may increase probabilities of persistence of dependent species. Empirical evidence suggests that the movement of host species reduces the diversity of assemblages of dependent species (case study of insects on plants). This result is consistent with that of nonthreatened hosts moved to new environments. Parasite assemblages of introduced peacock groupers (*Cephalopholis argus*) and cichlids are much less diverse than in individuals within their native environment (Vignon et al. 2009; Roche et al. 2010), as

are insect assemblages on introduced tallow trees (Hartley et al. 2010). Translocation programs often seek to conserve not only species, but also ecosystem function. For translocation programs that focus attention only on the host species, ecological functions associated with dependent species may be lost or reduced. For example, insect herbivores facilitate the cycling of nutrients by consuming and excreting plant material; loss of some herbivores from the insect assemblage may lower rates of nutrient cycling (Hooper et al. 2005).

Globally, 29 plants and 18 vertebrates considered extinct in the wild exist ex situ, predominantly in botanical gardens (12 plant species), domestic gardens (13 plant species), or captive-breeding programs (12 animal species) (IUCN 2010). Although it is difficult to estimate how many dependent species associated with these 47 host species are extinct, the likelihood of movement of dependent species with their hosts may be predictable from life-history characteristics. Endoparasites are most likely, ectoparasites moderately likely, and external parasites (e.g., mosquitoes, herbivorous beetles) least likely to move with their host species. Despite this, some of the surviving ex situ populations of hosts are likely to be too small to support sustainable populations of many dependent species.

For the plants that the IUCN lists as extinct in the wild but existing ex situ, each species could possibly host 1.3–10.6 monophagous insects (Fonseca 2009; Moir et al. 2012). Under this assumption, the removal of the 29 plant species from the wild (IUCN 2010) means it is possible that 37–307 monophagous insect species associated with these plants have already become extinct unless they were able to persist in ex situ populations of the hosts. If one were to extend this assumption, the 3,853 plant species listed as endangered or critically endangered by the IUCN (2010) may support 5,137–40,842 dependent monophagous insect species that may also be at high risk of extinction. Unless these species are explicitly included in the design of conservation strategies, they may become extinct despite any ex situ conservation or translocation techniques applied in isolation from their hosts (e.g., seed banks, introductions). Moreover, these are likely underestimates because many taxa and regions have not been assessed with IUCN methods (e.g., individual countries often list more threatened and extinct taxa than the IUCN). This lack of assessment includes most dependent species, particularly those with varying degrees of host specificity (Moir et al. 2010). Alternatively, if numerous dependent species are able to shift to alternative host species, then the number of extinctions could be overestimated (Moir et al. 2010).

Conservation of Species Dependent on Threatened Hosts

In some cases, given ongoing threats and rates of decline, the ex situ conservation of a host species may be one of

the few viable options available to prevent its extinction. There are some options for conserving threatened dependent species. Dependent species can be less host-specific in captivity (Whiteman & Parker 2005); thus, dependent species associated with rare host species could possibly use other hosts (Vesk et al. 2010). A dependent species released onto another host might not establish or it might displace other species, which may also occur when introducing species to a new location (Minteer & Collins 2010). Others suggest establishing parasite banks similar to the Malaria Parasite Bank of India (Whiteman & Parker 2005) and maintaining populations in zoos (Shirey & Lamberti 2010). Such measures may not be possible for dependent species with complex ecological requirements (e.g., those that require several host species during their life cycle) or for dependent species specific to highly endangered hosts (e.g., case study of endoparasitic coccidians in birds). Parasite banks and zoos are also expensive and are probably not an option for large numbers of species, especially when the translocated dependent species require ongoing monitoring and genetic management (e.g., Fischer & Lindenmayer 2000; Watts et al. 2008).

Few species in our case studies were identified as requiring further conservation action. Despite high failure rates of translocations and reintroductions of vertebrates (Fischer & Lindenmayer 2000), dependent species with simple life cycles and high host-specificity are good candidates for conservation via movement with their hosts. This is provided that monitoring and prerelease actions are undertaken to maximize the probability of successful translocation of the dependent species (New 1994). Our case studies, however, highlight that most threatened dependent species are undescribed (insects on plants, Hihii coccidians) or their taxonomy is uncertain (mite of *Duvaucel's* gecko). Furthermore, many dependent species may not be considered charismatic enough to warrant conservation attention. We believe this lack of taxonomic information impedes conservation of threatened dependent species. Assemblages of dependent species identified as TECs (e.g., case study of insects on Stirling Range beard heath) are more difficult to conserve given the number of species involved, the associated lack of taxonomic information, and the risk that they may not establish on translocated hosts. In situ conservation is, therefore, likely to be the most effective and cheapest option for maintaining these assemblages, except in situations where extinction of the hosts is imminent and some form of ex situ conservation is necessary.

We have demonstrated a practical means of assessing the status of dependent species and identifying potential actions to conserve host-dependent assemblages. However, we believe social (e.g., economic costs, cultural acceptability) and ecological (e.g., feasibility, effect on species at translocation site) factors should also be incorporated into the assessment process as they be-

come known (Richardson et al. 2009). Use of branching decision-support methods can hide uncertainty, which would be concerning when a definitive *yes* or *no* is impossible. This should not limit the use of such assessments because uncertainties can be incorporated into branching methods, for example, by application of fuzzy logic. We see our assessment methods as a first step in the conservation of dependent species and that the process of the assessment will develop over time. For example, the tuatara and tick translocation case study shows that founder population sizes need to be incorporated into the assessment process in some cases. We believe that if the information resulting from use of the assessment is used to adjust conservation strategies for the hosts, it could substantially increase the returns (i.e., number of species saved from extinction) on investments in conservation of threatened host species.

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Supporting Information

Data on parasite assemblages of cat species (Appendix S1) and illustrations of the assessment methods for the case studies (Appendix S2) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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