ASSISTED COLONISATION AS A CLIMATE CHANGE ADAPTATION TOOL

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Summary

This report synthesizes the current literature on assisted colonisation, a climate change adaptation strategy, in the context of terrestrial biodiversity conservation. Assisted colonisation introduces at-risk species to new habitats, beyond their current range, in anticipation of more suitable climates.

The translocation of flora and fauna for conservation purposes has been increasingly practiced in Australia as the threats to our native biodiversity have become better understood. A body of literature now exists on theory, practice, and results of translocation for some groups of organisms and individual cases. For the most part, conservation translocations in Australia have to date been in response to threat processes other than climate change, such as habitat loss or fragmentation. Recently, the IUCN/SSC included climate change as a specific threatening process that might drive translocation actions using assisted colonisation techniques. Unlike other conservation translocations, the practice of assisted colonisations is limited and therefore there are few examples from which to guide practice. To assist in the preparation of biodiversity conservation actions under climate change, this report reviews the substantial and growing body of literature that concentrates on the theoretical aspects of assisted colonisation. In particular, the report discusses the risks and benefits of assisted colonisation and traits that may render organisms candidates in need of this type of conservation action. It also reviews the threats and situations under which a species may become at risk of the need for assisted colonisation and factors related to successful conservation translocations.

Acknowledgments

The authors would like to acknowledge the contributions of staff within the NSW Office of Environment and Heritage in helping to develop this document, specifically Linda Bell and James Brazil-Boast.
ABBREVIATIONS

ANPC: Australian Network for Plant Conservation
BOM: Bureau of Meteorology
CSIRO: Commonwealth Scientific and Industrial Research Organisation
IPCC: Intergovernmental Panel on Climate Change
IUCN/SSC: International Union for Conservation of Nature Species Survival Commission
OEH: New South Wales Office of Environment & Heritage
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Table 3. Existing frameworks for assessing candidate species and situations for assisted colonisation.

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Appendix 1. Compilation of established flora translocations in Australia

Appendix 3. Reviews of flora translocations

Appendix 4. Reviews of fauna translocations
SECTION 1 - What is Assisted Colonisation?

1.1 Definition
Assisted colonisation is a type of conservation translocation; a form of introduction, where organisms are intentionally moved outside their indigenous range to avoid extinction of populations of the focal species (IUCN/SSC, 2013). Assisted colonisation is most commonly identified as a conservation measure to counteract the projected effect of climate change on species survival. The movement of organisms beyond their indigenous range is also commonly referred to as ‘managed relocation’, ‘assisted migration’ and ‘benign introduction’ (Seddon, 2010; Maschinski & Haskin, 2012).

Anthropogenically-induced climate change is predicted to engender changes in the distribution, phenology and abundance of species. Potential responses of at-risk species to rapid climate change are:

(1) Remain within the current range and persist unchanged or adapt in-situ without migration¹ (via adaptive genetic variation or phenotypic plasticity)
(2) Migrate to more suitable climes
(3) Remain within the current range, without adaptation, leading to extinction if climate exceeds physiological tolerance

Species that cannot adapt in-situ or migrate are at risk and are candidates for assisted colonisation. Although the decision to implement assisted colonisation can be controversial and therefore may not be adopted, it is important to note that a failure to assist species may favour those species in categories (1) and (2) above, whilst disadvantaging those in (3) (Thomas, 2011). None of the three scenarios listed above can be treated in isolation and often the question of when or which species should be considered for assisted colonisation, or the various design and

¹ ‘migration’ and ‘migratory’ are predominantly used here, unless otherwise stated, to relate to unidirectional movement in response to climate change, not to regular annual movement.
It is important to note that species that are adversely affected by climate change may undergo declines under any of these three scenarios, and that any increased risk of decline imposed by climatic factors may interact synergistically and unpredictably with other threatening processes. These singly or in combination may cause decline below recovery thresholds regardless of a species’ genetic or phenotypic potential to adapt or migrate successfully in response to climate-related factors. The suitability of candidate species for assisted colonisation needs to be assessed taking into account the full suite of actual or foreseeable threat factors.

Carefully considered experimental translocations in the near future, whether with pre-critical threatened species or with non-threatened analogue species, could help to increase the level of empirical knowledge, refine techniques, improve suitability assessment, and obviate at least some of the risks for the future.

1.2 Global changes in climate and implications for biological systems

This section gives a brief overview of the current science of climate change and implications for biological systems.

- The earth has warmed by ~0.85°C since the 1880s. Current greenhouse gas emissions are tracking the upper range of the IPCC projections, indicating that global mean temperatures are likely to increase by 2.6 – 4.8°C (compared to the 1986 – 2005 baseline) by the end of this century (IPCC, 2013).

- Changes in seasonal patterns of rainfall, an increase in the frequency and magnitude of extreme weather events and a rise in sea levels are predicted globally. The frequency and intensity of droughts, floods and wild fires are also expected to rise (IPCC, 2013).

- In some locations, current climate regimes may disappear altogether, creating novel or non-analogue environments (Williams, 2007; Dunlop et al., 2012). Novel climates have been projected to emerge primarily in the tropics and subtropics, whilst current regimes in poleward regions and tropical montane sites are projected to disappear (Williams, 2007). Estimates of the percentage of global land surface area which may experience novel combinations of climate variables by 2100 range from 12-39% (Williams et al., 2007).
Globally, on average, shifts to novel climates are projected to occur by 2047 (±14 years) under a high emission scenario, with this change expected to be sooner (approximately 22.5 years), if calculations are based on pre-industrial variability in climate alone (Mora et al., 2013).

- Isotherm migration will allow some species with the potential to disperse to track changes in climate regimes and stay within their optimal climate conditions. However, many species are predicted to be unable to keep pace with the velocity of climate change. Terrestrially, the pace at which the climate is changing has been calculated at 27.3 km/decade (Burrows et al., 2011). The pace of climate change is predicted to be slower in mountainous regions than in homogeneous topography, but this effect is dependent on latitude (Loarie et al., 2009).

- Broadly, species will face changes in four key areas as a consequence of climate change: (1) physiology, (2) distributional limits, (3) phenology, and (4) rates of microevolution and adaptation (Hughes, 2000).

- Species are already responding to the recent, relatively modest levels of climate warming (Parmesan & Yohe, 2003; Rosenzweig et al., 2008; Chen et al., 2011).

1.3 Climate change in Australia

- In Australia, by 2070, climate warming is projected to be in the range of 1 – 5°C above the climate of the baseline period of 1980 – 1999 (if global greenhouse gas emissions are within the IPCCs projected range) (CSIRO & Bureau of Meteorology, 2012). The number of hot days/warm nights and cool days/cold nights are predicted to increase and decrease respectively (CSIRO & BOM, 2012).

- Large natural variability in precipitation is expected, with a general drying trend in the southern areas during winter and over southern and eastern areas during spring. Droughts are expected to become more frequent in southern Australia, however intense rainfall events are still likely to occur (CSIRO & BOM, 2012).

- NSW encompasses wide latitudinal variation (29° - 37°S) and consists of heterogeneous landscapes. Future climate projections for eight regions of
NSW have been produced at a 50 x 50 km scale (Department of Environment Climate Change and Water NSW, 2010a). Regional future climate scenarios vary but are broadly in line with large scale predictions of higher maximum and minimum temperatures, change in rainfall patterns and a suite of corresponding ramifications (Department of Environment Climate Change and Water NSW, 2010a).

SECTION 2 - When is assisted colonisation most needed and how can candidate species be recognised?

2.1 Introduction

Determining candidate species for assisted colonisation is multi-faceted and should be approached using diverse viewpoints across relevant disciplines. The capacity for an organism or population to adapt to climate change depends on biotic factors (such as adaptive genetic variation, capacity for phenotypic plasticity, inherent rates of evolution), abiotic factors (such as the availability and connectedness of suitable habitat, climate or soil conditions) and on the interaction between abiotic and biotic factors. Assisted colonisation is likely to be most needed for species which exhibit a combination of traits that limit their ability to disperse or to adapt to changing climate and the associated habitat change.

It is important to recognise that individual species are likely to respond to changing climates at different rates, leading to changes in the composition of current communities or assemblages (Ackerly, 2003; Urban et al., 2012). The potential for reassembly of communities poses a set of distinct management challenges, which includes the need to identify and clearly define the role that multi-species translocations may play in attempts to retain current function into the future.

2.2 Situations likely to exacerbate the impact of climate change

Whilst climate change is a global phenomenon and novel environments are predicted to be common (Dunlop et al., 2012), projected changes will vary spatially and seasonally, and influence vulnerability accordingly. Position in the landscape will influence exposure to environmental changes such as temperature extremes,
precipitation increases or decreases, changes in stream flow (floods and intermittent drying), changing fire regimes, frost exposure and reductions in snow cover and sea level rise (inundation and salinity). Homogeneous landscapes, particularly areas with low-relief topography, are projected to be particularly vulnerable to climate change impacts due to a reduced number of topographically determined microrefugia in which organisms can persist (Dobrowski, 2011).

Vulnerability to climate change will also be exacerbated by interactions with pre-existing stressors. These stressors may include (Lindenmayer et al., 2010):

- native vegetation clearance;
- habitat fragmentation;
- altered land use practices;
- loss of habitat;
- altered natural disturbance regimes;
- competition and/or predation from introduced weeds, pests and pathogens;

In addition, stochastic events may also increase vulnerability to climate change by reducing the size of individual populations or the breakdown of meta-population dynamics (IUCN/SSC, 2013).

2.3 Identifying at-risk ecosystems

Some ecosystems and their constituent species will be more vulnerable to the impacts of climate change than others. These ecosystems may require more active management solutions, including assisted colonisation of at-risk species, to maintain function. Six of the 10 major terrestrial and marine ecosystems in Australia identified as having vulnerable ‘tipping points’ (disproportionate large changes in ecosystems caused by modest environmental changes), are present in NSW (see Table 1 overleaf; adapted from Laurance et al., 2011).
The ecosystems listed in Table 1 are diverse and therefore ranking the factors that have rendered them vulnerable may not be informative. However, narrow environmental envelopes and restricted distributions are a recurring theme. The key drivers of vulnerability to tipping-points across all ecosystems studied were identified as (1) extreme weather events and, (2) changes in water balance and hydrology (Laurance et al., 2011).

In addition to the broad ecosystems types identified in Table 1, specific environments are projected to be particularly exposed to the impacts of changing

<table>
<thead>
<tr>
<th>Ecosystem</th>
<th>Vulnerability factors</th>
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<tbody>
<tr>
<td>Elevationally restricted mountains</td>
<td>Often narrow environmental envelopes; geographically restricted distribution; many are near climatic thresholds</td>
</tr>
<tr>
<td>Coastal floodplains and wetlands – particularly wetlands adjoining coastal areas with high tidal amplitudes (5–13 m)</td>
<td>Rising sea levels; extreme weather events (e.g. storm surges); plant invasions</td>
</tr>
<tr>
<td>Wetlands and floodplains of the Murray-Darling Basin</td>
<td>Clearing of vegetation; habitat loss; fragmentation; sedimentation &amp; nutrient changes; rising temperatures and sea levels</td>
</tr>
<tr>
<td>Offshore islands</td>
<td>Restricted size, physical isolation; often narrow environmental envelopes; endemic biodiversity; species invasions</td>
</tr>
<tr>
<td>Estuarine wetlands (salt marshes and mangroves)</td>
<td>Narrow environmental tolerances; geographically restricted; proximity to dense human populations in coastal regions; fragmentation; reliance on a few key (framework) species</td>
</tr>
<tr>
<td>Temperate eucalypt forests – only parts of the geographic range</td>
<td>Habitat loss; fragmentation; reliance on ‘framework’ species; close proximity to humans; loss of key fauna; synergisms between weed invasions and fire</td>
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</tbody>
</table>
climate regimes in NSW. For instance, the alpine area of NSW is considered to be highly vulnerable to climate change due to changes in summer temperatures and snow cover (Hughes, 2003; Whetton et al., 2003). Temperate subhumid areas in north-eastern NSW have also been identified as likely to undergo significant ecosystem-level change due to warming, especially in winter, and changes to fire regimes that may lead to changes in vegetation structure and composition (Hughes, 2011; Shoo et al., 2012). The south west of NSW has also been highlighted as particularly vulnerable to the impacts of climate change due to a predicted decrease in rainfall (Department of Environment Climate Change and Water NSW, 2010b). Marine and freshwater ecosystems have also been recognised as acutely susceptible to climate change (e.g. coral regions, coastal fringe habitats and wetlands and the Murray-Darling Basin (Steffen et al., 2009; Hughes, 2011)).

2.4 Traits associated with species most likely to be affected by rapid climate change and in need of assisted colonisation

Individual species’ vulnerability to climate change will depend on its sensitivity (the potential to persist in-situ), exposure (the degree to which the physical environment will change) and adaptive capacity (persistence due to its ability to cope with microevolutionary change or dispersal) (Dawson et al., 2011; Foden et al., 2013). Species with the traits listed below have been identified as the most likely to be affected by rapid climate change (note that several of the traits listed are interconnected and should not be viewed in isolation):

- Species with **small effective population sizes** are likely to have reduced *genetic diversity*, a heightened risk of inbreeding depression and an increased vulnerability to demographic and environmental stochasticity, all factors that increase the risk of extinction (Hoffmann & Sgro, 2011). The effects of inbreeding depression are deleterious for reproduction and survival regardless of climate change (Frankham et al., 2010). However, the process of rapid climate change is thought to exacerbate the rate of reduction in genetic diversity and temperature stress has been demonstrated to heighten inbreeding depression in some species (Armbruster & Reed, 2005). Inbreeding depression also reduces the capacity for species to evolve and
therefore adapt to changing conditions (Frankham et al., 2010). The importance of recent demographic history to genetic variation (i.e. small populations having experienced gradual decline and/or subject to directional selection), should be treated differently to those experiencing rapid decline (e.g. bottleneck effects) (Frankham et al., 2010).

- Species with **long generation times** have slow replacement rates, fewer chances for genetic recombination and reduced opportunity to increase evolutionary responses to climate change than species which reproduce more frequently (Renton et al., 2012; Buckley & Kingsolver, 2012). In the case of the persistence of annual versus perennial plants under climate change, the outcome may be context dependant. For instance, annual plants have an increased likelihood of rapid adaptation to withstand *in-situ* climate change. However, annual plants have an increased vulnerability to unfavourable environmental extremes compared to perennials. Adult perennial plants may have an increased chance of persisting through climate extremes until conditions are favourable for seedling establishment (Peters & Darling, 1985).

- It is generally expected that adaptation to climate change may be more difficult for species with **narrow distributions** (narrow endemics) than those with broader habitat tolerances. Species with restricted ranges or limited distribution may not have sufficient genetic diversity to cope with changes affecting phenological events, thermal responses and/or resilience to stressful climatic conditions (Hoffmann & Sgro, 2011). Species with narrow ranges and small effective populations are more likely to have low genetic diversity and therefore reduced adaptive capacity (Pauls et al., 2013). Populations with narrow distributions that are also adapted to colder climates (either from high altitude or high latitudes) are at further risk (Pauls et al., 2013). Furthermore, narrow-range species with no environmental heterogeneity may no longer possess the phenotypic plasticity to cope *in-situ* with changing conditions (Hoffmann & Sgro, 2011). In contrast, widespread species, specifically plant and insect species, usually have sufficient genetic diversity to provide resistance to stressful climatic conditions (Hoffmann & Sgro, 2011). However, widespread species should not be overlooked as candidates for assisted colonisation because (1) migration barriers (geological or anthropogenic) may
prevent migration; (2) connectivity to new areas may not be feasible; (3) disruption to disturbance/obligate relationships may preclude persistence or colonisation in new areas; (4) differential rates of dispersal / migration and adaptation capacity may occur among the leading, central and lagging populations of widespread species (Zakharov & Hellmann, 2008; Mellick et al., 2012).

- The degree of vulnerability of **narrowly-endemic species** will be related to the extent of their distribution. For example, ‘organisms occupying a single lake, a single mountain top, an isolated mountain range or a single geological outcrop’ are at risk if they cannot be connected to suitable habitat (Thomas, 2011). High-elevation species may be adapted to cooler temperatures. For these species, such as the mountain pygmy possum (*Burrarays parvus*), there may be limited options for translocation when suitable climate space is not projected to occur under future climate regimes (Brereton et al., 1995). Of particular concern are high-elevation endemic plants that are components of the relictual Gondwanan rainforest, rare species in lowland rainforests and cloud forest species at the level of the cloud base (800-900m) and other regionally endemic montane fauna constrained by physiological & ecological traits (Shoo et al., 2013).

- **Specialist** species (those with a narrow range of climatic conditions, habitat or diet) are predicted to be more sensitive to climate change than generalists (Thuiller et al., 2005; Buckley & Kingsolver, 2012). Specialists can include those species that are depended on or have a **mutualistic relationship** with another organism. Under climate change, the alteration to distribution, phenological events and / or abundance of one species may negatively affect the other and trophic interactions may be disrupted (Bernazzani et al., 2012). Specific mutualistic relationships between plants and mycorrhizae may also become decoupled.

- **Higher trophic level** species will be disproportionately affected if their host or prey species shifts distribution under climate change, particularly if the interaction represents a highly specialised relationship (Thackeray et al., 2010).
**Changing biotic interactions**, other than those involving specialists and mutualisms, have been identified as an important proximate cause of population declines and local extinctions under climate change (Cahill *et al.*, 2013). It has been argued that the preservation of biotic interactions is one of the most important factors in the ability of species to migrate and to successfully colonise under changing climates (Hellmann *et al.*, 2012). Many examples of changes to biotic interactions as a result of current climate change have been identified including: increases in rates of insect herbivory (Blois *et al.*, 2013), changes in the outcome of competitive interactions in European dragonflies (Suhling & Suhling, 2013), and the various case studies outlined in Hellmann *et al.* (2012). Disequilibrium in ecosystems will also be created by differential response rates to climate change from the introduction of non-native weeds, pests and pathogens. Non-native species may become more abundant or competitive than native species if they are advantaged due to changed climatic conditions, altered life cycles and/or rising atmospheric CO$_2$ levels (Bernazzani *et al.*, 2012).

Increasing levels of atmospheric carbon dioxide (CO$_2$) will affect vegetation community structure and function with flow-on effects at the ecosystem level. The ability of different plant species to take advantage of available CO$_2$ differs according to their photosynthetic pathway. Generally, C$_3$ plants (usually woody species) are able to take advantage of higher concentrations of atmospheric CO$_2$ relative to C$_4$ plants (often grasses). Incursions into grasslands and grassy woodlands by C$_3$ plants are therefore expected (Hovenden & Williams, 2010). Increased growth rates vary considerably within taxa and depend on interactions with nutrients, temperature and rainfall. Increasing CO$_2$ will also reduce leaf nitrogen content and increase secondary metabolites, altering plant herbivore relationships and nutrient recycling processes.

Species close to their physiological limits are at risk from climate change. Currently, there is evidence that fauna is at risk from thermal stress affecting rates of locomotion and feeding, reducing food availability and changing biotic interactions (Buckley & Kingsolver, 2012; Cahill *et al.*, 2013). The relative importance of physiological tolerance to high temperature stress may develop
over time as temperatures and extreme weather events increase in frequency and magnitude (Cahill et al., 2013). For example, death from over-heating (hyperthermia) in flying foxes in NSW was observed during an extreme temperature event in 2002 (Welbergen et al., 2008). Growth, survival, reproductive rates and generation time for fauna and flora are expected to alter in response to increasing temperatures, but with substantial variation amongst species (Buckley & Kingsolver, 2012; Cahill et al., 2013). The degree of species' vulnerability is expected to be highest in the tropics (especially for insects and terrestrial ectotherms) due to limited seasonality offering a narrow thermal range (Deutsch et al., 2008). The risk to species of temperature stress lessens towards higher latitudes where species have broader thermal optima and are often living in conditions cooler than their optima (Deutsch et al., 2008).

Hydric limits apply to species sensitive to intermittently dry conditions or restricted to moist conditions if precipitation patterns change (Hoffmann & Sgro, 2011; Buckley & Kingsolver, 2012). In addition, the interaction of thermal and hydric stress may prompt a unique suite of responses. For instance, birds are predicted to suffer high mortality rates from hyperthermia where extreme heat events combine with water stress (McKechnie & Wolf, 2010). Different organisms may be vulnerable to hydric and thermal limits at different life stages. Seedlings, for example, are more likely to be vulnerable to dry conditions than adult plants (McDowell et al., 2008) and some larval insects may be more immune to heat stress than the adults (Buckley & Kingsolver, 2012). Different seasonal rates of climate change may give early season developers a longer exposure to growing seasons (Buckley & Kingsolver, 2012). Where changes arise in the timing of life-cycle events there is potential for disruption of predator/prey relationships, mutualisms, pollination and competition. Mismatches between organisms that rely on photoperiod cues and thermal activity may also occur (Buckley & Kingsolver, 2012).

- Bergmann’s rule states that there is a trend for mean body size in endotherms to decrease with decreasing latitude (interpreted as body size decreases with increasing temperature). There are many lines of evidence to
support this rule. For instance, Gardner et al. (2009) found a latitudinal shift in size for 6 out of 8 species of Australian birds; wing length has reduced over the past 100 years. However, it has been demonstrated that the magnitude and direction of size response varies between endotherms and may depend on the nature of the change in temperatures, water stress and extreme events (Gardner et al., 2011). McCauley & Mabry (2011) argue that organisms with larger body size are more likely to be long-distance dispersers and successful colonisers. These traits may render large body mass species less vulnerable to climate change and smaller body mass species more likely to suffer adverse in-situ impacts. Furthermore, small-bodied desert birds are predicted to be most vulnerable to dehydration and mortality on extremely hot days (McKechnie & Wolf, 2010). There is no clear consensus as to whether body size changes are a plastic or evolutionary response to climate change (Teplitsky & Millien, 2014). It has been argued that changes in body size for all animals and plants may also be a function of altered diet and precipitation, rather than temperature alone (Bickford et al., 2011).

- **Species lacking dispersal capability** will have limited capacity for shifting their range in response to a rapidly changing climate. For sessile organisms like plants, species with seeds that lack adaptations for dispersal, such as aerodynamic appendages (pappus, coma, wings) for wind dispersal or fleshy fruits to attract dispersal mutualists such as birds or bats (Leishman et al., 2000) may be disadvantaged. Dispersal-limitation can arise in species that have been exposed to stable conditions over long periods. This can lead to: (1) a lack of genes to code for new functions, such as a change in thermal tolerance; (2) DNA decay in genes that are functionally important; and/or (3) low levels of genetic variation (Hoffmann & Sgro, 2011). Species incapable of achieving long-distance dispersal are particularly vulnerable to the potential effects of shifting climatic conditions. However, quantification of long distance dispersal events is typically subjective and varies substantially among taxa.

- The ability of species to disperse or migrate may be constrained due to **physical barriers in the landscape**. Physical barriers can be created by geological factors (e.g. edaphic and/or topographic), as well as man-made pressures such as urban/agricultural land use. Species with poor dispersal
capacity will be at risk of extinction where habitat connectivity cannot aid their migration to more suitable climates or to micro-refugia.

Candidate species for assisted colonisation may have a combination of the above described traits, be involved in several biotic interactions, suffer from multiple anthropogenic stressors and may respond using numerous modes. For example, large-bodied taxa that are ecological specialists at higher trophic levels with long generation times, poor dispersal ability, low or delayed reproductive output occupying small geographic ranges, may be particularly at risk (Dawson et al., 2011).

Generalisations across taxa may not be feasible because different groups vary in the relative importance of their traits in determining exposure to climate change. For example, Table 2 compares the relative importance of a range of traits associated with vulnerability to climate change and highlights the limitations of taking a broad approach in assessing at-risk species.

Table 2. The relative importance of biological and environmental traits, in order of influence on vulnerability to climate change in birds and amphibians (adapted from Foden et al., 2013)

<table>
<thead>
<tr>
<th>Biological &amp; / or environmental trait</th>
<th>Birds</th>
<th>Amphibians</th>
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</thead>
<tbody>
<tr>
<td>Limited dispersal ability</td>
<td>Slow turnover of generations (poor ability to evolve)</td>
<td></td>
</tr>
<tr>
<td>Low reproductive output</td>
<td>Limited dispersal ability</td>
<td></td>
</tr>
<tr>
<td>Slow turnover of generations</td>
<td>Habitat specialist</td>
<td></td>
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<tr>
<td>Changes in mean precipitation</td>
<td>Changes in mean temperature</td>
<td></td>
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<tr>
<td>Changes in precipitation variability</td>
<td>Changes in temperature variability</td>
<td></td>
</tr>
<tr>
<td>Changes in mean temperature</td>
<td>Narrow temperature tolerance</td>
<td></td>
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<tr>
<td>Changes in temperature variability</td>
<td>Narrow precipitation tolerance</td>
<td></td>
</tr>
<tr>
<td>Intolerance of disturbance</td>
<td>Changes in precipitation variability</td>
<td></td>
</tr>
<tr>
<td>Narrow temperature tolerance</td>
<td>Disease (Interspecific interaction)</td>
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<td>------------------------------</td>
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<tr>
<td>Geographical barriers (poor dispersal ability)</td>
<td>Changes in mean precipitation</td>
<td></td>
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</tbody>
</table>

For birds and amphibians, low adaptive capacity (poor ability to evolve and disperse) and changes in environmental traits are relatively influential for assessing at risk species (Foden et al., 2013).

Predicted responses to climate change within taxon have been modelled for some species, including plants and butterflies, and these findings can be used to make some generalisations. For example, a recent study showed that the best performing predictors of plant persistence under climate change include effective fecundity, years to maturity and dispersal distance (Renton et al., 2012). Functional trait approaches have also been used to predict the migratory potential of a range of butterfly species. An observational study comparing the distribution of 48 butterfly species in Finland over two time periods demonstrated that the following traits were correlated with successful migration: wide distribution; mobile; large body; overwinter as adults; feed on woody plants at the larval stage and use forest edges as their main breeding habit (Pöyry et al., 2009). In addition, changes in the timing of butterfly life-cycles have been linked to diet type, generation time, overwintering stage and dispersal ability (Buckley & Kingsolver, 2012).

2.5 Existing approaches to assessing candidate species for assisted colonisation

Various frameworks exist for assessing the need for assisted colonisation in the literature. The complexity surrounding the question of when assisted migration is most needed and which species are at risk from rapid climate change suggests that a combination of decision-making tools may be necessary. Table 3 (overleaf) lists and describes appropriate frameworks that can be employed to assess candidate species.
Table 3. Existing frameworks for assessing candidate species and situations for assisted colonisation

<table>
<thead>
<tr>
<th>Framework</th>
<th>Description</th>
<th>Reference</th>
</tr>
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<tbody>
<tr>
<td>Planning &amp; implementing successful assisted colonisation</td>
<td>Questions to ask to maximize the success of assisted colonisation</td>
<td>Chauvenet et al., 2013</td>
</tr>
<tr>
<td>Vulnerability to climate change: exposure; sensitivity and adaptive capacity</td>
<td>Assess vulnerability of species or ecosystems to climate change with conservation responses</td>
<td>Dawson et al., 2011</td>
</tr>
<tr>
<td>Vulnerability to climate change: exposure; sensitivity and adaptive capacity</td>
<td>Assess vulnerability using biological traits of birds, amphibians and corals</td>
<td>Foden et al., 2013</td>
</tr>
<tr>
<td>Framework for recipient site selection</td>
<td>Supplementation of IUCN guidelines</td>
<td>Harris et al., 2013</td>
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<td>Criteria for ranking good and bad managed relocation proposals</td>
<td>Uses rare plants as case studies</td>
<td>Haskins &amp; Keel, 2012</td>
</tr>
<tr>
<td>Potential actions to prevent extinction or ecosystem collapse</td>
<td>Assess feasibility of species movements under possible future climate scenarios</td>
<td>Hoegh-Guldberg et al., 2008</td>
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<tr>
<td>Optimal timing for managed relocation</td>
<td>Focus on when to move a population where objective is to maximize population size</td>
<td>McDonald-Madden et al., 2011</td>
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<tr>
<td>Assessing the need to translocate dependent assemblages</td>
<td>Assemblage-level assessment</td>
<td>Moir et al., 2012</td>
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<td>Evaluation of individual cases of managed relocation</td>
<td>Assess ecological and social criteria and acceptability and feasibility from different stakeholders perspectives</td>
<td>Richardson et al., 2009</td>
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<tr>
<td>Decision tree describing conservation introduction problems (under climate change)</td>
<td>Framework to assist decision making on whether or not to introduce, the choice of candidate species</td>
<td>Rout et al., 2013</td>
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<tr>
<td>Framework for socially &amp; scientifically acceptable managed relocations</td>
<td>A series of ethical, policy, ecological &amp; integrated questions to support proposed managed relocation proposals</td>
<td>Schwartz et al., 2012</td>
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<td>Management actions</td>
<td>Identifies movement, evolutionary and <em>ex-situ</em> options</td>
<td>Shoo et al., 2013</td>
</tr>
<tr>
<td>Collection strategies for potential target species</td>
<td>Determines, prioritizes and develops collection strategies for plant species</td>
<td>Vitt et al., 2010</td>
</tr>
<tr>
<td>Vulnerability to climate change</td>
<td>Assesses vulnerability including the effects of regional and local factors, potential for evolutionary and ecological responses, resilience, active management remediation and feedback effects</td>
<td>Williams et al., 2008</td>
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</table>

In addition to the generalised frameworks presented in Table 3, there are several quantitative methods to assess individual species in most need of assisted colonisation and four methods are briefly discussed below.

- **Predictive models** (e.g. correlative or mechanistic species distribution models) combine various climate variables (e.g. both mean and extreme measures of temperature or rainfall) with species’ current known distributions to project their ranges under future climate change scenarios. A more thorough discussion on this topic is provided in Section 4.

- **Paleoecological records** can provide insights into how abundant and widespread organisms responded to past climate change and are used to infer likely responses to the projected novel environments (Blois et al., 2013). However, even where the paleoecological records imply that species were able to persist through periods of hostile climate they may not be directly
comparable to the current situation where a higher rate of change and a different habitat-fragmentation regime apply, and where mass extinctions are predicted (Moritz & Agudo, 2013).

- **Phylogenetic analyses** allow the testing of the hypothesis that closely related species will respond similarly to climate change. Where phylogenetic clusters are found to be associated with a particular response to changing climatic conditions, predictions of which clades will be most vulnerable can be made (Angert *et al*., 2011; Buckley & Kingsolver, 2012).

- **Functional traits** have been used to predict which species and communities are most vulnerable to climate change (Jiguet *et al*., 2007; Gallagher *et al*., 2012). The predictive power of traits for identifying at-risk species varies, depending on factors such as the position within the range where measurements were taken (core versus leading edge populations), position in the landscape (especially for elevationally restricted organisms), and projected exposure to climate change (Angert *et al*., 2011). However, no single trait-based approach is a consistently strong performer and results will vary based on both individualistic responses of species and the empirical limitations of each approach. Trait-based approaches perform better in predicting the likelihood of phenological changes in species as opposed to the extent and magnitude of potential range shifts (Buckley & Kingsolver, 2012).

**SECTION 3 – Potential risks and benefits of assisted colonisation**

### 3.1 Benefits of assisted colonisation

The primary benefit of assisted colonisation is the potential to prevent extinction of species whose persistence is threatened either directly, or indirectly, by rapid climate change. Proponents of assisted colonisation argue that a decision not to assist species threatened by climate change by deliberately extending their current range dooms candidate species to extinction (McLachlan *et al*., 2007; Hoegh-Guldberg *et al*., 2008; Thomas, 2011). A recent review of 50 peer-reviewed articles on assisted colonisation found that 60% of studies supported the validity of assisted colonisation as a climate change adaptation strategy (Hewitt *et al*., 2011). However, despite this majority, 20% of studies indicated no clear position on the effectiveness of assisted
colonisation for climate change adaptation and the review’s authors note that debate around the practice of assisted colonisation is intensifying, rather than trending towards resolution (Hewitt et al., 2011).

Scenarios under which individual species, species assemblages or entire communities may benefit from assisted colonisation to in response to climate change include:

1. **Loss of suitable climatic habitat within the current range.** In some circumstances levels of change may fall outside the natural climatic variability experienced by some species under both current conditions and through evolutionary time. There is evidence that some organisms are already shifting their ranges to track optimal conditions for growth and reproduction under the relatively modest levels of climate warming already experienced globally (Parmesan & Yohe, 2003; Rosenzweig et al., 2008; Chen et al., 2011). The response of species is likely to be highly idiosyncratic, leading to the potential for substantial reassembly of the composition of communities, with flow-on effects to ecological interactions and ecosystem services (Gilman et al., 2010; Lavegne et al., 2010; Urban et al., 2012). Assisted colonisation may be the only option available for species that cannot keep pace with species with which they have co-evolved and are dependent upon.

2. **The emergence of novel or non-analogue climates.** Projections for the emergence of novel climatic environments both globally and within Australia in a relatively short space of time (between 2050 and 2100) indicate that whole assemblages of species may be under threat and may need assistance (Williams, 2007; Dunlop et al., 2012; Mora et al., 2013). The large scale of these projections may make it difficult to prioritise species for assisted colonisation, however pre-emptive feasibility studies could aid in to identifying potential candidates or flag the need for multi-species translocations.

3. **Multiple stressors.** Many natural areas are now so anthropogenically-modified (with no end in sight to the modifications) that managing for conservation to historically referenced conditions may no longer be
achievable (Thomas, 2011). Anthropogenically-modified habitats are typically characterised by the presence of invasive species, alterations to and disruption of disturbance regimes and changes to soil, hydrological and other conditions, altering both biotic and abiotic interactions (Fischer & Lindenmayer, 2007). In addition, modified habitats generally preclude effective species’ migration and gene flow by fragmenting populations into a matrix of land-uses (Cushman et al., 2006). As a result of extensive alterations to natural systems Shackelford et al. (2013) argue that restoration goals should no longer aim to reflect historic or reference sites.

(4) Replacement of the loss of an ecological function/service. Assisted colonisation can allow for the replacement of species that have been lost from an ecosystem (and the corresponding loss of the role that species performs) and can thereby restore a similar ecological service and/or fill existing gaps in biological function (Hewitt et al., 2011).

3.2 Risks associated with assisted colonisation: lessons from invasion biology

Opponents of the use of assisted colonisation as a climate change adaptation strategy typically draw examples from the lessons learnt from invasive species research, including biological control methods. Parallels have been drawn between species moved beyond their natural range for assisted colonisation purposes and the negative impacts of invasive species on recipient communities, such as:

- Reduced native plant recruitment through impacts on colonisation-extinction dynamics (Yurkonis & Meiners, 2004);
- The disruption of key ecological interactions (i.e. plant-animal mutualisms) (Ricciardi & Simberloff, 2009);
- The loss of fitness and/or local extinction of populations due to potential hybridization and introgression of the target species with close relatives at or nearby the recipient site (Hewitt et al., 2011 and references therein);
- Unintentional introduction of novel pathogens or pests to ecosystems (Ricciardi & Simberloff, 2009 and references therein);

Therefore, the need to move species beyond their range to secure viable populations under climate change needs to be weighed against the potential non-
target effects on communities at recipient sites. However, whilst the risk of invasiveness and flow on effects from assisted colonisation are valid, it is important to note that candidate species often possess biological traits incompatible with the tendency to become invasive (e.g. poor dispersal, long generation times, low fecundity; this subject is further discussed in Section 2). The selection, refinement and application of appropriate Invasiveness Risk Assessment techniques needs to be an integral part of the identification and prioritisation of candidate species for assisted colonisation.

3.3 Evidence of native species becoming invasive

There is potential for species considered native to become invasive within their country or region of origin if introduced beyond their historical, indigenous range (Gallagher & Leishman, in press; Mueller & Hellmann, 2008). In Australia, over 500 species of native plants have been introduced to areas outside their documented native range (Randall, 2007) with species in the genus *Acacia* (wattles) offering the most compelling example of the potential for native species to become invasive. Ecological studies of *Acacia* populations in their native and introduced ranges, and in common garden experiments, show that the introduced populations often have a competitive advantage over other species due to factors such as lower seed predation (M. Leishman, unpublished data) and different microbial associations (Birnbaum et al., 2012). Species such as *Acacia saligna*, introduced into NSW from WA, have been identified as significant threats to species listed under the TSC Act in NSW (Coutts-Smith & Downey, 2006).

A study comparing within-country and between-country introductions in the United States as a proxy for the likely impact of assisted colonisation versus introduction of non-native organisms found that species moved as part of assisted colonisations were unlikely to become invasive. However, those that did become invasive could cause significant harm (Mueller & Hellmann, 2008). By comparing the proportions of invasive species of intra- and inter-continental origins, Mueller & Hellmann (2008) concluded that relatively small portion (15% of the 468 species studied) resembled assisted colonisations (i.e. were native species, introduced from within the United States). The lower risk for moving native species within the United States, as compared to introducing those from outside the country, was attributed to
the higher likelihood that close congeners of the species are already present and occupying established niches in the recipient habitat or ecosystem. Consequently, ecological control factors (e.g. predators, pathogens) are also more likely to be present at the site reducing the risk of invasion. In their study, plants were found to be least likely to become invasive, with fish and crustaceans most likely to become serious invaders (Mueller & Hellman, 2008).

Species introduced as biological control agents also have the potential to cause harm to non-target species, often with serious ecological impacts on native species (e.g. Cane Toads (*Rhinella marinus*) introduced to Australia to control Cane Beetle; Burnett, 1997). However, the potential detrimental impact of biological control releases needs to be weighed against the ecological benefits conferred by these species in limiting pest abundance. For instance, in the United States over the past 100 years only 0.76% of deliberate releases of biological agents to control weeds have been deemed harmful to non-target species, of which nearly all were minor (McFadyen, 1998; Mueller & Hellmann, 2008). Protocols for screening the harmful effects of biocontrol agents could be a useful source of information when designing assisted colonisation projects.

### 3.4 Operational hurdles and strategies for minimizing risk when undertaking assisted colonisation

Both advocates and critics of assisted colonisation recognise the significant operational obstacles to implementing this type of conservation action. These obstacles include, but are not limited to:

- Financial costs of design, implementation and on-going monitoring;
- Coordination across political boundaries or jurisdictions;
- Failure of species to colonise, despite a well-designed program;
- Difficulty identifying suitable recipient sites. This may be due to habitat loss or a paucity of suitable land tenures, as well as the uncertainty inherent in identifying areas projected to contain suitable climatic habitat in coming decades.

Once the decision to undertake an assisted colonisation has been made, strategies are required to minimise the risk of failure and to protect the large financial
investment being made in this conservation action. Failures may occur due to biological factors such as inappropriate genetic mixing of populations leading to inbreeding depression, genetic swamping or hybridisation; introduction in recipient sites with unsuitable edaphic or climatic conditions in either the short or long term; or a lack of co-evolved mutualists, such as pollinators, leading to reproductive failure. In addition to these biological factors, a failure to implement long-term stewardship of and resources to assisted colonisations may result in a lack of monitoring, or the control of threats such as weed invasion or feral animals.

Strategies for minimising risks associated with assisted colonisation may include:

1) Completion of comprehensive pre-translocation assessments for each species targeted for assisted colonisation prior to the implementation of projects. This step would likely involve a desktop study to undertake activities such as: collating information on species biology and current range, bioclimatic modelling to locate areas of suitable habitat in coming decades as potential recipient sites, assessment of invasiveness risk, and seeking opinions from relevant experts in the taxon. For plant species, such a preliminary assessment should build on the protocols recommended in the Guidelines for the Translocation of Threatened Plants in Australia (Vallee et al., 2004) by including specific questions about the likelihood of climate change to adversely affect the species ability to survive in its current range.

2) The use of the principles of adaptive management to tailor the design, implementation and on-going monitoring of projects. In order to refine the practice, assisted colonisations should be designed and implemented on an experimental basis. This experimental approach allows for various methods to be tested and evaluated leading to an increase in the knowledge-base for future projects. Previous research into minimising specific risks should be used to inform the needs of individual assisted colonisation projects. For instance, various frameworks exist for assessing the risk of genetic mixing (e.g. determining the probability of outbreeding depression between two populations (see Frankham et al., 2011); evaluating the risk to native populations of adverse genetic change from revegetation (see Byrne et al.,
2011a); considering the risk of genetic pollution in eucalypts from exotic pollen dispersal (see Potts et al., 2003).

3) The use of existing frameworks (see Table 3) and guidelines to inform specific aspects of assisted colonisation. Various international agencies and governments recognise the need to prepare for the potential use of assisted colonisation as a conservation tool in coming decades. Table 4 provides examples of the types of strategies and guidelines being implemented globally.

Table 4. Examples of policies and guidelines with reference to assisted colonisation as a response to climate change for biological conservation purposes.

<table>
<thead>
<tr>
<th>Organisation</th>
<th>Publication</th>
<th>Reference</th>
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<tbody>
<tr>
<td>Ontario Forest Research Institute</td>
<td>Colombo, S., 2008, Ontario’s Forests and Forestry in a Changing Climate. Climate Change Research Report CCRR-12,</td>
<td>Page I: ‘Climate change will increasingly make species and local populations of tree species less well adapted to the climate where they occur’. ‘Increasingly, forest managers will consider planting non-local species and populations. Such potential adaptations, however, need to be carried out with consideration of potential negative consequences and if implemented should be well documented and monitored’.</td>
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<tr>
<td>(1) National Committee for Wild Flora and Fauna and (2) State Commission for Natural Heritage and Biodiversity, Spain</td>
<td>Comisión Estatal para el Patrimonio Natural y la Biodiversidad. 2013, unpublished report. Directrices técnicas para el desarrollo de programas de reintroducción y otras traslocaciones de conservación de especies silvestres en España. Ministerio de Agricultura, Alimentación y Medio Ambiente, Madrid, Spain Translated by <a href="http://translate.google.com">http://translate.google.com</a>:</td>
<td>Page 12: ‘Moreover, in general, the decision to initiate reintroduction programs or other translocation for conservation should be based on the situation in which he finds a taxon, recommending implementation:- When the species / population is susceptible to negative effects of human activities, including anthropogenic climate change, or stochastic events,’…… Page 15: ‘The expected effects of climate change must be taken into account when planning all types of translocations, so that the results of”</td>
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<td>Source</td>
<td>Document Title and Details</td>
<td>Quote or Key Point</td>
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<td>Technical guidelines for the development of programs and other re-introduction translocations conservation purposes wildlife in Spain. July 24, 2013</td>
<td>these actions may be viable in the long term. (Excerpts from the translated version of the Spanish document. No copy available in English. Translation provided by <a href="http://translate.google.com">http://translate.google.com</a> and Emilio Lacuna, Senior Officer Plant Conservation, in the Generalitat Valenciana section of Natural Resources Protection (pers. comm. 10 January, 2014).</td>
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<td>Council of Europe</td>
<td>Convention on the conservation of European wildlife and natural habitats, Standing Committee 32nd meeting Strasbourg, 27 - 30 November 2012</td>
<td>Recommendation No. 158 (2012), based on the IUCN Guidelines: Page 12 ‘To also consider ex situ measures, such as relocation, assisted migration and captive breeding, among others, that could contribute to maintaining the adaptive capacity and securing the survival of species at risk, taking into account the precautionary approach in order to avoid unintended ecological consequences including, for example, the spread of invasive alien species’</td>
</tr>
<tr>
<td>IUCN</td>
<td>IUCN/SSC, 2013, Guidelines for Reintroductions and Other Conservation Translocations. Version 1.0. Gland, Switzerland: IUCN Species Survival Commission, viii + 57 pp.</td>
<td>‘Whilst assisted colonisation is controversial it is expected to be increasingly used in biodiversity conservation’. ‘The climate at the destination site should be suitable for the foreseeable future’.</td>
</tr>
<tr>
<td>Royal Society for the Protection of Birds (UK) (RSPB)</td>
<td>Internal policy on translocations</td>
<td>Acknowledgement that ‘conservation introductions, including assisted colonisation in response to climate change, can have a valid role in conservation. It is recognised that conservation introductions pose far greater potential risks and uncertainties than reinforcements or reintroductions, and should therefore be only progressed when there is a high level of confidence over the organisms’ performance after release’ (pers. comm. Mary Davis,</td>
</tr>
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Committee on Climate Change (Independent, evidence-based advice to the UK Government)

Managing the land in a changing climate – Adaptation Sub-Committee progress report, 2013

Page 51: ‘Other adaptation actions may be needed to accommodate inevitable changes. These are likely to include: translocating some species, and adapting conservation objectives and site management regimes to reflect changing climatic conditions and shifting species distributions’

3.5 **Legislative, policy, social and ethical considerations when pursuing assisted colonisation**

Assisted colonisation as a conservation strategy has ethical, social, cultural, economic and political implications. Programs and projects require a clear set of goals and objectives and procedures to anticipate and minimise the potential for conflict and maximise positive synergies. The implementation of assisted colonisations has the potential to create tension amongst stakeholders with competing interests. For instance, adherence to the obligations to save species/ecosystems from extinction at one site (by way of translocation) but to also guarantee that the action will not be detrimental to ecosystems or land-uses at the recipient site may not be possible. However, if assisted colonisation is not implemented, the focal species/ecosystem may suffer extinction, thereby causing a neglect of duty (Schwartz et al., 2012).

Conservation strategy and operational actions, in all Australian jurisdictions, occur within general legislative and policy frameworks. These in turn represent a synthesis of prevailing social values and priorities, a prior history of conservation management practices, a scientifically informed legislative and policy development process, and an empirical determination of the necessary and acceptable scope of regulation and deployment of resources. Regulatory frameworks do not capture the full range of philosophical or ethical considerations that exist in society, nor do they provide resolution per se of potential differences over these, or of unresolved scientific questions, or of conflicts of perception and interest among stakeholders.
They do however provide a framework around which those issues can be identified and potentially resolved, and into which new subordinate policies and guidelines can be developed on an adaptive management basis.

However, if there is one consistent theme over recent decades in all law and policy relating to biodiversity conservation, it is a recognition that we face an accelerating rate at which species are being driven towards or into extinction, as part of a general decline in natural systems, and that conservation strategy should be directed to slowing and eventually reversing that process.

In some overarching legislative and policy documents relevant to NSW, the commitment to prevention or minimisation of species extinction is explicit. The primary example is the **NSW Threatened Species Conservation Act 1995**:

*Section 3 (Objects of Act):*

(a) to conserve biological diversity and promote ecologically sustainable development, and

(b) to prevent the extinction and promote the recovery of threatened species, populations and ecological communities, and

(c) to protect the critical habitat of those threatened species, populations and ecological communities that are endangered, and

(d) to eliminate or manage certain processes that threaten the survival or evolutionary development of threatened species, populations and ecological communities, and

(e) to ensure that the impact of any action affecting threatened species, populations and ecological communities is properly assessed, and

(f) to encourage the conservation of threatened species, populations and ecological communities by the adoption of measures involving co-operative management.

In other overarching legislation and policy the commitment to minimisation or prevention of species extinction is implicit within generalised conservation objectives, e.g.:

- Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (section 3, Objects of the Act);

- Australia’s Biodiversity Conservation Strategy 2010–2030 (e.g. ‘Our Vision’, ‘Principles underpinning the Strategy’, and ‘Call to action’ sections); http://www.environment.gov.au/node/14488

There is thus a clear legislative and policy mandate for conservation actions aimed at reducing extinction risk. Insofar as such governmental documents also embody a social ethical consensus, this overarching goal can also be assumed to have broad social acceptability, at least to the extent the actions proposed are compatible with other social objectives.

Generalised acceptability however does not mandate all potential actions nor ensure best practice. A large literature, too large to survey in this report, documents inadvertent adverse effects on biodiversity from inappropriate actions in natural resource management in general and sometimes directly in conservation programs. As a result, conservation agencies and responsible practitioners place heavy stress on nesting conservation actions within broadly agreed science-based and (preferably) critically evaluated programs, which are often government-led. Actions involving vertebrate animals, whether for research or direct conservation management, are now also usually subject to one or more specific institutional ethics approval processes. Agencies and most practitioners are properly cautious where highly interventionist techniques are proposed for the conservation of wild species. Translocation in general, and assisted colonisation in particular, are techniques of this sort.

There is an ultimate dilemma, to be assessed and resolved on a case by case basis, between acting to reduce extinction risk versus the risk of failure and/or inadvertently making the situation worse for the subject species or others. One of the main precepts of medical bioethics (“first, do no harm”) is an appropriate guideline. The decision for or against a strongly interventionary action is the core conservation question (along with associated issues of shortfalls in knowledge-base and technical capabilities). It needs to be addressed, to the extent possible, before or at least
distinct from the evaluation of other considerations not driven by the conservation goal (e.g. cost/benefit, competing socio-economic interests, conflicts with other policy), although any of the latter may in practice be sufficient to negate the purely conservation considerations in any one case.

While assisted colonisation projects remain at a relatively low level of frequency, tensions and conflicting goals may be, and usually are, resolvable on a case by case basis. However it will be better in the long run, and particularly if assisted colonisation becomes a more frequent technique, for the necessarily unique factors in each case to be situated within a policy and guidelines framework that applies to all conservation translocations (including climate-related assisted colonisation), and provides planners and managers with general guidance on consultation, stakeholder identification, proposal development, and socio-ethical case analysis. A requirement for a formal translocation proposal for each project, coupled with a pro-forma for such proposals that requires that scientific, socio-ethical and consultation factors all be addressed, would help to ensure consistency to the degree possible and anticipate, prevent or minimise conflicts.

The uncertainties and risks associated with assisted colonisations for legislators can also be lessened by the use of structured and transparent decision making processes. It will be imperative that policy makers coordinate at the national through to local government level to provide consistent legislation and policy. The fate of alpine species that are elevationally restricted in one state but not another is a prime example of where State co-operation will be required. Co-ordination at the international level may also be required for migrant species protected under treaties such as the Ramsar Convention on Wetlands. The input of researchers and groups with practical conservation experience such as Birdlife Australia, Australian Network for Plant Conservation, Greening Australia, and other various ecological experts and conservation stakeholders should also be incorporated.

At the societal level, Schwartz et al. (2012) argue that legislators must be vigilant that any new laws encompassing assisted colonisation do not weaken society’s commitment to biodiversity protection. Public consultation and education on the merits and possible downsides of assisted colonisation will be critical in maintaining legislative strength and agency leadership, and acquiring acceptance of new species in new locations. The acceptance and involvement of local groups in
the colonisation of new species as care takers and protectors of the focal species at the new site may also be helpful in translocation success.

The financial cost of assisted colonisations is likely to require additional government funding and the amount will be context specific, as are all types of translocations. As a guide, the cost of flora translocations in Australia varies enormously between projects: ~ $5,000 for basic projects with sub-optimal monitoring; $30,000 for large projects with >1000 individuals with limited monitoring; at least $100,000 over 15 years; $10,000 to $1,000,000 (Hancock et al., 2014). In comparison, the reported cost of a Chinese translocation project that moved orchid species beyond its current range was 1.5 million USD in 2012 (Liu et al., 2012). As with all innovative conservation techniques, both the chances of success and greater efficiencies are likely to increase ‘with practice’. Encouragement of experimental actions to test procedures and identify problems will pay dividends in the future.

3.6 Is assisted colonisation the appropriate solution? Investigating lower risk strategies for climate change adaptation

Translocations, whether specifically designed for climate change adaptation (assisted colonisation) or for managing more proximal threats to species, are expensive, difficult to implement, and require a long-term commitment of resources. Even after considerable effort, success may not be guaranteed. Therefore, all alternative management actions that may help to ensure the survival of a species or community should be considered prior to making the decision to create new populations. Whilst predictions of the adverse effects of climate change on species have rapidly accumulated over the last three decades, evidence for the efficacy of assisted colonisation as a tool for adaptation has been slower to emerge. The lack of empirical information on which to base assisted colonisation guidelines increases the uncertainty surrounding the planning, implementation and monitoring of this type of conservation action.

Implementing lower-risk strategies that can be undertaken now may help to increase the capacity for at-risk species to adapt to a changing climate, and ultimately may negate the need for assisted colonisation in some circumstances. Actions for facilitating climate change adaptation in species may include:
• Increasing and/or restoring landscape connectivity; (Heller & Zavaleta, 2009; Shoo et al., 2013);
• Identifying and protecting potential micro-climate refugia (Corlett & Westcott, 2013; Shoo et al., 2013);
• Reducing multiple stressors to increase resilience (Lindenmayer et al., 2010);
• Expanding ex-situ conservation (Vitt et al., 2010);
• Revising protocols on provenance to take account of climate change (Broadhurst, 2008; Breed et al., 2013; Sgro et al., 2011; Whalley et al., 2013);
• Updating conservation translocation guidelines to include translocations outside of their indigenous range due to threats from climate change (assisted colonisation) (Byrne et al., 2011b).

It is generally recognised that assisted colonisation will not be effective as a climate change adaptation measure if implemented in isolation (Lindenmayer et al., 2010). In-situ measures to mitigate other threatening processes that are likely to be exacerbated by climate change should be tackled first, with assisted colonisation only recommended if these strategies fail or are deemed inappropriate.

SECTION 4 - When does assisted colonisation have the most likelihood of success?

4.1 Evaluating the success of past translocations

Evaluating the success of past translocation projects provides insight into how to improve the likelihood of an assisted colonisation being successful. Unfortunately, there are relatively few well-documented examples of successful translocation projects where self-sustaining populations have been formed. The value, efficiency & effectiveness of translocation in general (and hence also assisted colonisation) remains largely unknown given that the success of most past translocation projects is uncertain or unevaluated & there has been a general lack of monitoring, evaluation & comparative studies. Improved monitoring and evaluation of translocation activities will increase the effectiveness of decision making regarding the role of translocation and specifically assisted colonisation. Appendices 1-4 of this report catalogue known flora translocations undertaken in Australia, published reviews of flora and
fauna translocations and a series of case-studies on experimental assisted colonisations.

**4.2 Introducing six key themes for successful assisted colonisation**

The IUCN states that ‘the pivotal criteria for justifying any conservation translocation will be situation-and species-specific’ (IUCN/SSC, 2013). However, six key and inter-related themes repeatedly appear in the literature with general suggestions or recommendations for successful assisted colonisations. The themes: site selection and preparation, predictive tools, genetic diversity, lessons from invasive species research, species information and adaptive management are discussed below.

**4.3 Recipient site selection and preparation**

The characteristics of the sites that receive assisted colonised species (recipient sites) will have a major influence on the success or otherwise of assisted colonisations. The purpose of assisted colonisation is to move species to areas with suitable future climate and hence reduce the extinction threat present at the donor site. Species distribution models are helpful in selecting suitable sites (see Predictive Models section below), provided long term reliable climate projections are available for the site. In general, translocations and ecological restorations are more likely to fail where there is a large difference in the biogeographic regions between the source and the recipient populations (Dalrymple & Broom, 2010; Thomas, 2011). Potential recipient sites for plants, for example, should be representative of predicted future climate and edaphically match the source site.

The introduction of organisms into sites free from predators, pests and / or pathogens may increase the chance of translocation success. In a review of Australian vertebrate translocations, predation was cited as the key reason for the project failure for mammals and birds and a significant issue for reptiles and amphibians (Short, 2009). Furthermore, sites free of key threatening processes yielded greater translocation success than sites that were less secure. For plant translocations, in some cases, the exclusion of grazing herbivores is desirable (Vallee et al., 2004). However, the financial and potential ecological costs of fencing need to be considered and managed in any conservation translocation (Hayward & Kerley, 2009).
Site selection should also consider the availability of co-dependent organisms, and the adequacy of site size and tenure. The decision where to introduce populations to ensure continued stewardship and protection will also have a bearing on success (Bennett et al., 2013). Preferably, sites will be located inside the protected area network, such as national parks and reserves. It should be noted, however, that the most suitable habitat may lie outside the protected area network. For instance, land owned by non-government organizations such as Bush Heritage, Birdlife Australia, or Arid Recovery as well private land-holders may become increasingly important for recipient site selection for assisted colonisation projects. Potential sites may include those where there are existing gaps in biological function that can be filled by candidate species or fall within migration corridors in order to aid meta-population establishment (Harris et al., 2013).

A range of alternative strategies for locating suitable recipient sites and/or expanding existing populations into suitable habitats have been identified, including:

- the rehabilitation of unproductive or marginal agricultural land and degraded mining leases with ‘climate ready’ plants (Lindenmayer et al., 2010);
- active promotion of the benefits of pollination services to agricultural sectors to promote the introduction of bee species (Lunt et al., 2013);
- use of golf courses, public spaces and private gardens as stepping stones or corridors to, or as recipient sites (Harris et al., 2013).

However, the use of recipient sites on non-government land will also need to take account of socioeconomic factors important to the stakeholders to provide secure tenure.

### 4.4 Predictive Models

The use of predictive tools to assist in facilitating successful assisted colonisations is a vital part of planning and implementation. Here we describe three predictive tools: climate models, species distribution models and population viability analysis.

**Climate models** (e.g. Global Circulation Models or Global Climate Models (GCMs)) are used to predict the future climate of both source and recipient sites. The current generation of climate models reproduce observed continental-scale surface temperature patterns and trends over many decades with a high-level of confidence
Similarly, model simulation of extreme weather events and other climate phenomena such as ENSO has improved substantially, increasing the level of confidence assigned to predictions of future climate conditions under higher greenhouse gas concentrations (IPCC, 2013). Whilst there is high confidence in models of climate at the global scale, predictions from regional climate models are essential for finer-scale identification of at-risk species or communities and recipient sites for assisted colonisation projects (e.g. NARCLIM – NSW/ACT Regional Climate Modelling project producing an ensemble of regional climate projections for south-east Australia.

**Species distribution models** (SDMs), are used to predict the distribution of suitable habitats for species, under both current and future climate scenarios. Also termed ecological niche, environmental niche, habitat suitability and bioclimatic envelope models (Pearson, 2007), SDMs employ either a correlative or mechanistic approach. Correlative models combine species occurrence records with gridded environmental data (e.g. climate, soils) to project the location of suitable habitat for individual species, or suites of species (Guisan & Thuiller, 2005). By contrast, mechanistic SDMs project species’ ranges by incorporating information on physiologically-limiting mechanisms which govern tolerance to conditions with environmental data (Pearson, 2007). Correlative models are more commonly employed due to the availability of occurrence records, climate data and free-software, whereas detailed biological data needed for mechanistic models is often unavailable (Pearson, 2007). There is some evidence that mechanistic models may be more informative for selecting suitable recipient sites than are correlative approaches. For instance, Mitchell *et al.* (2013) showed that mechanistic models performed better than correlative models at identifying recipient sites for range-restricted species, such as the Critically Endangered Western Swamp Tortoise (*Pseudemydura umbrina*) in Western Australia.

It is important to note that the utility of models for identifying candidate species or recipient sites is substantially improved when a combination of tools and parameters are used (Pearson & Dawson, 2003; Angert *et al.*, 2011). For instance, extending the type of parameters used to calibrate SDMs beyond climate variables to include such factors as topography, soil conditions, dispersal distances,
evolutionary change, biological traits and biological interactions will fine-tune predictions. Where dispersal distances are combined with SDM output, estimates can be made as to whether the rate of dispersal will match the rate of climate change (Engler et al., 2009). A recent example of this approach used genetic information to project the future distribution of a widely distributed Australian gecko, Gehyra variegata (Duckett et al., 2013). Genetic analysis showed that dispersal rates for the gecko varied across the landscape (from a mean of 1.69 - 6.26 km year \(^{-1}\)) and that the current range of the gecko is predicted to decline by 0.87 (±0.25) million km\(^{-2}\) by 2030 and further by 0.62 (±0.24) million km\(^{-2}\) by 2070 (Duckett et al., 2013). The authors inferred from the results that 17 – 41% of the species’ current distribution will not keep pace with the predicted velocity of climate change. The study also identified populations that may become isolated and suffer the consequences of environmental stochastic events and population viability problems (Duckett et al., 2013).

Other examples of more complex modelling approaches include:

(1) Assessing the vulnerability to climate change of 16,857 species of birds, amphibians and corals species using biological traits alongside of habitat suitability projections under climate change (Foden et al., 2013).
(2) Dispersal kernel models that simulate migration rates, allowing for rare, long distance dispersal events in order to predict the probability of colonisation beyond the current range over long time periods (Iverson et al., 2004).

However, predictive models have limitations which include:

(1) Species distributions used as baseline observations may not be in equilibrium with the current climate;
(2) Realized niches used in models may not represent absolute ranges, effecting predictions for future distributions;
(3) Impact of dispersal barriers are not always identified;
(4) Modelling done at the global scale does not always allow for widespread application at the regional scale or for specific species.

These limitations need to be taken into account when interpreting the outputs.

**Population Viability Analyses** (PVAs) model the effects of different life history, environmental and threat factors (deterministic and stochastic) on the population size
and extinction risk of populations or species (Frankham et al., 2010). Historically, PVAs have been used to predict reproductive and survival parameters for threatened species. PVAs would arguably be of benefit in assessing potential assisted colonisation projects, for both the source and recipient populations. Information on reproduction, mortality rates, population size, carrying capacity, frequency and effects of threats and genetic information (susceptibility to inbreeding depression and gene flow) are required for analysis (Frankham et al., 2010). In reality, there are low prospects of being able to meet PVA data requirements for more than a small set of species. Hence, the initial choice of species for investment in PVA needs to screen for the likely generalisability of some/all outcomes, and all actual outcomes need to be carefully evaluated.

4.5 Genetic diversity
The presence of genetic diversity is an important factor in the success of any conservation translocation. Genetic considerations in successful translocations include the avoidance of inbreeding and outbreeding depression, a loss of genetic diversity due to founder effects or genetic drift and the ability to maintain appropriate breeding systems (Menges, 2008). Genetic diversity can be divided into two categories: neutral and adaptive. Neutral genetic diversity reflects population dynamics and evolutionary forces such as genetic drift, mutation and migration and is not under natural selection (Sgro et al., 2011). Adaptive genetic diversity is under natural selection and gives rise to the ability to adapt to new environments (Sgro et al., 2011). It is therefore important that translocations that introduce populations outside of its known historical range (assisted colonisations) have adequate levels of both types of genetic diversity to enable adaptive capacity.

Estimates of both types of genetic diversity may be difficult to attain. Commonly-proposed strategies to retain maximum genetic diversity are to source the transplant genetic material from multiple sources (Frankham et al., 2010) and from provenances that match likely future climate at the recipient site to increase adaptability (Sgro et al., 2011). The minimum number of reproductive individuals required to keep pace with environmental change by maintaining adaptive potential is estimated to be in the low thousands (Willi et al., 2006; Frankham et al., 2014). A review of plant reintroductions found that mixing material from diverse populations
resulted in higher survival rates of restored individuals than where sourced from a single population (Godefroid et al., 2011). Information that attributes translocation success or failure because of levels of genetic diversity is, however, not always available.

Consideration should also be given to potential detrimental effects on the genetic diversity of source populations where extensive harvesting of seed material is carried out to help establish new populations. Population viability analysis that model genetic factors will be helpful in determining the degree of detrimental effect on the source population and the numbers needed for new populations to persist.

In the context of assisted colonisation, it is important to identify the financial and ecological trade-offs between moving unthreatened extant populations, currently with substantial adaptive capacity, and prioritizing threatened species with reduced genetic diversity, but with a high need for translocation due to other stressors. A review of Australian vertebrate translocations found that translocation success rates were higher for non-threatened bird and mammal species, than for threatened species (Short, 2009). If the assumption is made that threatened species have low levels of genetic diversity (due to small population numbers), the results of this study implies that it is better to translocated before genetic diversity is further reduced. Frameworks of genetic considerations for translocations under climate change are provided by Weeks et al., (2011).

**4.6 Species information**

There is a long list of biological information that is lacking for many species to enable confident predictions of successful assisted colonisations. All translocations are undoubtedly more successful where there is knowledge of:

- Climatic breadth and niche;
- Expected successional trajectories;
- Disturbance regimes required for reproduction;
- Breeding system;
- Genetic diversity;
- Ideal growing conditions (e.g. moisture, light, nutrients) for plants;
- Mutualisms;
• Critical life stage requirements (e.g. early progeny needs, reproductive triggers);
• Habitat preferences;
• Diet breadth (for animals);
• Physiological constraints on growth and survival.

However, the time taken to acquire knowledge before acting needs to be balanced against the potential consequences of failing to act soon enough. The risk of acting without knowledge also needs to be evaluated.

Species and situations that present low-risk opportunities for assisted colonisation should be prioritised. For instance, the IUCN/SSC (2013) have identified a situation where a species has gone locally extinct, creating a gap in ecological function, as a potentially low-risk translocation project. In this situation, a functionally similar species that is threatened by climate change elsewhere could perform a replacement ecosystem function. Species known to have high levels of phenotypic plasticity may be better adapted to withstand extreme events and to therefore successfully colonise recipient sites (Bernazzani et al., 2012) and make ideal candidates to serve as ecological replacements (on the basis that high levels of plasticity may mean that they are less likely to be threatened in-situ).

4.7 Learning from invasive species research

To mitigate against the potential for species which are subject to assisted colonisation becoming invasive, rigorous monitoring of recipient sites and dispersal corridors needs to be conducted. However, conflicts may arise when trying to define ‘success’ in an assisted colonisation project where large numbers of new individuals are being recruited, leading to a reduction in the abundance and diversity of extant vegetation at the recipient sites. A clear, working definition of optimal success should therefore be outlined prior to the commencement of an assisted colonisation to avoid ambiguity between appropriate population growth and invasive behaviour (Vallee et al., 2004). Practitioners of assisted colonisations need to balance both how to prevent invasive behaviour in translocated species, and how to increase the potential for success. Research shows that the following factors are crucial in successful
invasions and should be taken into account when designing assisted colonisation projects:

(1) **Propagule pressure and introduction effort.** The number of times and the number of individuals that have been introduced to a new range is a critical predictor of the ability for species to naturalise and become invasive (Lockwood *et al*., 2005; Simberloff, 2009).

(2) **Minimum residence time.** The amount of time since the species was introduced to the new range. Invasions typically progress through a lag phase for several years immediately following introduction where population numbers build as species adapt to novel conditions. Once a critical threshold of individuals has been reached, population growth is exponential and introduced species can spread rapidly (Pyšek *et al*., 2005). Total eradication is only considered feasible during the lag-phase, when population numbers are low and locally contained.

(3) **Climate matching.** Species are more likely to become invasive if the recipient site where they are introduced has similar abiotic conditions to their native range (Gordon *et al*., 2008; Bomford *et al*., 2009). Whilst species do have the capacity to shift their niche upon introduction (Gallagher *et al*., 2010), similarity in climate and abiotic conditions between the native and exotic range is repeatedly identified as crucial in invasions.

### 4.8 Adaptive management

Due to the nascent nature of assisted colonisation, it follows that management will need to be structured and adaptive. Actions should be based on the best available knowledge and any uncertainties should be acknowledged (Chauvenet *et al*., 2013). Theoretical guidelines, risk management tools and decision frameworks have previously been developed to assist all aspects of decision making surrounding assisted colonisation (Table 3). The use of more than one framework may be necessary and should incorporate risk management tools that are not specifically designed for assisted colonisations but are nonetheless applicable. A comprehensive synthesis of some of these tools appears in Chauvenet *et al*. (2013)
which also provide a comprehensive list of pre-translocation questions to ask and details of where to find the answers.

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