

Conserving Australian bird populations in the face of climate change

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WILDLIFE MANAGEMENT IN AN UNCERTAIN FUTURE

In the best of times, wildlife management is an uncertain science (Sinclair *et al.* 2006). A climate-change world will not be the best of times, with a large increase in stress on many wildlife species and ecosystems along with a major increase in the uncertainty of management outcomes. Dealing with these challenges will at once require a major re-think of attitudes and approaches and, in seeming paradox, an intensification of existing management practices. As explained in the next section, the paradox arises mainly because the greatest threat to wildlife is likely to be the interaction between climate change and other stressors, and because dealing with those other stressors may often be the most, or even only, practical way of reducing stress from climate change.

In the face of climate change, some values that conservationists and wildlife managers have long held dear may become a hindrance rather than a guide to good management. The pre-European state of Australian ecosystems, for long the aspirational benchmark for conservation, is likely to decline in relevance with time and eventually become obsolete (Hannah *et al.* 2002). National parks and other reserves set up to conserve the *status quo* may end up conserving something quite different, undermining their current philosophical foundation and demanding a fresh perspective (Scott and Lemieux 2005; Monzón *et al.* 2011). In regeneration projects, local provenance may no longer be best for the maintenance of biodiversity (Sgrò *et al.* 2011). To save species we may need to move them to new locations, in the process undermining (or at least questioning or redefining) concepts of 'native' and 'invasive' (Webber and Scott 2012). At the very least, we will need to collectively broaden our spatial and temporal perspectives about conservation benchmarks. More fundamentally, we may need to 'move away from concepts of natural' and embrace ecosystem processes as benchmarks rather than patterns from the past (Heller and Zavaleta 2009).

One management consideration is worth dealing with up-front. It has been argued that limiting the emission of greenhouse gases is an essential element of any portfolio of biodiversity management actions (e.g. Hansen *et al.* 2010; Lindenmayer *et al.* 2010). Indeed, it can and has been argued that the ethics of some management actions (assisted colonisation) depend on efforts to mitigate climate change (Albrecht *et al.* 2013). We agree with these arguments, but note that global or national actions to curb emissions clearly do not lie in the portfolio of wildlife managers. Wildlife managers, however, can provide advice to policy-makers on the consequences of current policies for biodiversity, including issues such as which

greenhouse-gas capture schemes are favourable to biodiversity (van Oosterzee *et al.* 2010; Phelps *et al.* 2012).

Regardless, the curbing of emissions is beyond the scope of this chapter. Here, we focus on practical (on-ground) strategies and actions that wildlife and land managers can take to minimise the loss of bird species at regional, national and global levels, and the concepts underlying these. We first review general principles including the impact of interactions between threatening processes, dealing with uncertainty in setting management goals, and key decisions in the management of species. We then review information about actions that managers may take relating to habitat management, intensive management of species in the wild, last resort preservation of species, and the research needed to enhance management. Finally, we consider the costs of management and some issues with the decision-making process.

GENERAL PRINCIPLES

Synergies among threatening processes

Many of the taxa identified in this study as potentially threatened by climate change face other threats or stressors as well. This is to be expected as traits that describe or act as proxies for exposure and/or sensitivity often also contribute to or measure threatened status. For example, taxa with small ranges are often both rare and occupy small climate niches. Small population size describes limited genetic variability and thus limited capacity to evolve with climate change; small population size is also used directly in the assessment of general threat status (IUCN 2001). Species with slow life histories are often vulnerable to a range of threatening processes of which climate change is but one (Lee and Jetz 2011).

How these threat processes interact is critical for the future of Australia's birds – and for the management strategies that can be employed to combat adverse outcomes from climate change. If there is no interaction among threat factors, then each can be treated in isolation. However, if the effect is synergistic (interactive), then the combination of threats is often worse than the sum of each (Brook *et al.* 2008) – a perilous and uncertain situation (Sala *et al.* 2000). However, interacting threats can create advantages for management, because the mitigation of other threat factors may reduce the impact of climate change even though dealing directly with the threat posed by climate change may prove intractable (Noss 2001; Driscoll *et al.* 2012). Indeed, CCWAPWG (2009) argued that dealing with 'existing stressors is one of the most valuable and least risky strategies available for climate change adaptation, in part because of the large existing body of knowledge about their impacts and solutions'. It is for this reason that some of the management actions discussed in this chapter are traditional (e.g. *expand the protected area network* and *maintain and improve habitat quality*) and don't obviously address the impacts of climate change directly.

The extent of synergies between climate change impacts and other threatening processes is largely unknown. Many ecologists have argued that such synergies are or will be widespread and *the* major source of threat under climate change (e.g. Root *et al.* 2003; Stork *et al.* 2009; Brodie *et al.* 2012; Şekercioğlu *et al.* 2012), though Darling and Cote (2008) presented

contrary evidence. Some synergies are obvious and have received strong theoretical support: for forest specialists with limited dispersal capacity, clearing vegetation not only fragments and reduces the viability of populations but also reduces their ability to track a shifting climate, thus synergistically increasing extinction risk (Travis 2003; Opdam and Wascher 2004). Fundamental ecosystem shifts are more likely in synergy with human-induced stress (Folke *et al.* 2004). Threats are demonstrably linked among tropical forest species (Laurance and Useche 2009). Laboratory experiments with populations of microorganisms demonstrate synergistic effects between increased temperature and other stressors (harvest, habitat fragmentation) (Mora *et al.* 2007). Mantyka-pringle *et al.* (2012) demonstrated the generality that habitat loss has already interacted with current temperature and secondarily with rainfall change, the synergy being greatest where temperatures are high and least where rainfall has increased.

A synergy between invasive species and climate change has been postulated on the basis that the traits that favour invasiveness (e.g. behavioural adaptability, habitat generalism [Sol *et al.* 2012]) should promote the success of species in the face of climate change (Weltzin *et al.* 2003; Pyke *et al.* 2008). There is some evidence of emergence of a general trend as postulated (Dukes and Mooney 1999; Morrison *et al.* 2005; Chown *et al.* 2007; Ward and Masters 2007; Winder *et al.* 2011), but not much evidence ‘yet’ that invasive vertebrates have become more prevalent under climate change (Brook 2008).

There are several putative and potential specific examples of synergies between climate change and other stressors on bird populations in Australia and elsewhere. Mac Nally *et al.* (2009) documented ‘collapse’ of the avifauna of box-ironbark woodlands in Victoria with prolonged drought where vegetation loss has been concentrated in the more fertile and moisture-retentive parts of the landscape. Though current hunting pressure on Magpie Geese (*Anseranas semipalmata*) is well below threat level (Delaney *et al.* 2009), modelling suggests that the population could collapse with continued hunting and the loss to rising sea levels of as little as 5% of coastal wetlands (Traill *et al.* 2009, 2010). If coastal habitat could migrate upslope with rising sea levels, breeding area for the threatened Piping Plover (*Charadrius melodus*) in New York state (USA) should increase, but instead a decrease of 5–12% is indicated because coastal development constrains the migration (Seavey *et al.* 2011). Avian malaria, a listed threatening process for several birds (Brito *et al.* 2012), is demonstrably increasing most in areas exposed to greatest recent temperature change (Garamszegi 2011). A combination of past land-use and projected climate change are likely to drive several Hawaiian honeycreepers to extinction (Benning *et al.* 2002).

We concur with Brook *et al.* (2008), Stork *et al.* (2009), Brodie *et al.* (2012) and many others in arguing that synergies among threats will play a major role, and perhaps the most important role, in precipitating adverse outcomes for birds and other biodiversity in the face of climate change. It is imperative that managers be cognisant of synergies among stressors and their implications for management – and that the impact of climate change may be obscured by seemingly more immediate threats.

Adaptation (management) strategies

Management actions to deal with the consequences of climate change are often collectively

referred to as ‘adaptation strategies’ (Smithers and Smit 1997). These may relate to ecosystem and wildlife management (Hulme 2005; Mawdsley *et al.* 2009; Cross *et al.* 2012), or more generally to human ecology such as the impact of sea-level rise on coastal communities and the suitability of future climates for agriculture (e.g. King 2004). This reflects a key reality that the goals of management need to be re-set to incorporate climate change as the absence of climate change management will very likely ultimately lead to failure (Lawler *et al.* 2010).

Use of the term ‘adaptation’ in the context of wildlife management can generate confusion over whether it is managers or wildlife that is adapting, or need to do so (Mawdsley *et al.* 2009; Lawler *et al.* 2010). In many cases, wildlife will adapt phenotypically and/or genetically. Phenotypic adaptation could include, for example, altering the timing of breeding or migration. Genetic change can occur through selection on either existing variability or phenotypes with new mutations (Visser 2008; Donnelly *et al.* 2012). It has been argued that a key goal of wildlife management under climate change should be to promote adaptation by species, for example in Australian birds (Chambers *et al.* 2005), and the ability of species to adapt is a key unknown for management by assisted colonisation (Moritz 1999). Further, the phrase *adaptive management* has special meaning in resource management as a form of learning by doing – treating management as an experiment (Walters 1986) – with particular application in dealing with the uncertain impacts of climate change (Conroy *et al.* 2011). The term ‘adaptation’ clearly has multiple meanings even when application is restricted to the context of climate change and wildlife. Consistent with Lawler *et al.* (2010), in this chapter we refer to the actions that managers may undertake simply as *management actions*, a suite of such actions designed to achieve a particular aim as a *management strategy*, and the process of monitoring outcomes in order to re-evaluate actions and strategies as *adaptive management*.

Table 1. Conceptual approaches to the management of wildlife in the face of climate change (modified from Millar *et al.* 2007 and CCWAPWG 2009).

Management goals	Explanation and example
Promote <i>resistance</i>	Actively manage to maintain the status quo, or to a previous more desirable state, and forestall impacts of climate change Examples: - control competitive invasive species (could involve controlling climate change refugees) - revegetate to historical habitat states
Promote <i>resilience</i>	Improve the capacity of populations to recover from disturbance Examples: - ensure reserves contain viable populations able to recover after drought, wildfire or cyclone - create or maintain corridors between similar environments
Facilitate <i>response</i>	Enable transition to new conditions Examples: - create or maintain corridors between contrasting environments - assist migration so that species can track shifting climate envelopes - enhance climate-adaptedness of population gene pool with stock from other climate zones

- | |
|---|
| <ul style="list-style-type: none">- create or modify habitat to suit changed climatic conditions- allow fire regimes to trigger vegetation change appropriate to the new climate |
|---|

For wildlife managers, developing strategies that incorporate climate change presents multiple challenges. Ecological uncertainty arises for at least four levels which compound each other (Table 1 in the Introduction to this book). Lawler *et al.* (2010, Fig. 2) provide a useful two-dimensional perspective on uncertainty. Possible management responses to ecological uncertainty include focussing on actions that are most robust to the state of our knowledge and thus inherently less risky (Groves *et al.* 2012), and on actions that do not require detailed local projections of climate change and its impacts (Cross *et al.* 2012). Cross *et al.* (2012) offer a flow-chart for making management decisions. Where detailed climate change projections are available, these may usefully be represented in terms of uncertainty and incorporated into simple heuristic models for management purposes (e.g. Lawler *et al.* 2010). More quantitative approaches to dealing with uncertainty of costs and benefits of management actions are also available (e.g. Conroy *et al.* 2011, 2012).

The principal strategy for coping with uncertainty is through a risk management framework. Under such a framework actions should generally be commensurate with the risk to the taxon (or ecosystem). For example, assisted colonisation is generally inherently more risky than improving the quality of existing habitat but may become necessary in more extreme cases of climate change stress (Morecroft *et al.* 2012). Poiani *et al.* (2011) provide a review of 'adaptation' strategies across a wide range of biodiversity conservation projects, and Hansen *et al.* (2010) a more detailed analysis of three such projects. It should be unnecessary to say that *adaptive management* is a key to dealing with uncertainty (Walters 1986; Hulme 2005; Conroy *et al.* 2011). In practice, however, management is rarely genuinely treated as an experiment (Walters 2007) and monitoring is inadequate or used merely to measure outcomes and justify funding.

The time frames for management may pose a particular conundrum. Goals for species recovery and related biodiversity projects are often stated in terms of aims or achievements by a particular date, but with increased uncertainty in a time of change these may become less relevant. Where threat is related simply and quantitatively to climate change, as in the case of rising sea-levels, goals may be expressed in terms of a quantum of change rather than a moment in time. Alternative approaches that offer greater flexibility could involve expressing goals in terms of targets for *resilience*, *resistance* and *response* (next paragraph).

In conceptual terms, the goals of conservation management under climate change may be 3-fold: to avoid the impacts of climate change (*resistance*), to promote resilience to those impacts, or to facilitate response to novel situations (Table 1). Many management strategies are now pitched in terms of these three concepts (e.g. Millar *et al.* 2007; Galatowitsch *et al.* 2009; Prober *et al.* 2012). *Resistance* is an extension of current management practice to include climate change as simply another threatening process. Management for *resilience* is an extension of current management practices in ensuring the viability of populations but including recognition that climate change may generate specific new challenges that increase the population size/connectedness/area of habitat required for viability (Morecroft *et al.* 2012). Managing for *resistance* and *resilience* are lower-risk strategies but may fail with more drastic

change in the longer term. Only the *response* goal specifically acknowledges that environments and their populations will change in some cases quite fundamentally, necessitating novel and innovative management responses such as the creation of novel habitats and acceptance that populations may need to colonise new areas and unavoidably disappear from former habitats.

Table 2. Management options for the conservation of birds in a climate-change world (modified extensively from Mawdsley *et al.* 2009; see also Shoo *et al.* 2013).

Type of management	Practical options
<i>do nothing</i>	–
<i>maintain and enhance habitat</i>	<ul style="list-style-type: none"> * expand the protected area network * maintain and improve habitat quality * identify, protect and expand refugia * maintain and extend landscape connectivity * create new habitats
<i>facilitate the response of wild populations (intensive species management)</i>	<ul style="list-style-type: none"> * assist colonisation by translocation * enhance the genetics of subspecies * manage other threatening processes, e.g. by predator control, habitat manipulation, captive breeding
<i>preserve populations – the last resort</i>	<ul style="list-style-type: none"> * save species in captivity * store germplasm
<i>understand what is happening and prepare for what may happen</i>	<ul style="list-style-type: none"> * monitor bird populations - general surveys (e.g. Atlas) - targeted species-specific monitoring * monitor habitats and threatening processes * investigate the ecology of species and communities * model habitat and climate envelopes in more detail * model management options

Should we be managing landscapes, ecosystems or species?

Although the assessments provided in this study are at the level of species and subspecies, biodiversity management commonly occurs at the level of landscapes, ecosystems or protected areas. Manipulation of habitat is rarely undertaken with the interests of just a single species in mind, yet habitat issues remain at the core of both biodiversity and species management (Table 2).

Response to climate change may at times require an emphasis on ecosystem processes and function (Cumming and Child 2009; Heller and Zavaleta 2009) in which species may be tools rather than the endpoint of conservation effort. For example, a key process for the maintenance and enhancement of ecosystem *resilience* is to ensure functional redundancy in which many species occupy similar and potentially overlapping niches. At a broad level, biodiversity richness alone has important links to ecosystem processes (Cardinale *et al.* 2012; Hooper *et al.* 2012). A range of insectivorous birds may help to prevent insect outbreaks (Loyn *et al.* 1983), and a forest is more likely to be robust to climate-induced change in insect populations

if it contains many rather than just a few insectivores. Under climate change it is conceivable that assisted colonisation of species might prove necessary to maintain ecosystem function (Dunwiddie *et al.* 2009), a scenario which can also provide an opportunity for species conservation (Lunt *et al.* 2013). The pay-off for species management is that diverse ecosystems by definition support more species, and resilient ecosystems provide a greater buffer for populations (Karp *et al.* 2011).

Action hierarchies

A key decision to be made in the management of species is whether it is possible and appropriate to manage them where they are, including where they might move to without assistance (*in situ*), whether it is necessary or desirable to manage them somewhere in the wild where they do not occur (assisted colonisation) or, should extinction be deemed inevitable, whether *ex situ* management such as captive breeding is necessary, desirable and possible. For any species, the answers to these questions may change over time, and a species management strategy may involve positive answers to more than one question simultaneously. Species may be managed within part of their existing range and some individuals translocated outside their current range. Assisted colonisation may occur within the former or current range. Species may establish in new areas unassisted, raising a different set of management questions. Captive breeding may be employed to assist *in situ* management or migration, in addition to as a last resort option. For this reason, and whilst acknowledging that our alternate classification is also far from categorical, in the following sections we discuss management options following Table 2, in which we emphasise the distinction between habitat management which will often apply to many species and be undertaken by a wide range of land managers, intensive species-specific management to maintain populations in the wild which is often undertaken by specialist groups and personnel, and last-resort measures to preserve species or their genetic code when conservation in the wild appears impossible. We also discuss issues related to research to facilitate management, including monitoring.

There is yet another management option – to do nothing. For many species, this may be a realistic and appropriate option, but it can only be appropriate under three conditions:

- 1 after (at least generic) consideration of the consequences;
- 2 with on-going monitoring in at least a generic manner, e.g. Bird Atlas (Barrett *et al.* 2003); and
- 3 the flexibility to recognise and respond to changed circumstances in a timely manner.

Given uncertainty about the response of species to climate change, and in particular the possibility of synergistic effects, ecological surprises are inevitable (Doak *et al.* 2008), and if we are to conserve Australia's birds we must be able to both detect and respond to both incremental and abrupt change in their fortunes.

ACTIONS: MAINTAIN AND ENHANCE HABITAT

In this section, we discuss management actions that have general application to the conservation of birds and other biodiversity in the face of climate change. These actions will mostly be applied to ecosystems and regions for the benefit of many species and only sometimes as part of a species-specific response to the threats posed by climate change.

Expand the protected area network

The protected area network embraces not only national parks and other crown conservation reserves but also the private conservation estate and landscape features such as corridors and scattered trees within areas where conservation is not the primary land use. The protected area network also includes Marine Protected Areas, a priority for seabird conservation (Croxall *et al.* 2012) with demonstrable or likely positive effects for biodiversity in the face of climate change (Micheli *et al.* 2012; see Ling and Johnson 2012 for a Tasmanian example). A key management strategy will be to retain habitat not just where species are currently but also where they are predicted to move to. In the face of climate change, protected areas will play a key role in the maintenance of biodiversity in general (Hannah *et al.* 2007; Monzón *et al.* 2011) and bird populations in particular (Hole *et al.* 2009; Şekercioğlu *et al.* 2012), notwithstanding the need for a major shift in thinking about their goals. The interaction between climate change and fragmentation means that a greater proportion of fragmented landscapes must be set aside for conservation to maintain current levels of biodiversity (Travis 2003; Opdam and Wascher 2004; Van Teeffelen *et al.* 2012). Geographic shifts in climate may require the establishment of protected areas in regions where none currently exist (Hole *et al.* 2011). It is even conceivable, and perhaps even likely, that novel habitats will need to be created in areas currently committed to other purposes. Methods for identifying optimal placement of protected areas under climate change are under development (Pyke and Fischer 2005; Carvalho *et al.* 2011; Hole *et al.* 2011; Davison *et al.* 2012).

Australia is no exception to these needs (Lindenmayer *et al.* 2010) even in parts of the landscape that remain relatively undeveloped (as recognised well ahead of time by Whitehead *et al.* 1992). Given a trend for south-eastward movement of the climate space of some Australian birds in the developed parts of southern and eastern Australia (VanDerWal *et al.* 2013), alternative land uses may compete with conservation for space. Ironically, climate change may ultimately force abandonment of some agricultural areas in the south, and pre-emptive identification of this possibility and early conversion to conservation purposes may be the best use of land in some areas. Similarly, managed retreat from rising sea levels can include acquisition of habitat for coastal breeding seabirds (Stralberg *et al.* 2011).

Maintain and improve habitat quality

As previously argued, high habitat quality as indicated by functional redundancy and structural diversity provides robust support for species – and the stressors that reduce habitat quality are likely to interact with climate change to produce worse outcomes for biodiversity than either alone. It follows that efforts to maintain, restore and improve habitat quality are key to conservation of many species in the face of climate change.

Management actions may include a range of traditional actions and methods such as control of weeds and feral animals and management of fire, as well as novel management actions such

as the creation of new habitats (see *Create new habitats* below). Traditional actions need to reflect the realities of climate change such that maintenance of the management *status quo* may not be satisfactory. For example, fire management needs explicit biodiversity goals that incorporate both the effect of climate change on fire regimes and the changing needs of biodiversity. We counsel for consideration of local circumstances carefully embedded within regional, national and international aspirations and realities, and avoidance of management based on glib generalities, or local actions taken without consideration of the wider context, both of which currently plague fire management in Australia (Clarke 2008; Lindenmayer *et al.* 2011).

The literature on the management of habitat and protected areas under climate change is already extensive and it is beyond our scope to review it here; it is necessarily either generalised and somewhat theoretical or consists of case studies that may lack wider applicability. None is prescriptive. Managers need to evaluate options for each area, embedding local perspectives into wider spatial (and temporal) context (Heller and Zavaleta 2009). Lemieux and Scott (2011) and Prato (2012) provide perspectives on the decision-making process for protected area managers additional to citations elsewhere in this chapter.

Identify, protect and expand refugia

Refugia as an effective strategy for species persistence There is ample evidence that refugia have facilitated the survival of species during past climatic changes (Taberlet *et al.* 1998; Tzedakis *et al.* 2002; Byrne 2008; Binney *et al.* 2009; Carnaval *et al.* 2009). Generally, refugia have been effective because of microclimatic differences between them and the regional climate, allowing species to persist in an otherwise hostile environment. Past effectiveness has prompted theoretical and practical interest in future applications (Keppel *et al.* 2012).

Refugia have been defined as ‘habitats that components of biodiversity retreat to, persist in and can potentially expand from under changing environmental conditions’ (Keppel *et al.* 2012). Their efficacy under climate change will depend on the degree to which a refugium protects the habitat and resources that species need – and sometimes the birds themselves – from direct impacts such as changes in temperature, precipitation, and extreme events and indirect impacts such as changes in species interactions and fire regimes. In addition, species persistence in refugia will depend on the degree to which the refugia meet species’ ecological and evolutionary requirements. For example, individual refugia need to be of sufficient size to sustain a population with adequate genetic diversity for population viability and to allow evolutionary processes (Ovaskainen 2002). To be effective, refugia must also have the appropriate spatial context in relation to the species’ current range, i.e. species must be able to access the refugium. In order to assess refugial quality, these factors need to be evaluated in relation to the particular threatening process and the species in question.

Identifying refugia

Identification of refugia for a species or suite of ecologically-similar species is a significant challenge; identifying refugia that will protect large numbers of species is considerably more difficult. Species’ differing requirements mean that the ideal refugia will depend on the species

in question. Three contrasting approaches can be taken to identify refugia: a top-down identification of areas where climate change will be ameliorated in the future (the ‘climate-based’ approach); a bottom-up identification of areas where species are likely to persist (the ‘species-based’ approach) (Ashcroft 2010); and the identification of past refugia (Pearson 2006).

For the climate-based option it is possible to identify areas throughout Australia that are likely to show the greatest buffering from extreme conditions of temperature, rainfall, fire and biotic interactions in the future. To do so requires an understanding of the factors limiting species and how these have related to climate (Dobrowski 2011). For example, areas where water is available constantly are likely to provide habitat and resource refuge for species in arid areas; and at high latitudes, a buffer against extreme minimum temperatures may act as a refugium (Dobrowski 2011). Refugia in the face of extreme events include areas that buffer species from cyclones (Murphy and Legge 2007) or heat waves (Welbergen *et al.* 2008). Studies predict spatial heterogeneity in climate change, in particular warming (Ashcroft *et al.* 2009); therefore, in order to predict the location of refugia, this heterogeneity must be measured or predicted at a scale at which species experience it (Pearson 2006; Gillingham *et al.* 2012). Terrain can influence the decoupling of local climates from that experienced by the region by three main mechanisms: cold air drainage, elevation, and slope and aspect effects (Dobrowski 2011). For cool-adapted species, refugia are most likely to occur where cold air pools at the bottom of slopes and on slopes that face towards the poles and thus away from the equator. Species needing a mesic environment will benefit from refugia that accumulate water and are protected from wind and desiccation (Dobrowski 2011).

A key component of long-term refugia is their stability (Jansson 2003; Carnaval *et al.* 2009; Mosblech *et al.* 2011), particularly for species with low dispersal abilities (Graham *et al.* 2006). Where temperature and rainfall is projected to be relatively stable into the future, fire regime and biotic interactions are also likely to be relatively stable; these places are more likely to accommodate the current species assemblages into the future than areas of lower stability. Identifying areas of greater stability also negates the need to identify refugial properties suitable for a range of species because stable conditions will continue to suit the species already there.

Substantial challenges exist in identifying the best metrics to identify and describe refugia in terms of climate buffering and stability and the temporal and spatial scales over which to measure them. Different metrics and scales can alter the refugial ranking of locations (Ashcroft *et al.* 2012). Once potential refugia are identified, the next stage is to link the refugia to the species they are likely to protect. This process involves identifying species’ climatic and habitat requirements, their proximity to the refugial areas and their relative dispersal abilities.

The species-based approach to identifying refugia involves the use of species distribution models relating current range to climate and environmental data and projecting these onto future climate scenarios to predict the location of suitable future habitat (Pearson and Dawson 2003). The advantage of this approach is that the climate is ‘biotically-scaled’, and decisions about refugia metrics such as the weighting of temperature over precipitation are circumvented. Advances in this approach include the use of topographically-corrected climate data in species distribution models (Dobrowski 2011). The species-based approach is only as reliable as the

species distribution models employed so that the accuracy and scale of climate and environmental variables is critical.

Past refugia will only be useful for management of species under climate change if the threats driving species to refugia in the past or present are the same as those in the future, and if the refugia so identified will retain key refugial characteristics into the future. Areas that provided greater climatic stability in the past are those with the highest level of endemism, and these may change the least with future climate change (Eeley *et al.* 1999; Jansson 2003). Whilst refugia based on the retention of moisture may be similar under climate change to past drier periods, global warming of 3–5°C creates challenges that have no precedent during recent geological time, limiting inference based on past refugia.

Spatial context of refugia

Maintaining connectivity of habitats throughout the landscape will be an important conservation strategy to facilitate species movement to refugia, especially as the spatial location of refugia themselves may shift. Mosblech *et al.* (2011) suggested that (micro)refugia are important for conservation, not for the species they contain, which are likely to turn over with time, but for species not yet identified as at risk. Understanding species dispersal abilities in relation to potential refugia is a crucial consideration. Active facilitation of dispersal to refugia is likely to be an important management option (see below).

Past refugia in Australia

Rainfall is and has been a major limiting factor in many of Australian landscapes via its contribution to soil moisture to support vegetation, and in some arid areas as a surface resource for birds to drink. Areas that have attracted rain or retained moisture even during arid periods have been of particular importance as refugia for Australian biodiversity. During the Pleistocene – the last c. 1.8 million years until c. 10 000 years ago – Australia was subject to oscillations between warm-wet interglacial and cool-dry glacial ‘Ice Age’ periods. Widespread aridity, with extensive drying of rivers and lakes, developed during more recent glacials (Nix 1982). During glacial periods, areas of relatively high rainfall provided refugia and became zones of speciation (Keast 1961; see Ford 1987a,b for Australian birds). The wetter evolutionary refugia were generally around the continental periphery where mountains are more likely to capture rainfall, around the drainage basins of major river systems, but also in central Australian mountain ranges. Even today, when the climate is much wetter than during recent glacials, many bird species in Australia are restricted to moister areas and appear unable to disperse across arid barriers. During arid periods, rainfall was also more seasonal and seasonal water deficits are likely to have placed further constraints on some bird species (Nix 1982).

Rocky ranges and deep gorges are well documented refugia in Australia. Over 170 plant species are restricted to the sandstone plateau of western Arnhem Land (Woinarski *et al.* 2006). A few deep gorges in the Blue Mountains near Sydney provide refuge for the relict Wollemi Pine (*Wollemia nobilis*), though the fossil record shows that *Wollemia* species were once widespread (Offord *et al.* 1999). A suite of Australian reptile, amphibian, bird and mammal species are rocky habitat specialists (Freeland *et al.* 1988), and many species occur

in higher abundance in rocky areas (Bowman *et al.* 2010). Rocky habitats provide refuges for mammals occurring in the savanna during periods of low groundwater (Braithwaite and Muller 1997).

As there have been no major recent episodes of mountain-building in Australia, current ranges have provided refugia throughout the Pleistocene glaciations and for some taxa for even longer. The Arnhem Plateau has an unbroken presence in the landscape for over 100 million years (Hoatson *et al.* 2000).

Sources of water in a dry landscape

Because they receive moisture by run-off from other parts of the landscape, and because they often also have deeper and more moisture-retentive soils, riparian areas are particularly important for maintaining a large variety and abundance of birds and other wildlife in otherwise water-limited Australian environments (Williams *et al.* 1994; Bentley and Catterall 1997; Mac Nally *et al.* 2000; Soderquist and Mac Nally 2000; Woinarski *et al.* 2000; Lynch *et al.* 2002; Klein *et al.* 2009). Rock pools in riparian environments where soils are skeletal may provide permanent water sources. Mountains usually receive more rainfall than adjacent lowlands, slopes generate run-off that concentrates water in valleys, and variation in aspect provides sites that are relatively sheltered from heat and solar radiation. Depending on their structure, rock outcrops may also provide moist refugia due to localised run-off and concentration of moisture in sheltered crevices.

Moist refugia often also provide protection from fire because the additional moisture reduces the flammability of vegetation, topographic effects that favour moisture retention may also shelter vegetation from fire, and rocks provide fire-breaks. Moist refugia have often become refugia for fire-sensitive species in Australia (Bowman *et al.* 2000; Woinarski *et al.* 2006). Rocky outcrops with less protection from fire have lower endemism than those with greater protection (Bowman *et al.* 2010). The fire-sensitive conifer *Podocarpus grayae* is restricted to a few canyons in western Arnhem Land (Russell-Smith and Bowman 1992). A number of species of grasswren (*Amytornis* spp.) require mature (long-unburnt) spinifex (*Triodia* spp.) hummock grasslands and are restricted to inland and north Australian mountain ranges (Christidis 1999) where rock outcrops act as firebreaks.

Summary

Identifying refugia that will be crucial for the persistence of many species in the face of climate change is a high priority. Further technical development of methods for identification is required. A combination of approaches – climate-based, species-based and historical – is likely to be optimal. Building in knowledge of species ecology and evolutionary processes is an additional and important challenge. Particularly in an Australian context, both thermal and hydric considerations need to be incorporated into refugia identification. Once potential refugia are identified, their management to restore, maintain and enhance the characteristics that generate that potential, will be critical.

Maintain and extend ecological connectivity

Connectivity describes the structure of landscapes that enable birds and other wildlife

including plants to move through them to reach new habitat patches. Whilst this may involve continuous corridors of habitat (Beier and Noss 1998), this is not always so (Fischer and Lindenmayer 2006); with particular application to many birds, for whom size and flight enhances the capacity to cross gaps, habitat may be suitably arranged in ‘stepping stones’. It is also worth noting that habitat for one species may be a barrier for another; for example, some rainforest specialists may be unwilling to cross open country, but for the Emu (*Dromaius novaehollandiae*) rainforest may be an impassable barrier.

In the context of climate change and bird conservation, ecological connectivity serves two management goals (Table 1; also Opdam and Wascher 2004):

- to enhance the *resilience* of populations in the face of catastrophic local events and other local habitat change by the formation of metapopulations (demonstrated both empirically and by modelling studies as reviewed by Van Teeffelen *et al.* 2012), and
- to facilitate change in the range of populations in *response* to changed circumstances (Travis 2003).

The structural and spatial analogue of the two purposes is that the former involves connectivity between similar habitats that may be in the form of narrow corridors at local scales following fragmentation or where habitat is naturally patchy (Saunders and Hobbs 1991), whereas the latter involves connectivity along ecological gradients often at much larger scales of width and distance. There are functional analogues in the way species use such connectivity as well. In local scale between-patch connectivity, corridors do not necessarily provide more than avenues for the transit of individuals between patches, they tend to be of most use to species that disperse repeatedly over the course of a life time and for recolonisation after local extinction, and movements have little or no inherent directionality. In contrast, macroscale connectivity along ecological gradients frequently implies directional movements (VanDerWal *et al.* 2013) over multiple generations, requiring habitat suitable for feeding and breeding of substantial populations at multiple locations along the way. An important case is species shifting to reach long-term refugia. Thus, macroscale connectivity is more than simply the sum of many local connectivities. However, there is also intergradation in structure and function between the two purposes, and in particular we note that the movement of migrants may be repeated many times in the life of an individual yet are (bi-)directional across ecological gradients and are often at large spatial scales.

For terrestrial birds, local connectivity in fragmented landscapes may be provided by roadside, riparian or windbreak corridors, or scattered trees in open areas such as farmland. That connectivity is compromised by fragmentation is evidenced by the selective local absence (implied extinction) of some species from smaller remaining fragments (see Saunders and de Rebeira 1991; Loyn 1985; Rankmore and Price 2004 and Radford and Bennett 2006 for examples among Australian woodland and forest birds). The rule-of-thumb that when less than 10–30% of habitat is retained during land clearing, biodiversity declines steeply (see Cooper and Walters 2002; Radford *et al.* 2005 for Australian examples), is based on static habitat; modelling suggests that the percentage requirement doubles to 20–60% when habitat is dynamic in time or space (Van Teeffelen *et al.* 2012) as is likely to be the case under climate

change.

In fragmented landscapes there is abundant evidence that a range of birds use corridors, along with information about the circumstances in which they do so, both in Australia (e.g. Saunders and de Rebeira 1991; Bentley and Catterall 1997; Fischer and Lindenmayer 2002; Kinross and Nicol 2008; Lentini *et al.* 2011) and elsewhere. Modelling provides strong support for the role of such connectivity between patches in generating metapopulations and thus enhancing population viability in the face of habitat instability (reviewed by Van Teeffelen *et al.* 2012). However, empirical demonstration of metapopulation functionality is by definition hard to obtain (Hobbs 1992). Gilbert-Norton *et al.* (2010) provided a meta-analysis of experimental studies that tested the proposition that ‘corridors increase movement of plants and animals between habitat fragments’ and identified only seven studies involving birds. They found that corridors do increase movement, though the effect was weaker among birds than other taxa, an effect they attributed to the ability of many bird species to fly across open spaces. The effect is species-specific. For example, in the wet–dry tropics of north-west Costa Rica, Gillies and St Clair (2008) found that the Barred Ant-shrike (*Thamnophilus doliatus*) ‘returned faster and with greater success in riparian corridors’ and detoured to track forest corridors, whereas the Rufous-naped Wren (*Campylorhynchus rufinucha*) used fencerows. In south-eastern Australia, the Brown Treecreeper (*Climacteris picumnus*) is more likely to use scattered trees and less likely to use corridors than by chance and compared to four co-existing species (Doerr *et al.* 2011).

In fragmented landscapes, connectivity is generally assumed to be the original and preferable state, with management aimed at restoring connectivity, and this has been proposed as a key management strategy in the face of climate change in Australia (Lindenmayer *et al.* 2010; Turton 2012) and internationally (Krosby *et al.* 2010; Groves *et al.* 2012). However, there are some risks associated with this as connectivity may also allow the spread of threatening processes such as dominant or invasive species, disease, and fire, increasing rather than decreasing the risk of metapopulation extinction (Hess 1994; Brudvig *et al.* 2012). Some of these risks, particularly those posed by disease and invasive species, may be greater in highly disturbed environments (Jackson and Pringle 2010). However, Reed (2004) found that the value of connections among populations outweighed the risk from dispersal of disease. Whilst the overall benefits of local-scale connectivity are indubitable, it may be appropriate when planning establishment or restoration of connectivity to consider the risks and how they may be ameliorated. Where, as a special case, management for connectivity is planned for the benefit of one or a few threatened populations or species, assessment of these risks is imperative. In such circumstances assisted colonisation (see below) may be more cost-effective (Lubow 1996; Hulme 2005) and less risky. For further information about connectivity in fragmented Australian landscapes, see Saunders and Hobbs (1991), Bennett (2003), Doerr *et al.* (2009) and Martin *et al.* (2012a).

As demonstrated in this study, the climate envelope of some species is likely to shift, in some cases to the extent that it will no longer overlap with the current range. The extent to which this will correspond with actual or required changes in distribution, and the mechanisms underlying this, remain unclear. However, there is abundant evidence that some birds have already tracked shifting climate envelopes (e.g. Rappole *et al.* 2011; Brommer *et al.* 2012;

Pautasso 2012).

The successful *response* of some species in the form of range changes will undoubtedly be enhanced by maintenance and restoration of ecological connectivity across ecological gradients. Indeed, fragmentation across ecological gradients in Australian and other rangelands has already adversely affected wildlife (and humans) (Hobbs *et al.* 2008). The realisation of the importance of this form of connectivity (e.g. Whitehead *et al.* 1992; Brodie *et al.* 2012; Lindenmayer *et al.* 2010) has led to the development of several plans to re-establish or maintain ecological connectivity at larger scales (Whitten *et al.* 2011). At an international scale, the *Gondwana Link Corridors* are noteworthy (de Wit and Anderson 2003). Australian examples include:

- the Alps to Atherton (Great Eastern Ranges) Initiative (DECC 2007);
- the Gondwana Link project in Western Australia (Gondwana Link 2012);
- the Territory Eco-link project in the Northern Territory (PWCNT 2012);
- the Trans-Australia Eco-link, an extension of the Territory Eco-link through South Australia to the south coast of the continent (DSEWPC 2012);
- the Habitat 141° Ocean to Outback project along the border between South Australia, Victoria and New South Wales (Habitat 141° 2012); and
- re-establishing altitudinal links in the Wet Tropics of north Queensland.

Stepping stones are key to the survival of many migratory species and by inference local nomads as well. Very little is known about the necessary configuration of these for species persistence and implications are likely to be species-specific, ranging from scattered trees several hundred metres apart (Fischer and Lindenmayer 2002; Manning *et al.* 2006; Doerr *et al.* 2011) to feeding sites for migratory shorebirds that may be separated by thousands of kilometres (Battley *et al.* 2012). Price (2006) provided rare empirical data on the movement of frugivorous birds between naturally-scattered patches of rainforests in northern Australia.

At both local and larger scales, questions arise as to which birds will use and depend on connections that are susceptible to management, and on the type of connections worthy of management effort. Many Australian birds are nomadic or migratory at local to national scales (Keast 1968; Davies 1984; Ford 1989) and well adapted to crossing even large gaps in favourable habitat. Direct inference about gap-crossing ability from even radio-telemetered birds is difficult, but the Grey Teal (*Anas gracilis*) are clearly capable of travelling hundreds and possibly thousands of kilometres between waterbodies (Roshier *et al.* 2006), as doubtless are many other waterbirds that exploit ephemeral waters in the Australian arid zone. At the extreme end are some populations of Bar-tailed Godwit (*Limosa lapponica*) which undertake an annual non-stop trans-oceanic flight of more than 11 000 km (Battley *et al.* 2012). At the other extreme, some resident woodland birds of south-eastern Australia appear reluctant to cross treeless gaps of more than several hundred metres or a kilometre (Cooper and Walters 2002; Robertson and Radford 2009; Doerr *et al.* 2011). The Palm Cockatoo (*Probosciger aterrimus*) appears reluctant to cross water barriers (Garnett 1991). Roads may even inhibit the movements of some rainforest interior birds (Laurance 2004; Thinh *et al.* 2012).

Care should be taken when applying the above generalisations to perceptions and management decisions about landscape connectivity. Gap-crossing constraints are rarely absolute and best understood in terms of relative resistance to movement (Richard and Armstrong 2010; Taylor and Goldingay 2010; Desrochers *et al.* 2011). However, probabilistic conclusions about the frequency distribution of dispersal distances may not adequately describe infrequent but important longer-distance movements (Clark 1998; Higgins and Richardson 1999). Day-to-day movement patterns may not extrapolate well to the capacity for long-distance dispersal as these often make quite different contributions to life history (Van Dyck and Baguette 2005). The net result may be that the capacity of birds to cross barriers is greater than observed and reported. On the other hand, individual dispersal does not necessarily constitute successful dispersal of a population, and the physical capacity to fly distance does not necessarily indicate a willingness to cross inhospitable habitat. The limits to gap-crossing ability in birds are both physical and behavioural (Harris and Reed 2002). Whilst noting that exceptions may abound, some generalisations based on species traits are possible:

- 1 small species have less gap-crossing ability than large species (Grubb and Doherty 1999; Creegan and Osborne 2005; Lees and Peres 2009);
- 2 habitat specialists, and especially those of woodland or forest interiors or understorey, are less likely to cross gaps (Harris and Reed 2002), though the effect appears to be weak or non-existent in birds of some Australian open forests and woodlands (Mac Nally *et al.* 2000; Shanahan and Possingham 2009); and
- 3 resident species are less likely than migrants or nomads to cross gaps (Harris and Reed 2002).

Herein lies a management conundrum. The species most in need of improved connectivity will at times be those least able to make use of it; conversely, improving connectivity is particularly beneficial to those with greater dispersal capacity (Johst *et al.* 2011). In practice, management to improve connectivity will be of most value to those species for whom the pre-management level of connectivity was marginal. For both the resilience of populations and their ability to respond to climate change, improvements to connectivity will serve many useful purposes, but even if the land and financial resources available for it are made far more extensive than they are at present, landscape connectivity will not serve all bird connectivity needs. Improving connectivity and assisted colonisation are complementary strategies for the spatial management of birds in the face of climate change.

Create new habitats

The need for new habitats under climate change is perhaps most obvious when the climate envelope suitable for a species moves to areas where no suitable habitat currently exists, as already discussed in the section *Expand the protected area network*. Given the time lag involved in the creation of habitats such as forest or woodland, the early identification of target species and areas such as undertaken preliminarily in this study, is imperative.

An additional and conceptually more challenging context for the creation of new habitats

will arise within existing natural areas. As climate changes, some vegetation types will become maladapted and the optimal strategy for the conservation of biodiversity may be to modify or replace them. An extreme case may arise following widespread death of trees due to extreme disturbance such as drought (e.g. Fensham *et al.* 2009). To facilitate regeneration of the same species (and especially of local provenance) may be to consign the area to further exposure, and lack of vigour may compromise the ability of vegetation to support birds. The possible facilitation of forest transitions has already been flagged in a forestry context (Millar *et al.* 2007). Similar thinking may also be necessary for biodiversity conservation.

Use of non-local plant species for revegetation is a logical but more radical extension of use of non-local provenance of local species. Traditional emphases on local provenance will in many climate change circumstances prove less than optimal for biodiversity conservation (Sgrò *et al.* 2011). The issue of choice of species and provenance will doubtless generate much discussion and research in the future; Australian perspectives on use of non-local provenance for revegetation in the face of climate change are already emerging (Booth and Williams 2012; Booth *et al.* 2012; Hancock and Hughes 2012). An analogous local-provenance situation may also arise in the conservation of bird species – see sub-section *Enhance the genetics of populations* below – and even among bird species in the maintenance of ecosystem functions (Dunwiddie *et al.* 2009).

ACTIONS: FACILITATE THE RESPONSE OF WILD POPULATIONS (INTENSIVE MANAGEMENT)

For many species, habitat protection and management alone will be insufficient to secure their conservation. A landscape may appear healthy and its management may be best practice, but without targeted action it can still lose species. In this section, we describe management actions that are species-specific and intended to facilitate the *response* of species to threats generated by climate change. Under current management practice in Australia and in contrast to those actions described in the previous section, these will generally be implemented after formal and usually intense evaluation of the options and consequences by dedicated personnel, often in the form of a recovery team.

Assist colonisation by translocation of individuals

As climate changes, situations will undoubtedly arise in which the choice for wildlife managers and society is stark: assist colonisation of new locations or lose – or risk losing – a species (or subspecies) (Minteer and Collins 2010; Thomas 2011; Schwartz *et al.* 2012). This study has identified several possible candidates among the Australian avifauna, in particular habitat specialists with limited dispersal capacity and small current ranges. More subtle, but perhaps equally important situations will arise in which the choice will be between assisted colonisation and loss – or risk of loss – of ecosystem function (Hutton *et al.* 2007; Kreyling *et al.* 2011; Lunt *et al.* 2013).

Assisted colonisation (defined broadly to include reintroduction, contra Seddon 2010)

already plays an important role in the conservation of threatened species in Australia (Short 2009; Sheean *et al.* 2012) and elsewhere, and its application to mitigation of the impacts of climate change on biodiversity is a logical extension of this practice (Thomas 2011). Its potential role in dealing with climate change impacts on taxa is 3-fold:

- to reduce extinction risk by creating additional populations (the insurance policy);
- to restore populations following catastrophic local events (reintroduction); and
- to facilitate change in the range of populations in *response* to changed circumstances.

Of these, only the third raises new and largely untested challenges because it is more or less unique to climate change mitigation (Burbidge *et al.* 2011). Recognition of this new role for assisted colonisation is now widespread among conservation biologists (e.g. Hulme 2005; Hoegh-Guldberg *et al.* 2008; Mawdsley *et al.* 2009; Early and Sax 2011; Loss *et al.* 2011; Thomas 2011). The contribution of assisted colonisation in dealing with climate change impacts has already been recognised and successfully implemented for two British butterfly species (Willis *et al.* 2009), and its potential assessed for a range of other species (e.g. Morrison *et al.* 2011; Broome *et al.* 2012; Fordham *et al.* 2012).

Notwithstanding, there are substantial ethical and practical concerns centred on the risks involved and concerns about the manipulative approach to wildlife management (Minteer and Collins 2010; Schwartz *et al.* 2012). These are occasionally crystallised as outright opposition to any form of assisted colonisation (Ricciardi and Simberloff 2009) but more often as concern about *ad hoc* and poorly-planned assisted colonisations (e.g. Schwartz *et al.* 2012; Sheean *et al.* 2012) and about translocation as a method for solving human-animal conflicts (Fischer and Lindenmayer 2000). The latter has nothing to do with climate change mitigation or threatened species management and should not be allowed to detract from legitimate use of assisted colonisation (Craven *et al.* 1998). Potential risks of assisted colonisation are to the source population, to the founder individuals, and to the recipient ecosystem. For a considered discussion of these and related issues, we recommend Schwartz *et al.* (2012). Further key conceptual reading includes Hoegh-Guldberg *et al.* (2008) and Richardson *et al.* (2009), who provided evaluative frameworks and/or decision trees for assisted colonisation, and McDonald-Madden *et al.* (2011) for their exploration of the complex issue of timing (wait until a taxon is threatened or pre-empt the risk?). Kreyling *et al.* (2011) provided a detailed tabulation of the *pros* and *cons* of assisted colonisation for climate change mitigation, Weeks *et al.* (2011) elucidated the underlying and often unstated genetic aims associated with a range of assisted colonisation scenarios, and Burbidge *et al.* (2011) considered the policy implications from an Australian perspective.

Perhaps the greatest potential ecological risk with assisted colonisation is that the translocated species will have a negative impact on the recipient ecosystem or species therein, including the risk that it will become invasive. The link between characteristics of successful invasions and successful assisted colonisations is notable and informative (Cassey *et al.* 2008), allowing the distinction between inherent traits favouring invasiveness and those that relate to human manipulation such as the number of individuals released at the new site. Careful prior evaluation of the risks should also include an evaluation of the risks of not

undertaking the assisted colonisation (Hoegh-Guldberg *et al.* 2008) and of alternatives to assisted colonisation (Loss *et al.* 2011). In the context of climate change mitigation and threatened species management, the risks may not be great because the species being translocated are by definition narrow endemics and thus generally not invasive (Thomas 2011). Further, invasiveness has primarily occurred when assisted colonisations have occurred across continental or biome divides, but this is not what is (mostly) proposed for the mitigation of climate change impacts (Hoegh-Guldberg *et al.* 2008; Kreyling *et al.* 2011; Thomas 2011). Mueller and Hellmann (2008) found that the probability of invasiveness is low with intra-continental assisted colonisation, but the consequences when a species becomes invasive are as great as with inter-continental assisted colonisations.

The reported success rate for attempts to assist colonisation of wildlife species in Australia is ~50% though the real rate is likely to be lower due to non-reporting of failed outcomes (Short 2009; Sheean *et al.* 2012). Some failures are inevitable and we do not regard failure as a reason for criticism or even of concern if planning, monitoring and reporting were adequate. With proper project design, assisted colonisation is necessarily experimental (Burger and Both 2011) and failure can (and should) be informative (Bennett *et al.* 2012); such an approach is readily understood as part of the adaptive, learning-by-doing management framework (Walters 1986) that is critical to the longer-term success of climate change mitigation.

Enhance the genetics of populations

In this study, we have identified several scenarios in which the climate space of a subspecies is predicted to disappear but that of another subspecies extend to embrace that of the former. This suggests the possibility that a decline in one subspecies may be mitigated by the introduction of genetic material from another. This scenario can be generalised to occurrence among populations including those not formally distinguished as subspecies. In the parlance of Weeks *et al.* (2011), the objective would be to enhance genetic adaptation in a recipient population.

This scenario raises questions about local adaptation, outbreeding risks and the distinctness and integrity of subspecies or populations, as well as disease transmission, although the issues are far more tractable than may seem to be the case (Weeks *et al.* 2011). Numerous potential problems may be avoided or minimised by moderating the rate of genetic influx typically to little more than one genetically-effective individual per generation (Weeks *et al.* 2011). Of course, rigid adherence to the integrity of subspecies or populations is not compatible with enhancement of genetic adaptation, but will be inappropriate where climate change threatens the persistence of the population. More broadly, for conservation purposes subspecies are most relevant as the units of evolution (Fraser and Bernatchez 2001). Rigid adherence to them, however, may in some circumstances undermine the very process of adaptation (to climate-changed circumstances) that we seek to conserve.

Manage other threatening processes

We have already argued in detail that one of the greatest threats to biodiversity associated with climate change is the interaction of climate change with existing threats, and that this creates an

opportunity for mitigation of climate change impacts where direct attention to its impacts may prove intractable. A key issue arising is that we may need to lower acceptable thresholds of other stressors because of the interaction (Hansen *et al.* 2010). A good example in the marine sphere is to increase the escapement requirements for fisheries, i.e. the number of fish that fishers need to leave behind, to allow a greater share for marine birds (J.Croxall *in litt.*). For other taxa, the acceptable losses to threats may be lower under climate change. Increasing fecundity and/or survival by more intensive management of existing stressors may compensate for the less-controllable losses resulting from climate change. Thus, an important adaptation strategy for many taxa, particularly threatened species for which intensive management is already under way, is to continue to with current practice, only more effectively.

ACTIONS: PRESERVE POPULATIONS – THE LAST RESORT

Save species in captivity

As climate changes become more extreme with time, it is almost inevitable that some species and many populations will go extinct in the wild even where the resources are available for intensive management. For birds more than some other forms of biodiversity (notably plants), this raises a complex series of ethical and practical questions. Should species and subspecies be ‘preserved’ in captivity, and if so, with what objectives and at what cost? With extinction rates bound to escalate, society must consider *ex situ* conservation techniques (Pritchard *et al.* 2012) and determine the level of investment it wishes to make.

It is usually an implicit aim in any last-resort act of preservation to ultimately re-establish the species in the wild (e.g. Orange-bellied Parrot, *Neophema chrysogaster*; Martin *et al.* 2012b). There have been occasional successes in re-establishing wild populations using only captive-bred individuals (Mallinson 1995) and some ongoing attempts to do so (e.g. Californian Condor, *Gymnogyps californianus*, Walters *et al.* 2010). For many species affected by climate change, however, returning birds to the wild from a captive population may not be realistic, or even ethical – at least for the many centuries it may take for climate to ameliorate after the current phase of warming. There may be no habitat remaining, that which does remain may be either fully occupied by other species, or assisted colonisation may be socially unacceptable. Further ongoing management of the wild population may be more expensive and risky than maintaining a captive population. Planning to maintain a captive population in perpetuity, however, comes with a raft of major issues including the ratio of costs to benefits, genetic integrity and atrophy, and technical difficulties.

Captive breeding has played a key role in the intensive management in the wild of some threatened species (Butchart *et al.* 2006) but is generally considered an option of last resort among the tools available for conservation biology (Snyder *et al.* 1996). Much may be learnt from these efforts with application of last-resort preservation. Snyder *et al.* (1996) listed the following problems with captive breeding for conservation: (1) establishing self-sufficient captive populations, (2) poor success in reintroductions, (3) high costs, (4) domestication, (5)

preemption of other recovery techniques, (6) disease outbreaks, and (7) maintaining administrative continuity.

The question of costs and benefits of captive breeding for last-resort preservation is an ethical question that cannot be solved in a technical report. In this analysis we have necessarily assumed that society will retain the same value for subspecies that is currently expressed through threatened species legislation – i.e. that there should be an ambition to retain genetic variability in an untainted form, even if no longer in its wild state. We do not know whether this is what society wants to happen – whether there is a belief that only wild birds have value, how much society is willing to pay, either directly or through government, or the type of prioritisation that society is willing to adopt. For instance, society may want to save species but not subspecies, birds but not insects, species perceived as useful or aesthetically pleasing but not those from which no direct benefit is gained. Resolution will come through public debate, it being assumed that the current *laissez faire* approach to prioritisation of biodiversity conservation will be replaced by a more methodical approach (e.g. Joseph *et al.* 2009). However, funds are already limited to the extent that prioritisation is being considered to maximise conservation benefit, which may result in extinction of expensive and intractable species. Captive breeding for conservation purposes can be expensive (e.g. Restani and Marzluff 2001; Walters *et al.* 2010) so any decision to adopt comprehensive captive breeding will need a far larger budget.

In the meantime there are planning and management consequences of recommendations that emerge from existing legislation. Of these, the most important for climate change management is that there needs to be planning to establish captive populations of any taxon for which extinction is likely. This may need to be sooner rather than later. The issues with timing are the same as those with the timing of assisted colonisation (McDonald-Madden *et al.* 2011) – establishment and maintenance costs need to be set against the need to establish the population soon enough to maximise the genetic variability of the captive population, and the danger of reducing still further the chances of adaptation among the wild birds by needing a significant number of individuals to establish the founder flock (as may have happened with the Orange-bellied Parrot; Martin *et al.* 2012b). However, a range of techniques are available to prevent or minimise the risk to the wild population, including collecting individuals that contribute little or nothing to the viability of the wild populations (e.g. floaters, Brown 1969) or for which survival prospects in the wild are low (e.g. eggs, Smales *et al.* 1992). If intending to establish a population that can be sustained in captivity indefinitely, the start-up costs are less important because they will be amortised across centuries. This implies that, in contrast to assisted colonisation, captive populations of what appear to be doomed taxa should be established as soon as possible.

Another consequence of attempting to retain all subspecies is that genetic supplementation of wild populations will need to be preceded by establishment of a captive population of the original subspecies. This leads to questions about the genetic integrity of the captive populations. Captive breeding can have profound effects on the fitness of subsequent generations when they are released to the wild (Araki *et al.* 2007), an effect that appears to vary markedly between species. Many bird species have extended histories of captivity without replenishment from the wild, but often differ so much from the wild progenitors that

they are barely recognisable as the same species, let alone the same subspecies. Adaptation to captivity can occur in a single generation of fish, although the genetic basis for adaptation is unknown (Christie *et al.* 2012). In birds, the expression of phenotypic traits in long-term captive populations are at least as different from their expression in wild populations as the differences between subspecies, even if life-history strategies and investment trade-offs are relatively stable (Tschirren *et al.* 2009).

Genetic issues are likely to be particularly important if costs are to be mitigated through private breeding, including mixed public/private cooperative strategies (Cannon 1996). The involvement of private individual breeders may allow a much larger effective population size to be maintained, but genetic management is likely to be more difficult (El Alqamy *et al.* 2012), especially given the high commercial value of hybrids and mutants for many captive breeders. Overall, the technical challenges of maintaining the genetic integrity of subspecies in captivity indefinitely are formidable and are yet to be solved.

For species, loss of genetic integrity through interbreeding among subspecies or populations is less of an issue than loss of variability. It has long been known that inbreeding among captive populations needs assiduous management if it is not to cause inbreeding depression and loss of genetic diversity (Ballou and Ralls 1982). While genetic drift, which is likely to be the main problem, can be countered by the occasional introduction of immigrants, such immigrants will no longer be available for species extinct in the wild so multiple captive populations are likely to be necessary (Lacy 1987). What is apparent is that the larger the founder population the better. While there is no magic number for the minimum population required (Flather *et al.* 2011), most populations, particularly those that were once large and have declined, are likely to need a broad sample of the existing population to conserve the full genetic variability (Jamieson and Allendorf 2012). That will be best achieved by establishing captive populations sooner rather than later while it is still possible to augment from the wild.

Finally, many technical issues related to breeding in captivity are still to be resolved. While there have been major advances in the keeping and breeding of many species, the proportion of taxa for which techniques are more than experimental is very small, with a tendency for zoos to concentrate on species that are easy to keep (Balmford *et al.* 1996). The physical requirements for keeping some birds are daunting. While albatrosses, for instance, can survive in captivity, there is no aviary large enough for them to fly.

Store germplasm

Genome resource banks store systematically-collected germplasm, embryos, blood products, tissue and DNA. These have long been thought of as the last option for the conservation of both wild biodiversity (Wildt *et al.* 1997) and preservation of rare breeds of poultry (Petitte *et al.* 1997).

The technologies for cryogenic storage of avian germplasm are in their infancy (Glover and McGrew 2012), with major impediments until recently. For some time it has been possible to store avian semen (Blackburn 2006), but the recent isolation and cryostorage of chicken primordial germ cells (van de Lavoie *et al.* 2006) makes it possible to preserve the female germ cell line. The technique has already been applied to Houbara Bustard *Chlamydotis undulata* with a live chick produced from Houbara PGCs transplanted into a chicken host

embryo (Wernery *et al.* 2010).

Currently, the efficiency of PGC collection is very low, with multiple embryos required to develop a single line of mixed lineage (Macdonald *et al.* 2010), but one can envisage that the technology will improve. If perfected, these technologies will not only be cheaper than captive breeding but will avoid many of the problems, particularly for subspecies whose nuanced variation may be lost within a few generations. Cell lines should be able to preserve the exact genetic make-up of the original form. It should also be possible to rapidly generate multiple offspring of threatened taxa where captive breeding programs take many generations. However, there remain formidable problems of creating wild birds that behave similarly to the original and fulfil the same ecological roles. Nevertheless, within the next 50 years, one can anticipate germplasm storage being an important tool in biodiversity conservation.

Store DNA code

DNA coding must offer the very-last resort for the conservation of biodiversity notwithstanding rapid advances in the technology. The cost of sequencing the entire human genome is now down to near \$1000 and simple genomes can be sequenced on a device not much larger than a USB for under that amount. The main limitation may soon be the computing power to organise such data, but that too is improving rapidly. As with germplasm storage, one can envisage the information in the genomes of threatened species being analysed and stored relatively cheaply, but the process of transforming it back into a living organism is a technological question that is a long way from solution. Whether the information alone can be considered a form of biodiversity conservation remains a philosophical question. Nevertheless storage of code, which does not have to be derived from living specimens, may eventually be a means for retaining taxa that would otherwise be lost.

ACTIONS: UNDERSTAND WHAT IS HAPPENING AND PREPARE FOR WHAT MAY HAPPEN

Monitor bird populations

While models of sensitivity and exposure can provide a guide as to which taxa may be affected by climate change, only monitoring will demonstrate which change is occurring and enable timely action to be taken (McDonald-Madden *et al.* 2011). For very few species is there sufficient information to predict with confidence how they will respond to climate change, especially given likely interactions with other environmental changes. Monitoring of biodiversity in Australia has been patchy (Lindenmayer and Gibbons 2012; Lindenmayer *et al.* 2012). Birds, however, have the advantage of having a wide following among the public. For the current book, the majority of records were from BirdLife Australia which has produced two atlases of Australian birds that provide baseline distributions of most species (Blakers *et al.* 1984; Barrett *et al.* 2002, 2003). Some rare species can be monitored effectively using presence/absence records (Joseph *et al.* 2006), while the increasingly sophisticated use of bird lists (Franklin 1999; Szabo *et al.* 2010, 2011) allows identification of trends that would

otherwise be difficult and expensive to detect. The costs of monitoring can also be reduced by targeting regions where exposure to climate change is expected to be particularly intense, either through dedicated surveys or by encouraging volunteers to search for target taxa in particular regions. However, for some taxa, and especially those that are rare and threatened, only dedicated surveys are likely to provide enough data to detect trends and partition sources of change among climate change, inherent climatic variability, and other environmental change.

Monitor habitats and threatening processes

At times, monitoring of birds alone will be of insufficient precision to detect change, and changes detected in bird populations will require context for interpretation including attribution to climate change. Thus monitoring of the taxa of interest frequently needs to be accompanied by monitoring of the habitat in which they live and/or the processes threatening that bird or habitat (Lindenmayer and Likens 2010). This may be as simple as remote sensing to detect habitat loss (Zerger and McDonald 2012) and changes to coastlines. At other times it will require intense research. Monitoring of habitat quality may also allow detection of a threat long before there is any decline in the species that relies on it, such as with the gradual loss of nest trees used by the Superb Parrot *Polytelis swainsoni* (Manning *et al.* 2013), allowing time for prophylactic management.

Investigate the ecology of species and communities

Many communities are temporary aggregations of interacting species that may disassemble as the constituent species move along different gradients of change (Gilman *et al.* 2010). The ranges of many species are constrained by biotic interactions (Pigot and Tobias 2013) which are likely to change as membership of communities alters. Thus, there is an imperative to be aware of the nature and trend of interactions among species, not just of trends in populations, their landscapes and their threats. Long-term monitoring of the strength of interactions (e.g. Krebs *et al.* 2001) may provide key information about the likely effects of climate change. The complex nature of interactions means they have great potential to produce ecological surprises.

The White Tern *Gygis alba* provided an interesting example of an interaction that suggests an outcome under climate change quite contrary to simpler assessment of exposure and sensitivity. The species is more successful at catching prey during periods of low productivity when larger competitors migrate away from breeding islands (Catry *et al.* 2009). Thus, counter-intuitively, they may benefit from the reduction in marine productivity near some breeding islands that is predicted to arise from climate change. A study of petrel foraging ecology (Baduini and Hyrenbach 2003) suggests an aspect of sensitivity with a potential outcome of climate change that might otherwise be missed. When feeding young, Black-winged Petrel *Pterodroma nigripennis* undertake foraging trips of several day's duration (Hutton and Priddel 2002), and are thus more likely to be able to track changes in marine productivity induced by climate change than similar species such as Gould's Petrel *P. leucoptera* that undertake shorter foraging trips (O'Dwyer *et al.* 2007). Extensive ecological information should be the basic underpinning of all wildlife management, including climate change adaptation.

Summary

Understanding existing trends and how they might play out in the future relies on sustained monitoring combined with fine-scale modelling. Such actions are an important part of any managers' toolbox but their value will be accentuated as climate changes (Morecroft *et al.* 2012). Various frameworks have been developed to help make decisions under uncertainty (Polasky *et al.* 2011) but all are improved with a sound knowledge of states, interactions and trends in the existing system. Ultimately, decisions should be based on optimal use of the resources available (Wintle *et al.* 2011), an approach that also includes analysis of the costs and benefits of monitoring itself (Possingham *et al.* 2012).

COSTS OF MANAGEMENT

Costing actions decades into the future involves a high degree of uncertainty which comes partly from uncertainty about the many drivers of change to global biodiversity and partly in the allocation of resources to its conservation. By adopting a zero future discount rate for biodiversity, as done by Stern (2006) in his seminal report on the implications of climate change for the global economy, current costs can be adopted as a proxy for future costs. We have used current costs to identify what may be needed over the next half century, using costs as indicative rather than definitive to illustrate the relative expenditure associated with different classes of actions.

Some estimates of the past and potential future costs of conservation are published. For example, the median annual cost required for conservation actions needed to downlist all 1115 IUCN Red-listed threatened birds to lower categories of extinction risk within 10 years was modelled at US\$848 000 per species per year (McCarthy *et al.* 2012). This compares with a median annual expenditure for each of 15 threatened bird species funded by the European Union during 2004–2010 of US\$560 000 (European Commission 2012), and of US\$120 000 per year for each of 111 threatened bird species in Australia during 1990–2000 (Garnett *et al.* 2003). For New Zealand, the median annual cost per taxon of successfully completing all actions needed to achieve long-term persistence of threatened and at risk species was US\$130 205 (McCarthy *et al.* 2012). Other sources of information include numerous recovery plans for threatened species. However, there are no summaries of the details of the cost estimates. For the purpose of the exercise being undertaken here, we therefore codified the costs for standard actions on the basis of unpublished estimates provided by threatened species managers for the *Action Plan for Australian Birds 2010* (Garnett *et al.* 2011).

Costs were categorised into the groupings listed above as follows:

Expand the protected area network: land purchase costs or the cost of developing covenants over private land could only be estimated very approximately in the basis of local rural real estate values in the regions where land will need to be secured for future use by taxa.

Maintain and improve habitat quality: land management was costed as the wages of one person plus on-costs (\$60K), and running costs for a field land manager at \$40K per species or per group of geographically colocated species, unless there had been a history of local

volunteer involvement that were assumed to defray costs to \$30K.

Identify, protect and expand refuges: modelling of terrestrial refuges needs to be undertaken at a local scale and was estimated as taking one technical officer (\$60K) one year. It was considered best to consider monitoring of marine refuges as an exercise in the future since it may not be necessary until declines are proven and management action is required. By that stage, any refuges may have shifted from where they are now. The costs were therefore estimated as the cost of a PhD student for three years: cost \$150K plus \$50 for tracking devices = \$200K.

Create new habitat: costs/ha of restoration have been published for different habitats (e.g. Louis Berger and Associates 1997 for saltmarsh). These were used to calculate the costs of restoration up to an area thought likely to contain a self-sustaining population of the species concerned.

Species management: many species already have active conservation programs or are likely to require them. These were mostly standardised to \$100K each as for habitat management above. Some species management is going to require research. These were generally attributed to a PhD stipend of \$150K over three years including supervision costs.

Assisted colonisation/enhance the genetics of populations: costs of assisted colonisation and of enhancing the genetics of wild populations by adding individuals from another population were estimated separately for each taxon. Some taxa have proven track records for reintroduction (e.g. Black-eared Miner *Manorina melanotis*; Clarke *et al.* 2003) so are likely to be relatively inexpensive to move to another new site, while for others it is likely to prove extremely difficult (e.g. Glossy Black-Cockatoo *Calyptorhynchus lathami*; Crowley *et al.* 1999). Given the controversy about assisted colonisation, particularly to mainland sites, costs usually include funds for a feasibility study to ensure that the receiving community supports the notion (Burbidge *et al.* 2011). These cover the cost of a social scientist plus the costs of mailouts to members of the receiving community and other interested parties.

Captive breeding: to estimate captive breeding costs we sought advice from managers of three captive breeding facilities (Alice Springs Desert Park, Healesville Sanctuary, Priam Australia Pty Ltd) on the costs of establishing and maintaining a facility with 200 of a taxon while maintaining a stud book to ensure genetic variability. On the basis of this we estimated the following costs:

Aviary building costs: \$20 000 per breeding unit × 100 units = \$2 000 000 = \$40 000 p.a. for 50 years
Food/expenses: \$30 000 p.a.

Veterinary: \$10 000 p.a.

Staff × 2 \$140 000 p.a. (including on-costs)

TOTAL \$220 000 p.a.

Monitoring: the costs of monitoring vary greatly, and must include not just the costs of accessing sites and counting birds but also the costs of data management, analysis and reporting. The estimate for any monitoring program, whether for a single taxon or for multiple taxa from a geographically defined area that could readily be monitored

simultaneously, was therefore standardised at \$30K, unless there were reasons to vary it because there has historically been a high level involvement from volunteers or because there were other reasons to think that the monitoring would cost substantially more or less. The Atlas of Australian Birds, however, had a single cost estimate of \$200K/year to cover coordination of records of multiple taxa.

TIMING

The timing of actions is critical to costs when these are averaged over a 50 year period. Some actions must begin immediately; others can be delayed until there is greater certainty about trends. Many of the factors identified by McDonald-Madden *et al.* (2011) for timing of decisions about assisted colonisation apply to all aspects of climate change adaptation response. For the analysis here we have standardised timing for most taxa as follows:

Immediate defined actions

Land purchase: for some taxa where there is high certainty about a need to acquire habitat for conservation purposes at a particular location (e.g. coastal locations where inundation is inevitable), land needs to be acquired now before it is alienated for other purposes even if it is not used for the purpose for which it is acquired until a later date.

Land and species management: many aspects of land management need to be undertaken immediately regardless of climate change or taxa will not persist long enough for climate change to affect them. Thus management of existing threats (e.g. Garnett *et al.* 2011) can be seen as an aspect of climate change adaptation.

Surveys: surveys, which are separate from monitoring, are needed for some little known taxa to create a baseline for subsequent monitoring and action. Surveys required will either involve estimates of known populations or the location of new ones.

Refugia modelling: modelling to identify refugia at a fine scale is becoming increasingly sophisticated. The earlier refugia are identified, the sooner their requirements can be incorporated into land-use planning. **Assisted colonisation:** for a few species, assisted colonisation is already occurring as part of conservation programs (e.g. Eastern Bristlebird *Dasyornis brachypterus*; Baker 2009). Given this is occurring, there is an opportunity to consider immediate adaptation of the assisted colonisation process to incorporate climate change predictions.

Ongoing actions

Many actions must continue indefinitely, so are costed over the full 50 year period under consideration. These should either start immediately or be continued from existing programs.

Monitoring: the sooner monitoring begins the sooner it will be possible to determine when or if action must be taken. Many bird taxa may not need future action so decisions are contingent on there being a solid monitoring baseline that shows trends through a range of

recent climatic perturbations so that climate change signals can be detected from inter-annual variation and other noise.

Captive breeding: for three species, the Orange-bellied Parrot, Helmeted Honeyeater *Lichenostomus melanops cassidix* and Regent Honeyeater *Anthochaera phrygia*, insurance populations already exist and are already being employed as part of reintroduction programs. Still others need to have insurance populations created immediately before they go extinct regardless of climate change (e.g. Western Ground Parrot *Pezoporus flaviventris*). While it is the intention that such captive populations be maintained at their current level of intensity for only a limited period of time before wild self-sustaining populations are established, current climate-change trends and the levels of exposure and sensitivity of many of the taxa already in, or being considered for, captive breeding, makes such ambitions seem optimistic.

Species management: many threatened taxa already have recovery programs which need to be continued, and probably intensified, as climate changes. For other taxa, management needs to start immediately (Garnett *et al.* 2011) and probably continue indefinitely if they are to persist.

Future defined actions

Assisted colonisation and genetic augmentation: for only a few taxa is it recommended that assisted colonisation be undertaken immediately. For most it is considered a distant event and the only action postulated is research on its feasibility, including its social acceptability among the receiving community (Burbidge *et al.* 2011), with the actual costs being deferred until after 50 years have elapsed.

Establish new habitat: for a few coastal taxa, new habitat will inevitably be needed. However, not only does this not need to happen now but the conditions to establish it have not yet arrived because the sea level has not yet risen sufficiently. It is also possibly that, as sea level rises, saltmarsh will occupy areas vacated by less salt-tolerant species so the expense of artificial creation of new habitat will be much reduced.

Marine refugia modelling: unlike terrestrial refugia, marine refugia are likely to shift location as marine features that result in patches of high productivity (upwellings, thermoclines etc.) change location. This means that any identification of locations where interventions might occur, such as reducing fishing catch rates, needs to be based on current information with the timing informed by monitoring. In this analysis, we have assumed that such information can only be obtained through detailed research, including tracking of individuals, which would be undertaken by PhD students at a time in the future to be determined by monitoring results.

Future ongoing actions

Some actions that will need to continue indefinitely do not need to start immediately. The actual time when they will be needed can, at this stage, only be guessed. However, it needs to be flagged that action is likely to be necessary at some point in the future based on current trends, so a standardised arbitrary date has been selected for each that can be

refined with greater knowledge.

Captive breeding: while it might be possible to say at what point the climate space of a taxon is predicted to disappear completely from a location, this depends greatly on the model employed, with models for some taxa producing very different outcomes. In any case, there is a strong probability that lag effects will allow a taxon to persist for longer at a location than its climate space. On the other hand, if it appears inevitable that a captive population will be needed, it is sensible to establish one while there is substantial genetic diversity because, once the population is gone, there will be no opportunity to augment it from the wild. For treatment here we have assumed that, for taxa where captive breeding is likely to be needed, it will not be needed for the next forty years. In terms of costing we have therefore calculated the cost for only ten years, averaging that across the full 50 years period to provide an idea of the average annual cost for the full period.

Refugia management: apart from sites that need to be purchased it is unlikely that, once refugia have been identified, that they will immediately require management that explicitly considers climate change. Here we have estimated that refugia management will be needed in 25 years' time and will then continue indefinitely.

Species management: some species management is not needed now but may be in future. This includes such actions as providing supplementary food at sites where natural food supplies may be reduced by climate change, management of coastal nests that may face unsustainable rates of inundation, and irrigation of food trees that may be affected by drought. Here we have assumed such actions will not be needed for 25 years.

CONCLUSION

This review of the management actions and costs of adapting to climate change demonstrates two truths. One is that options are limited. While action can be taken, the level of management intensity will have to increase across many more taxa than it does now, and many actions require a radical rethinking of what constitutes normal practice, and indeed of what constitutes normal objectives of conservation management. The other is that such management is likely to require a substantial increase in funding from the current modest levels if we aim to retain all biodiversity and fulfil the objectives set by society in legislation. Almost certainly there will need to be prioritisation (Wilson *et al.* 2006; Joseph *et al.* 2008), but as yet there are no rules beyond financial efficiency as to which taxa are considered to have low priority and can be officially abandoned to climate change. In lieu of global action that will reduce greenhouse gas emissions and reduce the rate of climate change, such rules need to be developed soon or effort will be misdirected as the effort required is overwhelmed by the size of the extinction crisis bearing down on us.

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