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G.P. Von Maltitz, R.J. Scholes, B. Erasmus and A. Letsoalo
CSIR, South Africa

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Direct correspondence to:
Graham von Maltitz, GvMalt@csir.co.za

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Adapting Conservation Strategies to Accommodate Impacts of Climate Change in Southern Africa[1]

G. P. Von Maltitz, R. J. Scholes, B. Erasmus, and A. Letsoalo

1. Introduction

Current predictions are that global climate change will have substantial impacts on southern Africa's biodiversity, including wide-scale extinctions over the next 50 years (Rutherford et al., 1999; Hannah et al., 2000*a*,*b*; Gitay et al., 2001, 2002 et al.; Midgley et al., 2002*a*,*b*; MA, 2005). At a global scale, Thomas et al. (2004) predicted that 15–37% of species in their sample (that covered 20% of the earth surface) may be at risk of premature extinction due to anthropogenically caused global change by 2050. The Millennium Ecosystem Assessment, using different models and assumptions based largely on habitat loss, reached similar conclusions (MA, 2005). Within South Africa, one of the few areas in sub-Sahara Africa where detailed analysis has been conducted, the predictions are that most of the current biomes will reduce in size and will be shifted to the east of the country. Up to half of the country will have a climatic regime that is not currently found in the country (Rutherford et al., 1999). The succulent karoo biome, (a succulent-dominated semi-desert located on the southwestern coast of southern Africa) is projected to be the most severely impacted, with the grassland and *fynbos* (a Mediterranean-climate sclerophyllous thicket that approximates to the Cape Floristic

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region) biomes also likely to suffer from high climate change impacts (Rutherford et al., 1999; Midgley et al., 2002*a*,*b*). *Fynbos* and succulent karoo are biodiversity hotspots of international importance (Myers et al., 2000), with the latter being one of only two globally important arid-climate biodiversity hotspots.

Two main aspects of the climate have widespread influence on animal and plant species distributions: temperature and water balance (a combination of precipitation and evaporation, which, in turn, is directly influenced by temperature) (Cubasch, 2001). The dynamics of plant and animal populations change at the edge of individual species' distribution, as net mortality becomes larger than net fecundity, with a spatial gradient of declining population numbers as a result. In a scenario of climate change, this will lead to the progressive extinction of nonvagile species in their natural range, beginning with population dieback in the so-called "trailing edge" of the historical distribution range (Davis and Shaw, 2001). This change in local population dynamics is affected directly by temperature and water balance, but it is also indirectly through aspects such as interspecies competition, fire frequency, pollinator distribution, herbivory and predation, food availability, soil type, topography etc. (e.g., Gaston, 2003). Few species occupy their "fundamental niche,", i.e., the range determined by the physiological tolerance limits. Their actual range, the "realised niche," is a subset of this resulting from the outcome of interactions with other species. The degree to which species distribution can be predicted based on their climatically defined habitat niche differs between species (e.g., Thuiller et al., 2006).

Increased mean temperatures during the coming decades are predicted for the majority of locations by all global circulation models (Cubasch et al., 2001). Within southern Africa, the inland areas are expected to experience the greatest increases in temperature (2–4°C this century), whilst the coastal areas are predicted to experience somewhat lesser increases (1–3°C), due to the thermal buffering effect of the oceans (Cubasch et al., 2001; Scholes and Biggs, 2004).

Changes to precipitation are more difficult to predict, and there is less agreement between models. Despite predictions for increased global precipitation, within southern Africa the majority of models predict that the western two-thirds of the continent south of 15ºS will have ~10% reductions in annual precipitation during the 21st century, while the eastern one-third may see an increase of the same order (Scholes and Biggs, 2004; Hewitson and Crane, 2005). A combination of increased temperature (and thus increased evaporative demand) with decreased rainfall will increase the aridity of affected environments, notwithstanding the slight offsetting beneficial effect of elevated CO_2 on plant water use efficiency (Scholes and Biggs, 2004). A combined increase in rainfall and temperature will increase primary plant production, but will still be detrimental to specific species (Gitay et al., 2001; Gelbard, 2003).

The current rate of climatic change far exceeds any climatic change records from the past and is likely to be too rapid for evolutionary adaptation in most species (Malcolm and Markham, 2000; MA, 2005). Excluding evolutionary adaptations, species can be classified into four functional groups based on their response to climate change as follows.

1. *Persisters*: These species have the climate tolerance for the new climate of their current location.

2. *Obligatory dispersers*: These species will have to physically move with the changing climate to track areas with suitable climates (autonomous dispersers), or alternately will have to be moved artificially to new areas with suitable climates if they are unable to move on their own (facilitated dispersers).
3. *Range expander*: These species may expand into new climatic envelopes that are not currently available, but to which the species are already well adapted.
4. *No hopers*: If the species cannot do one of the above then they will become prematurely extinct, although they may persist under unsuitable climates for some time.

Some species will experience range shifts that will result in them persisting partially in their previous range whilst dispersing into new areas. We have referred to these as partial dispersers. The time span involved and the intensity of the climate change experienced (or modelled) will determine to what extent species persist or are obliged to disperse.

Detailed modeling on the impacts of climatic change on individual species has been conducted in the *fynbos* and succulent karoo regions. The AIACC project studied the Proteaceae as a surrogate for the *fynbos* vegetation to understand how individual species would respond to changing climate over the next 50 years and used this to better understand future conservation strategies. The model predicted that 57% were persisters, 26% partial dispersers, 6% obligatory dispersers, and 11% were no hopers (Williams et al., 2005). In the karoo region, it was found that the Riverine Rabbit (*Bunolagus monticularis*) is likely to go extinct because of its specialized food and habitat requirements, whilst the tortoise (*Homopus signatus),* which is less selective, is unlikely to go extinct from climatic causes in the 50-year study timeframe (G. O. Hughes, personal communication, 2005).

The current anthropogenic induced climate change is largely being driven by rising CO_2. This increase in CO_2 will enhance plant growth up to a point and may increase the relative competitiveness of C3 plants over C4 plants. This fertilization effect starts to saturate in natural ecosystems at around 500 ppm (Scholes et al., 1999). The combined impacts of climatic change and CO_2 effects have been modeled in the AIACC project for the lowveld savanna regions of South Africa (R. J. Scholes, personal communication, 2005). Preliminary model runs suggest that the decrease in soil moisture and the increase in temperature overwhelm the small elevated CO_2 advantage that trees have, given that C3 and C4 plants respond differently to these factors. This study lets us consider the impacts of climate change on functional aspects of habitats rather than individual species. On the basis of this model, it is predicted that in the lowveld savannas of South Africa, the structural and functional habitat suitability for browsers and grazers is likely to remain relatively constant in the 50-year timeframe, provided that fire and elephant management are appropriate. Overall, the carrying capacity for large herbivores is projected to decrease by about 10%. The key control on future habitat structure in this example is the size of the elephant population and its interaction with the fire regime. Although this study does not consider individual species, it suggests that the functional integrity of the savanna habitat can be maintained near to current conditions through appropriate management.

2. A Brief History of Conservation in Southern Africa

The countries of southern Africa have extensive tracts of land that are managed as conservation areas (Table 1). The extent of conservation differs between individual countries. Approximately half of the countries in the region exceed the International Union for the Conservation of Nature (IUCN) guidelines of 10% of land area under formal conservation. Over the entire region, approximately 10% of land is conserved in IUCN categories I–V reserves (these categories are reserves set up strictly for conservation) with a further 8% conserved in areas managed for sustainable use, i.e., IUCN category VI areas. Some countries fall far short of the IUCN guidelines; for example, in the case of Lesotho only 0.2% of the surface area is conserved (Scholes and Biggs, 2004; WDPA, 2005).

Even where counties have a relatively high level of land conserved, the fraction of *biodiversity* conserved may be substantially less (Rodrigues et al., 2004; Orme et al., 2005). This is because the history of conservation has not been based on strategic conservation objectives but rather on the availability of land and in many instances the presence of big game species (Pressey et al., 1993; Heywood and Iriondo, 2003). The large reserves are therefore mostly situated in inhospitable areas, including arid areas, mountainous areas, areas with historically high levels of diseases, such as sleeping sickness and malaria, and areas with low agricultural potential, such as arid and semi-arid regions. Of the 52 unique ecoregions identified in Southern Africa (Olsen et al., 2001), 23% of ecoregions (15% of land area) have less than 3% conservation (Table 2). Forty percent of ecoregions representing 35% of the land area have less than 5% formally conserved in IUCN reserves. Southern Africa has an exceptionally high biodiversity, including a number of centers of endemism and three biodiversity hotspots (Myers et al., 2003). The Madagascar hotspot has only 2.9% of the area conserved in IUCN reserves with a further 1% conserved outside of IUCN reserves. The succulent karoo hotspot has only 1% conserved, although there are proposals to conserve an additional 19%. The Cape floristic region is well conserved in the mountainous areas but poorly conserved on the flats (see Table 3). By comparison, the *mopane* savanna regions (not a biodiversity hotspot) are well preserved, largely due to their low economic value for agriculture (see Table 3).

Formal conservation started in the late 19th century. From about 1910 to 1970, there was a steady expansion of protected areas (Figure 1). There tended to be two parallel paths of conservation, one leading to the formation of forest reserves, managed for sustainable wood extraction and/or catchment protection. The other led to the establishment of game and nature reserves, which originally tended to be centered in areas with high wildlife populations and had their history as hunting areas. These are currently managed for biodiversity conservation and ecotourism (von Maltitz and Shackleton, 2004). During this period, reserves had strong state support and were relatively well resourced with public funds. Strong policing maintained the reserves, and real or virtual fences excluded the surrounding population from the reserves.

The postcolonial period has seen a shift in government focus to social development issues. Protected areas have, in most instances, been maintained, but budgets have diminished. As a result of human population growth, there is now a high pressure on the borders of most reserves and conflict over resources is increasing. The ability to police reserves has decreased due to budget cuts. In a few cases, local communities have invaded the reserves and settled on them (Fabricius et al., 2004; von Maltitz and Shackleton, 2004; Child, 2004).

A trend since the 1980s has been toward sharing the management and benefits (both financial and natural) derived from protected areas with communities local to the reserves. This is a pragmatic approach resulting from a growing negative perception regarding conservation areas, and a decline in national budgets to maintain the integrity of conservation areas. This has been linked to change in government policies regarding resource ownership, with a trend to delegating ownership of wildlife and forestry resources from the state to those owning or resident on the land. This makes it possible for communities on communal land to enter into Community-Based Natural Resource Management (CBNRM) programs (Fabricius et al., 2004; Child 2004; Hutton et al., 2005), and the establishment of private wildlife ranches on commercial land (ABSA 2003). A number of studies have shown that the economic returns from wildlife can exceed the returns from cultivation and cattle ranching, particularly in agriculturally marginal areas (Child, 1988; Bond et al., 2004; Balmford et al., 2001, Naidoo and Adamowicz 2006). This change promotes biodiversity conservation in the communal and private areas as there is an economic incentive for conservation (Fabricius, 2004; Child, 2004; ABSA, 2003).

The successes that had been hoped for through community involvement and resource sharing in conservation areas have been less than expected, largely due to a lack of appropriate capacity, both in government departments and in communities (Hutton et al., 2005). As a consequence, the rationale for comanagement and resource sharing from conservation areas has come under increased criticism, with mounting support in some sections for a reversal to more conventional conservation approaches (Wilshusen et al., 2002; Hutton et al., 2005; Büscher, 2005).

A recent trend is toward international assistance for conservation in Africa, and the Global Environmental Facility (GEF) fund of the United Nations, as well as bilateral funds from first world countries have contributed millions of dollars in this regard. For the first time in decades, new areas are being proposed for conservation, and existing conservation is being strengthened. The introduction of strategic conservation planning tools such as Worldmap (http://www.nhm.ac.uk/science/projects/worldmap/index.html) and C-plan are making it possible to plan the location of reserves in a scientific and defensible manner to achieve agreed conservation targets (Pressey et al., 1993, Margules and Pressey, 2000; Pressey and Cowling, 2001). This ability is being exploited in the *fynbos*, thicket and succulent karoo regions of South Africa (Cowling and Pressey, 2003). The concept of transnational megaparks (sometimes referred to as "Peace Parks") has also become popular, with a number of new parks being developed such as the Limpopo, Kalagadi, and Maluti-Drakensberg Transfrontier Parks (van der Linde et al., 2001). The possible consequences of climate change to biodiversity are beginning to be considered (Hannah et al., 2002*a*,*b*; Midgely et al., 2003; Williams et al., 2005).

3. An Overview of Adaptation Options for Biodiversity Conservation in a Climatically Changing Environment

Conservation becomes a moving target in a climatically changing environment, and although current reserve systems are a starting point, there is no clear end point.

Biodiversity patterns in 50 years time represent only one period in an environment that is likely to go on increasing in temperature for at least 200 years because of the residual effect of CO_2 increases (Cubasch, 2001). At some point in the future, once CO_2 emissions have been reduced, there is likely to be a period of global cooling that will hopefully bring climatic conditions back to historical levels, but the time span for this is hundreds of years and hence exceeds most conservation-planning horizons.

The following potential adaptation options were identified as adaptations to prevent extinction of biodiversity given the predicted climate change:

- Do nothing (i.e., maintain the current conservation strategy);
- Reconfiguration of reserve system to strategically conserve areas that accommodate climate change;
- Matrix management, i.e., managing the biodiversity in areas outside of reserves.
- Translocation of species into new habitats; and
- *Ex-situ* conservation. This could include gene banking, cryopreservation, zoos, and botanical gardens.

Current understanding of how ecosystems will respond to climate change, based both on historical data and modeled predictions, suggests that individual species will respond at different rates. The consequence of this is that entire ecosystems will not move in unison, but species will move independently, leading to altered community composition (Huntley, 1991; Graham, 1992; Gitay et al., 2001; Williams, 2005; Thuiller et al., 2006; Bush, 2002). It is important that conservation strategies consider individual species when attempting to minimize losses. This does not negate the need to maintain habitats (ecosystems), but it needs to be accepted that the compositional structure of these systems will be different in the future, though in some instances the functional attributes may be similar (see lowveld savanna case study above).

On the basis of individual species responses to climate change, a set of adaptation options are identified in Figure 2 and their relative constraints and benefits are compared in Table 4.

3.1 Conservation of species that persist or expand their range

Where a species persists in large populations in an already-conserved area under future climates, there is no strong basis for concern. However if the species becomes invasive and its range expands then it may become a threat to other species and may need control. If the species is already threatened under current conditions, even if it persists, it might warrant extra conservation attention, especially if it is not currently found in existing conservation areas.

3.2 Conservation of obligatory dispersers

For obligatory dispersers, there are basically two scenarios, autonomous dispersal, where the species can reach a new habitat through natural dispersal mechanisms, at a rate sufficient to keep up with the shifting climate. If natural dispersal is inadequate for the species to reach a new habitat, then humans can facilitate dispersal through translocation of species to new suitable habitats. In the first instance, a climatically and environmentally suitable pathway must exist to allow the species to move through the landscape to track the changing climate. The time-slice methodology of Williams et al.

(2005) provides a way of identifying key areas that need conservation to ensure that autonomous obligatory dispersers are able to disperse and identifying species that will require facilitated dispersal.

For autonomous obligatory dispersers, the key question is whether there are suitable migratory pathways that will allow species to move from their current location to a future protected area. The extent of land transformation in dispersal corridors is a major concern (Hannah et al., 2002*a*). There are two options for protecting migratory pathways: expand the existing reserve network; or ensure that the matrix (i.e., those areas outside of formal reserves) is sufficiently protected by measures that do not require state ownership and exclusive use of the landscape for conservation objectives.

For facilitated obligatory dispersers, the only option for maintaining wild populations is to physically move the species to the new suitable habitat (Hossell et al., 2003). Movement of large mammals and birds is a well-established practice in conservation circles, regularly undertaken throughout the subregion. However, it is usually undertaken to reintroduce species to locations where they are believed to have occurred historically, or to increase genetic exchange. Introduction of species to places where they probably did not exist within the recorded past is frowned upon. To conservators, pre-emptive facilitated movement, of plants and invertebrates is a new concept. Facilitated dispersal will have ethical and practical considerations such as:

- What is the number of individual organisms per species that need to be moved to establish a new viable population, and how individuals for translocation should be selected (Heywood and Iriondo, 2003).

- Under what circumstances should a species be moved to an area where it did not historically exist, and what impact will this have on the species currently occurring in that area (or which will occur there naturally as a consequence of climate change) (Sakai et al., 2001; Hossell et al., 2003; Radosevich et al., 2003)?
- Which species need to be moved together, in order to preserve the community structure?
- How is the pattern of genetic variability within the population to be maintained?

3.3 Conservation of "no hopers"

For the no hopers, the only nonfatalistic option is to maintain the biodiversity in artificial situations such as zoos, botanical gardens, seed banks, and through cryopreservation, in the hope of perhaps introducing them to the wild at some distant future time. Such *ex situ* conservation practices are also a wise "insurance policy" for species with some hope of surviving in the wild.

3.4 The threat of invasive species

Some persisters, autonomous dispersers, and facilitated dispersers, are likely to become "weeds,",i.e., overabundant in their new habitats, to the detriment of other species (McDonald, 1994). We will need to reconsider the concept of invader species given climatic change. The most likely candidates to invade are primary succession species that are well adapted to dispersal into new habitats. Weed outbreaks will be further encouraged by the disruption of communities in the receiving environment, directly or indirectly due to climate change, and by the possibility that the invasive species will travel faster than their natural competitors and controlling agents (Malcolm and Markham, 2000). Range expansion is a potential threat to the species currently established in the new areas, and may be an indirect factor that prevents the species persisting in that habitat (even if it can persist from a climatic perspective). A further concern is that climate change may well favor introduced exotic species, increasing the chance that they become invasive. Aggressive control of invasive species may therefore be needed even more than at present.

3.5 Interventions to facilitate biotic adaptation

From Figure 4, it is clear that no hopers and facilitated obligatory dispersers require direct human intervention to prevent extinction. For the remaining species, extinction can be prevented through ensuring that key areas of the distribution are conserved both now and in the future and that the migratory pathways necessary for the species to move between protected areas remain permeable to the species concerned. For autonomous obligate dispersers, the same conservation objective can be achieved through two different approaches, either by ensuring conservation outside of protected areas (matrix management) or through reconfiguring or expanding the conservation area. In practice, it is a strategic combination of both these methods, rather than either on its own, that is likely to give the best results.

4 Economic Considerations Relating to Adaptation Options

The *fynbos* biome, and particularly the conservation of members of the Proteaceae, was used as a case study to investigate the costs and benefit of the various adaptation options discussed above. A modeling process was used to identify the areas critical for conserving migratory pathways, as well as identifying disjunct habitats and no-hoper species (Williams et al., 2005).

Reserve expansion was found to be a very expensive option if it is used as the only mechanism of protection. Reserve costs can be broken into the costs of land acquisition and the ongoing annual cost of land management. Operational costs per unit area decrease substantially as reserve size increases. On the basis of South African National Park data, a 1-km square park has a U.S.$104,793 annual operational cost, while a 100,000 km^2 park only costs U.S.$66/$km^2$ (Martin, 2003). The land management cost per hectare decreases nonlinearly as the reserve size increases, so from a cost-efficiency perspective, it is better to have a few large reserves rather than numerous small reserves (Frazee et al., 2003; Balmford et al., 2003).

Contractual reserves (on private land) would appear to be more cost-effective than the state purchasing the land and forming state reserves in most circumstances (Pence et al., 2003; A. Letsoalo, personal communication, 2005). In essence, the cost of managing the land for conservation is the opportunity cost of lost income to the farmer for not using the land for the most profitable alternative land use activity. This cost will vary greatly. It will be very low for extensive rangelands, low for dryland grain production, but high for irrigated crops and speciality crops such as horticulture. It is only for land used for high-value crops where it is more economical to establish a formal reserve, rather than a contractual reserve with the current land owner (A. Letsoalo, personal communication, 2005). In many instances, rangeland management is already biodiversity-friendly to many species, and to achieve the conservation objective may require little or no increased cost to the rancher. Reduced stocking rates or minor changes in management practices (for instance, withholding grazing during a critical period) may be sufficient to achieve the desired results. Where dryland cropping is involved, a spatially explicit strategic approach would be needed to ensure that viable biodiversity corridors are achieved.

Costs for facilitated translocation cannot be compared directly with autonomous translocations, as the approach is only likely to be used where autonomous dispersal is not an option. The cost is dependent on the number of organisms translocated and the establishment costs involved. Simultaneous translocation of communities of mutually interdependent organisms may have to be considered, including pollinators and seed dispersers in the case of plants.

Gene-banking and other ex situ conservation will not achieve the same level of biodiversity conservation as is achieved through in situ conservation, but remains a fall-back position when other opportunities are not available, as well as an insurance measure when they are. A common target for in situ conservation is to conserve at least 10% of the historical population. Ex situ conservation will only conserve a small number of organisms for each species; therefore, good representation of the genetic variation in the population is essential. Gene fingerprinting to ensure that the collection represents the broader population is therefore a significant cost consideration. The costs of ex situ conservation cannot be directly compared to the costs of in situ conservation, as they do not achieve the same end points. Table 4 compares the relative economic advantages and disadvantages of the different conservation strategies.

5. Adaptation Options To Allow Species Movement in Response to Climate Change

5.1 Considerations for migratory corridors

Movement in response to climate change is unlikely to be a single dispersal event by a group of individuals or species across an entire landscape. In general, the movement of species will be poleward or to higher altitudes as a response to global warming, but it will also be affected at the local level by changes in precipitation and microclimatic influences (Gitay et al., 2001). Species are expected to respond individually, and gradually, per generation. This raises the bar substantially for any parcel of land to qualify as a corridor, as resources need to be sufficient to sustain a life cycle, not just an individual passing through (Simberloff et al., 1992). Halpin (1997) presents a summary of management options for protected areas in the face of climate change. With regard to corridors as a means for species to escape climate change effects, Halpin (1997) mentions buffer zones and connectivity but reiterates the need for firm ecological evidence upon which to base corridor and buffer zone design. In a more recent review on management options for forests in the face of climate change, Noss (2001) identifies similar priorities.

Convincing ecological evidence upon which to base a particular corridor system will only be available if explicit studies on habitat use and habitat preference of a large number of species in any particular ecosystem are collated. A key development in this field is the spatially explicit nature of habitat use. However, for effective corridor design, we need to understand fluxes of organisms and matter in the landscape in a spatially explicit manner. The intuitive ecological advantages of wildlife corridors suffer from a lack of empirical supporting evidence (Saunders et al., 1991; Simberloff et al., 1992). An often-stated example of the usefulness of corridors is riparian vegetation. Simberloff et al. (1992) states that riparian vegetation does not constitute a typical corridor from a management point of view, as it is a unique habitat in itself that happens to be linear, and it does not connect discrete patches of like habitat. Birds are less constrained to continuous corridors, but they still need to access resources, even if patchily distributed. Connectivity and corridor design in a landscape with varying habitat suitability depends on a definition of what is considered as habitat for a particular species. Any analysis has to account for a large number of species, or groups of species, and the variables that influence the habitat selection of each of them. An alternative approach is to use *processes* in landscapes as spatial planning units, and design reserves and corridors to maintain local and regional processes. An excellent example of using such processes in conservation planning is that of Rouget et al. (2003*a*,*b*). The effect of this approach is to manage the landscape for heterogeneity. The assumption is that if the processes thought to be responsible for the observed heterogeneity are preserved, then heterogeneity will be maintained in the face of climate change. The limitation that we face is that, apart from knowledge of previous disturbance events, we do not know the measure of the heterogeneity that has to be maintained. This level of heterogeneity has been termed functional heterogeneity in the context of savanna herbivore assemblages (Owen-Smith, 2004).

5.2 Reconfiguring the reserve network

Formal conservation areas remain a critical component for biodiversity conservation in a changing environment (Dudley and Stolton, 2003). This benefit can be enhanced by ensuring that reserves are well configured to best conserve biodiversity, given the impacts of climate change. The conservation of potential refugia, environmental gradients and likely migratory corridors are adaptations to the current reserve network that will increase their effectiveness in relation to climate change. Systematic conservation planning has come of age in providing land-parsimonious algorithms to prioritize new areas quantitatively for addition to the existing reserve network (Pressey and Taffs, 2001; Pressey et al., 2000, 2001; Reyers, 2004; Rodriguez et al., 2004). The inclusion of a climate change component is, however, still in its infancy (Cowling et al., 2003; Hannah et al., 2002*a*, *b*; Williams et al., 2005). In many situations, current reserve networks are poorly planned to conserve current biodiversity patterns, let alone the additional requirements required as a consequence of climate change.

In a first for southern Africa, Williams et al. (2005) developed a method based on time-slice analysis of potential climate change-induced species migrations to understand how best to locate conservation areas in the *fynbos* biome. For the study area considered, a 50-year time frame and the limited taxa investigated (the Proteacea), they recommend an approximate doubling of the current reserve network to achieve the required level of conservation, although some of this reflects the inadequacy of current reserve networks to conserve current biodiversity in addition to the needs of a changing environment. The study acknowledges a number of limitations and assumptions, but it still provides a powerful tool for objectively considering impacts of climate change on reserve planning.

5.3 Managing areas outside of reserves (the matrix)

There are a number of practical and ecological reasons why matrix area must be a major part of a biodiversity conservation strategy, especially when considering the impacts of climate change (Hannah et al., 2002*a*; Gitay et al., 2001; Rodriguez et al., 2004; MA, 2006). Managing the matrix should be a complementary activity to formal conservation, rather than an alternative, though there is also the option for formalizing contractual conservation arrangements with landowners outside of formal reserves through the creation of contractual reserves (Pence et al., 2003). In South Africa, changes to legislation make it possible for the state to enter into a contractual arrangement with landowners to ensure conservation (Pence et al., 2003). This is potentially cheaper than outright purchase of land and may be a more acceptable option to current land owners. Economic incentives also lead to conservation on private land. Already many land owners are using their land for nonagricultural activities such as ecotourism and wildlife ranching because it provides better returns.

Even if conservation targets are being met, the area of the matrix is generally an order of magnitude larger than the area under conservation for most habitat types. It is clear that areas outside formal reserves generally contain a significant portion of the biodiversity, often close to an order of magnitude more than in the reserves (Rodrigues et al., 1999). For instance, R. Biggs, B. Rayers, and R. J. Scholes (unpublished data) estimated that 80% of South Africa’s biodiversity is outside of formally protected areas, this despite the high levels of degradation and land transformation. As such, nonreserve areas play a pivotal role together with the protected area network, to adequately conserve our biodiversity (Hannah et al., 2002*a*). The recommendation that 10% of the land area be protected (IUCN, 1993) was intended as a general rule of thumb and implicitly assumes that the protected area is representative. However, it has been shown for a savanna landscape example that this guideline may only represent 60% of species in an area and exclude up to 65% of rare and endangered species (Reyers et al., 2002). Up to 50% of the land area may be needed to preserve a representative portion of species (Soule and Sanjayan, 1998).

Although South Africa has only 5.4% of its land area under state conservation, it is estimated that an additional 13% is currently managed as private wildlife ranches (Bond et al., 2004, updated from Cumming, 1999). To a large extent, the growth of the game ranching industry has been a consequence of changes in legislation that has allowed private ownership of wildlife, something that historically had not been permitted. Not all game-ranching practices automatically result in improved biodiversity conservation, but it is argued that on balance, greater biodiversity benefits are achieved through this land use versus alternative agricultural practices (Taylor, 1974; Child, 1988; Bond et al., 2004). Although market forces and enabling legislation have switched land use in the drier areas to conservation, it is the higher-rainfall areas, and especially those suited to crop agriculture or forestry, where biodiversity is most threatened. In these areas, greater direct intervention may be needed to maintain biodiversity and migratory corridors.

This distinction between conserved areas and the matrix creates the impression that there are distinct structural boundaries and hard edges between reserves and the matrix. Although this is sometimes the case, such edges are more often differentially permeable to water, matter, species, and energy fluxes, and instead of quantifying the biological effects of a fragmented landscape (Saunders et al., 1991), we should consider a dynamic landscape with patch edges that act as species- and flux-specific filters at multiple scales. The process of forming such a landscape has been termed habitat variegation (McIntyre and Barrett, 1992), and it echoes the sentiments of Murphy and Lovett-Doust (2004) that a binary approach of suitable habitat versus the matrix is not a true reflection of landscape dynamics. These spatial linkages of energy, matter, and species fluxes across edges provide additional support for biodiversity-friendly, matrix management as part of formal reserve management.

The management of the matrix becomes even more crucial when considering the likely impacts of climate change. Biodiversity responses to climate change may take a variety of forms, and our current ability to predict this is limited due to uncertainties in both the climate scenarios and in how species will react to the change (reviewed by Walther et al., 2002; McCarty, 2001; Hughes, 2000; Parmesan and Yohe, 2003; Root et al., 2003). Matrix management practices need to anticipate an increased movement of species through the landscape, and therefore, connectivity between suitable habitat patches is important. This connectivity may translate into buffer zones around existing suitable patches or linear corridor features that link suitable patches. The effects of habitat fragmentation have been reviewed elsewhere (Saunders et al., 1991); for this paper, we take fragmented landscapes, as a given and important component for consideration in conservation planning.

The final hurdle in managing the matrix for species movement as a response to climate change is the implementation phase. An integrated procedure for determining land use is needed, and this procedure should recognize the need for robust ecological evidence, and provide opportunity to gather such evidence. Buy-in from local stakeholders is critical since the decision to use or not use any piece of land will affect individuals.

The use of matrix management as an adaptation strategy to enhancing resilience to climate change can be implemented in two ways, and both are potentially needed to achieve an effective mitigation strategy (adapted from Frazee et al., 2003)

1. Strategic conservation of critically important areas of the matrix. This would be areas that are identified as having a strategic importance for conservation, but that cannot be included into the formal conservation network for financial or other reasons. In these circumstances, the state can enter into a contractual agreement with the landowner that the land be managed for conservation purposes. The opportunity cost of not undertaking the next best agricultural practice would be a realistic way of calculating the compensation that the farmer would need (see economics section above).
2. General enhancements to biodiversity conservation on all nonreserve land. In this instance, less costly incentives could be used to promote more biodiversity-friendly farming practices. This would include incentives as discussed below for commercial land or the establishment of CBNRM in the communal areas.

5.4 Policy mechanisms for facilitating biodiversity conservation within the matrix

Matrix management is about seeking compromises, as well as ways of achieving them, which allow sustained economic benefits, but also the persistence of biodiversity. It may involve, for instance, the setting aside of riparian strips or woodland corridors, reducing the use of pesticides and fertilizers, reducing animal stocking rates, or reintroducing necessary disturbances such as fire. The wrong mix of land uses in the matrix can be inimical to conservation, for instance by increasing alien plant invasion, or by causing a retreating forest edge (Gascon et al., 1999). National policy frameworks need to promote or enforce strategic matrix management.

As a result of poorly developed markets for ecological services, there is minimal incentive for landowners to promote biodiversity or maintain migratory corridors essential to mitigate against biodiversity loss as a consequence of climate change. That many landowners do so of their own accord must be attributable to the strong land stewardship ethic often found among those who live close to the land. Perverse policies, such as state ownership of wildlife or excessively onerous burdens associated with protecting threatened species, may even result in land owners deliberately reducing biodiversity on their land.

We see land tenure as an important consideration when developing matrix management interventions, with a different set of incentives and approaches being applicable for private land versus communal land.

5.4.1 Incentives for matrix management on privately held land

Shogren et al. (2003) and Doremus (2003) suggest the following policy and economic incentive systems for promoting biodiversity on private land.

- Education. Many land owners have a conservation ethic, and provided that cost implications are low, may well change their land management practices to meet biodiversity conservation needs once they understand the pertinent issues.

- Direct incentives. These can be positive, such as direct economic payments, or negative such as taxes for poor land use. They may involve cash payment but can also be through tax credits or forgiveness of debt. Within South Africa, the newly established land tax could be a major driver for conservation. Zero rating land tax on key conservation areas would be one mechanism to promote conservation (Pence et al., 2003).
- Approval and recognition. Regional competitions with awards for conservation activities can be an incentive for conservation. For example, in the Kimberley area of South Africa, there is a landowner-targeted program to promote breeding of raptors.
- Market creation or improvement. The state can create markets for environmental services. Examples are carbon credits, promotion of ecotourism, provision of information on markets, and the introduction of certification schemes (such as "badger friendly" honey).
- Tradable development rights. Landholders are granted tradable development rights that are scarce. This creates a market value for resources.
- Regulatory control. The enactment of laws and their enforcement, including the types of social prohibitions that served this function historically.

Inappropriate agricultural subsidies need to be removed. Within southern Africa, and especially in South Africa, Namibia, and Zimbabwe, where there is extensive private land ownership, wildlife management is proving to be a more economically viable land use option than cattle ranching in the arid and semi-arid areas. It has been suggested that previously, cattle ranching survived as the main land use only because of the large direct and indirect subsidies that supported it (Child 1988; Bond et al., 2004).

5.4.2 Matrix management on communal land

"Common property resource management" is the phrase used to describe the management of shared resources. A common property resource has been defined as any resource that is subject to individual or group use but not to individual ownership and is used under some arrangement of community or group management (Mol and Wiersum, 1993). Hardin's (1968) paper popularized the concept of the "tragedy of the commons" and suggested that communal resources are particularly prone to overexploitation. However, the evidence is that many group-managed resources are not being destroyed, and it was realized that degradation is not an inevitable result of group management (Bromley and Cernea, 1989; Lawry, 1990; Ostrom 1992). A number of criteria have been identified under which group management is most likely to be successful (e.g., Baland and Platteau, 1996; IFAD, 1995; Ostrom, 1992; Wade, 1987; Lawry, 1990; Cousins, 1996; Shackleton et al., 2002).

Changes in human population density and resource use patterns also mean that new systems of resource management need to evolve. Building on communal property resource management theory, a new paradigm of Community-Based Resource Management (CBNRM) is starting to spread across the African continent. The Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) was initiated in Zimbabwe in the early 1980s as one of the first experiments in this regard. Most southern African states now have some form of CBNRM program (Murphy, 1997;

Fabricius et al., 2004), partly due to the support they received from official development aid agencies. Key to these early CBNRM programs was the identification of the need to devolve ownership and management to the lowest possible level. Although this devolution of power is still seen as important, it is clear that devolution on its own is not a sufficient criterion to initiate successful CBNRM. Fabricius et al. (2004) review the current status of CBNRM in southern Africa. Even though they identify many pitfalls in CBNRM and admit it has not always been as successful as initially envisaged, they still promote CBNRM as the means of achieving both community development and increased sustainability of the natural resource base. They identify seven principles that they see as paramount to sustainable CBNRM:

- A diverse and flexible range of livelihood options is maintained.
- The production potential of the resource base is maintained or improved.
- Institutions for local governance and resource management are in place and effective.
- Economic and other benefits to provide an incentive for wise use of the resource exist.
- There are effective policies and laws, they are implemented, and the authority is handed down to the lowest level where there is the capability to apply it.
- There is sensible and responsible outside facilitation.
- Local-level power relations are favorable to CBNRM and are understood.

CBNRM principles are being used in many areas of southern Africa as a mechanism to enhance the local biodiversity and to ensure long-term sustainability of the biodiversity. This includes all of the transfrontier parks throughout southern Africa, the Wild Coast Initiative in South Africa, Administrative Management Design for Game Management Areas (Zambia); Communal Areas Management Program for Indigenous Resources (CAMPFIRE) (Zimbabwe); Community-Based Natural Resource Management Program in Conservancies (Namibia); and Community-Based Natural Resource Management Program in Controlled Hunting Areas (Botswana).

6 Conclusions

The current extensive conservation network in southern Africa is poorly configured to adequately protect the biodiversity of the subcontinent. It is even less suited to preserve biodiversity in a climatically changing environment. The largest proportion of biodiversity is still found outside of the reserve areas, despite the impacts of land transformation and degradation.

With the anticipated impacts of climate change over the next 50 to 200 years, many species will have to move from their current locations to track areas with suitable climates. To facilitate this process and minimize species loss, a multitude of strategies are needed. Creating an environment that is permeable to species migrations can be achieved through realigning reserves and ensuring that land use outside of reserves is biodiversity friendly. Where species are unable to move on their own, facilitated translocations will need to be considered. As a precautionary measure, and for species with no future habitats, it will be necessary to engage in ex situ conservation.

A combination of strategic conservation planning tools and individual species movement models makes it possible to design new configurations of reserves to better conserve biodiversity in a climatically changing environment. With climate change, strategic conservation becomes a shifting target, and it is therefore important to protect the migratory corridors and not simply a single end point. In this regard, it is important to realize that entire habitat will not move, but rather individual species will move at different rates, which will result in new habitat structures. Reconfiguring the reserve network, although strategically important, may be difficult in practice because of costs and the difficulties in acquiring new areas for reserves. The most cost-effective mechanism to both conserve biodiversity and to allow species to move to new habitats is to ensure that the areas between conservation areas are permeable to species migration. From an economics perspective, where reserve expansion is envisaged, it is areas with high-opportunity costs of alternative land use options that should be prioritized for formal reserves, and contractual reserves should be considered for areas of more marginal land.

Facilitating the movement of species to new suitable habitats will be required where autonomous dispersal does not take place. There is a long history of artificially transferring large herbivores, carnivores, and some birds between reserves in the region. It is the movement of smaller animals, insects, and plants that will be of greater concern.

Within southern Africa, the mechanisms to ensure biodiversity-friendly management of the matrix are likely to differ significantly between areas of private land ownership and areas with communal land ownership. Direct incentives such as tax rebates, assistance with vegetation management (e.g., subsidized alien vegetation control), and education may be sufficient to change behavior on private land. Contractual reserves, where the state compensates private land owners to manage portions of their farms as areas for biodiversity conservation, are also an option. On communal land, practices based on CBNRM principles are the most likely option. There is growing evidence that inappropriate agricultural practices are, in many instances, the result of distorted market forces and that more biodiversity-friendly practices may be favored if these are removed. Allowing private ownership of wildlife has greatly increased the extent of private game ranches. It is in the landscapes profoundly transformed for crop production where the greatest challenges to maintain a biodiversity friendly matrix will occur.

For some species, there will be no suitable future habitats, and ex situ conservation is the only option to prevent extinction. Ex situ conservation, although cheaper than other conservation options is less desirable, as it does not conserve ecological function and can only conserve small populations of individual species. It is, however, a safety precaution given the large number of unknowns around impacts on climate change and how individual species will be able to adapt to them.

7. Policy implications

Conservation planners will need a radical change in current thinking, in that they will have to plan for a future where the climate supports a different set of habitats to those supported in the past. No longer will it be appropriate to use historic records to determine which species should be maintained in a specific reserve. Indeed, there might be justification for moving species into areas where they did not occur in the past, yet areas in which in the future, they will have a suitable habitat. This could involve translocation of plants and insects to new areas, something that is not currently part of most conservation strategies. Simply maintaining the current status quo in conservation will result in species extinction from climate change.

There is no single strategy to ensure conservation of all species, rather multiple strategies are needed based on individual species responses to climate change. Some species will become extinct unless there is facilitated translocation or ex situ conservation. For other species, ensuring that they can move through the landscape to track climate changes is the best strategy.

Reserve expansion or realignment as a strategy for strategic conservation should be considered in exceptional circumstances and will be best justified where land for conservation has high opportunity costs, where large areas are involved, where there are clearly identified gradients needing protection, and where high levels of biodiversity loss can be prevented through reserve realignment. Strategic conservation tools coupled to time-sliced climate change predictions can help identify priority areas.

Use of management tools such as fire and grazing intensity (including grazing by mega-herbivores, such as elephants), can help maintain habitat functionality in a similar state to the present.

Managing areas outside of the reserves is the best and most cost-effective option to both ensure that species are able to track changing climatic environments and to strengthen the conservation of biodiversity, in general. Policies should, therefore, be in support of this, and include devolution of resource ownership and management to land owners and communities, securing of community tenure rights, and developing incentives for sustainable resource management. For priority land, the establishment of contract parks with the land owner may be appropriate and more cost effective than the creation of reserves in areas identified as key for conservation.

Climate change research and an understanding on how biodiversity will respond is in its infancy and contains many uncertainties. Ongoing monitoring, research, and model improvement is necessary; fortunately, there are many areas in which our current understanding is sufficient for us to start planning conservation for a climatically changing environment.

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Table 1. *The Area as a Percentage Conserved in Southern African Countries in IUCN Reserves (IUCN classes I-V), IUCN Sustainable Resource Use Areas (IUNC Class VI), and Other Non-IUCN Conservation Areas*

Country	**IUCN VI**	**IUCN I–V**	**Total IUCN**	**non-IUCN**	**Total**
Angola	0.0	6.7	6.7	5.5	12.2
Botswana	0.0	18.0	18.0	12.7	30.7
Burundi	0.0	3.7	3.7	0.0	3.7
Congo	0.5	9.3	9.8	8.5	16.6
Congo (DRC)	3.6	4.7	8.2	3.1	10.6
Equatorial Guinea	0.0	17.2	17.2	0.0	17.2
Gabon	0.0	2.5	2.5	14.5	16.4
Kenya	1.6	5.6	7.1	2.5	9.6
Lesotho	0.0	0.3	0.3	20.8	21.0
Madagascar	0.6	2.4	2.9	1.0	4.0
Malawi	0.0	8.5	8.5	0.0	8.5
Mozambique	1.4	4.0	5.4	5.9	11.1
Namibia	0.7	13.2	13.8	3.6	16.7
Rwanda	0.0	11.1	11.1	0.0	11.1
Seychelles	0.0	59.2	59.2	0.0	59.2
South Africa	0.0	5.5	5.5	0.8	6.2
Swaziland	1.0	2.1	3.0	0.0	3.0
Tanzania	0.1	14.8	14.9	16.0	27.8
Uganda	12.6	7.4	20.0	6.1	23.8
Zambia	18.8	8.1	26.8	9.5	35.4
Zimbabwe	4.8	7.9	12.7	15.3	27.9
Total	2.6	7.6	10.2	6.0	15.6

Non-IUCN conservation areas are mostly forest reserves. All data presented in this table are based on WDPA (2005).

Table 2. *The Amount of Conservation per Ecoregion*

	Conservation in IUCN Category I–VI Reserves			Total Conservation Including IUCN and Non-IUCN Reserves Areas. (Some of Which Are Only in the Planning Stage		
Percentage Conserved per Ecoregion	**Total Number of Ecoregions**	**Cumulative Percentage of Ecoregions**	**Cumulative Percentage of total Land Area**	**Total Number of Ecoregions**	**Cumulative Percentage of Ecoregions**	**Cumulative Percentage of Total Land Area**
<3%	12	23.1	15.1	8	15.4	10
3–5%	9	40.4	35.1	5	25	19.5
5–10%	10	59.6	53.2	4	32.7	27.1
10–15%	10	78.8	83	12	55.8	60.4
15–20%	3	84.6	86.1	8	71.2	68.7
>20%	8	100	100	15	100	100

All data presented are based on Olson et al. (2001)-studied ecoregions and the WPDA (2005) database of protected areas. This is for the same set of southern and East African countries, including Madagascar, as listed in Table 1. Note that non-IUCN areas include some planned areas that have as yet not been proclaimed. Most of the non-IUCN areas are forest reserves.

Table 3. *Extent of Conservation Versus "Need" for Conservation*

Vegetation Type	**Center of Endemism**	**Area in 1000 km^2**	**Percent Transformed**	**Percent Conserved**
Mopane Shrubveld	no	26	0%	99.8%
Mopane Bushveld	no	209	8%	38%
West Coast Renoster veld	yes	61	97%	1.7%
Mountain Fynbos	yes	247	11%	26.2%

Two extremes shown are based on south African statistics; all data presented are based on Low and Rebelo (1996).

Table 4. *Relative Financial Costs Compared to the Advantages and Disadvantages of Differing Adaptation Options*

	Relative Financial Cost	**Advantages**	**Disadvantages**
Do nothing, i.e., maintain the current conservation strategy	Zero additional cost but there is an existing high current cost of conservation management	The current reserve system is in place and funded. No new land needed. Easier to justify than new land acquisition. Will preserve a large percentage of current biodiversity. Maintains intact habitats and ecological interactions.	Not optimized for climate change. No provision is made for protection in a changing climate, so extinction of some species is inevitable. In most areas, the current reserves do not optimize biodiversity conservation, even for a static climate.
Reconfigure reserves	Very high additional cost if multiple small reserves are added, more cost-effective if existing reserves are expanded or realigned.	Ensure high conservation levels for a changing climate. Allows full state control and management of the land. If adequately funded reserves remain the most secure mechanism for ensuring biodiversity conservation. Maintains intact habitats, ecological process, and a large proportion of biodiversity. Most affordable when linked to existing reserves and for large areas. Best suited to land with high agricultural or development potential.	The high cost. The political aspects relating to acquiring land from private individuals or communities. Very difficult to acquire new land once the land is settled (as it is in many priority areas). Poor predictive capacity currently on how species will respond to CC; therefore, it is difficult to know which land to include. Requires strategic planning to identify priority areas. Unlikely to ever conserve more than a small percentage of the total biodiversity.
Use contractual reserves	Less expensive per hectare than state-run reserves, especially if small areas involved.	No capital cost for land acquisition. A more cost-effective strategy to deal with small parcels of land than formal reserves. May be less detrimental to other land-based economic activities (e.g. it may be possible to mix agriculture with strategically configured migratory corridors. Similar benefits to reserve expansion, though slightly less secure. Does not require relocation of current land owners and therefore politically sounder option. Cheaper than reserve expansion, especially on agriculturally marginal land. .	Less state control over the land. May require expensive administration and other infrastructure to administer. A recurring state budgetary item that my be cut in the future. May be difficult to secure long-term (indefinite) funding. This is still potentially an expensive option, particularly on land where high-value alternative land use options are available. Requires strategic planning to identify priority areas. Easier to implement on private land than on communal land. May be less effective at conserving some ecosystem processes than conventional reserves.
Matrix Management	Some options are very inexpensive.	Ensures migratory pathways even if limited information is available on	State has limited control. Land conversion will continue to

Conservation outside of reserves	All options are less costly than formal reserves. Because of the land area involved (potentially 5 to 10 times greater than conservation areas), the overall cost may be high.	priority areas. Potentially conserves the greatest amount of biodiversity. May be relatively inexpensive.	threaten some species. Some species cannot be accommodated in populated areas due to human animal conflicts.
Translocation	Relatively cheap compared to the above options, but actual costs will depend on the number of samples translocated and the species involved	The only option for facilitated dispersers, i.e., where habitat cannot be reached by natural distribution mechanisms. Far cheaper than ensuring migratory corridors. Will still require a conservation network into which the species can be reintroduced.	Only conserves a fraction of the genetic diversity within a species. Competitive interactions with other species will be an unknown element. Does not conserve ecosystem processes, but only species. Will need a sound understanding of individual species habitats. Will require extensive research and monitoring to know which species to move, where to move them to, and what species need to be moved jointly (e.g., pollinators or seed dispersers). Potential negative impacts of translocated species on the existing species in the new habitat.
Ex situ conservation	Relatively cheap once the infrastructure is in place, but varies between different types of species.	An “insurance policy” when there is uncertainty as to how species will respond in the natural environment. The only option for "no-hoper" species. The only option where there is total habitat loss. Relatively cheap (but the cost cannot be compared directly with in situ conservation as different objectives are achieved)	Conserves only a tiny fraction of genetic diversity. Conserves no ecosystem processes.

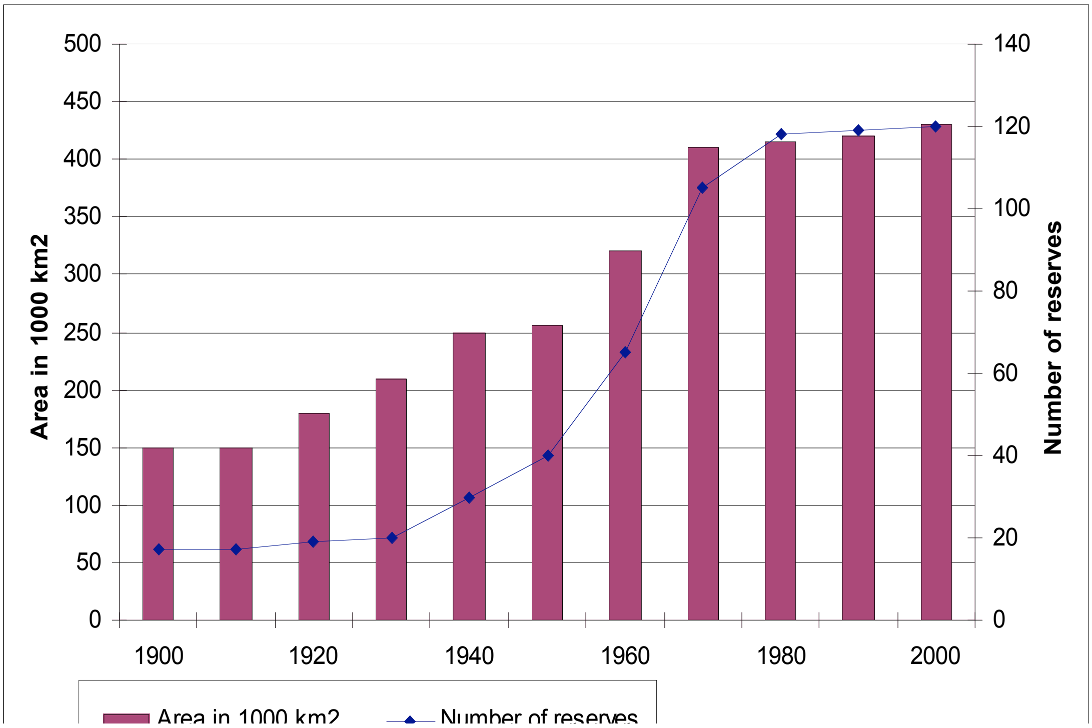

Figure 1. The increase in conservation areas conserved and the number of reserves in seven southern Africa countries (based on Cumming, 2004). **Note:** Only national parks and large reserves in South Africa have been included.

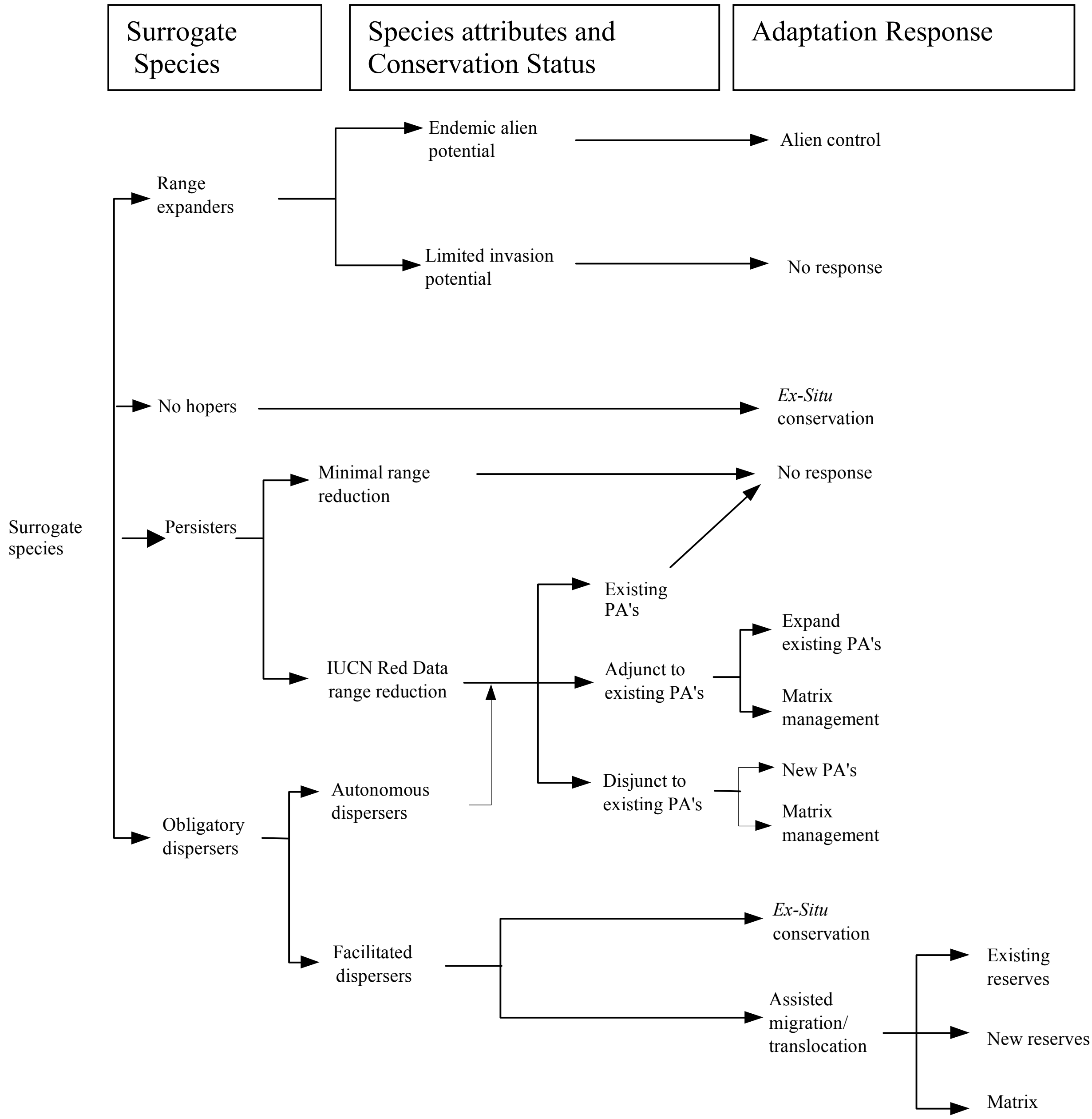

Figure 2. A decision tree for selecting adaptation strategies for different surrogate species based on their response to climate change.