The power of global biodiversity indicators to predict future policy outcomes

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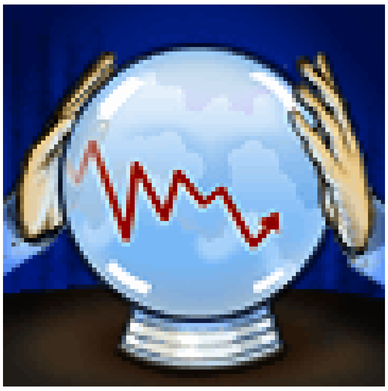

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Contents

List of tables

Abbreviations

RLI: Red List Index

LPI: Living Planet Index

PA: Protected area

NPA: Non - protected area

CR: Critically endangered

EX: Extinct

EN: Endangered

T: Threatened

NT: Near threatened

LC: Least concern

Abstract

We explored the potential of two global indicators of biodiversity to predict the likely outcome of expanding protected area coverage in sub – Saharan Africa by up to 10% of each countries terrestrial land mass, versus improving management effectiveness across the existing protected area network to a Southern African level. The Living Planet Index and the Red List Index were both sensitive to the different policy scenarios, revealing clear differences in their likely impacts upon the study species. The indicators showed that improving management effectiveness significantly reduced relative extinction risk and improved overall trends in abundance, compared to business as usual, or the expansion policy. The sensitivity analysis showed that these results were robust to some of the assumptions regarding external trends, although the study did reveal shortfalls in data availability. The study also highlighted a number of existing concerns regarding the RLI and the LPI, including the risks of using one indicator isolation - and the associated requirement for complimentary indicators. Future use of the indicators could be further improved by more targeted data collection and improved availability, thus reducing levels of uncertainty associated with their use. That the indicators were able to discriminate between competing hypotheses (Walther, Post et al. 2002) by simply communicating complex information in a representative, quantitative and tractable manner (Gregory, Strien et al. 2005), highlights considerable potential for their future application in decision making processes.

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1 Introduction

Representatives of 190 countries at the Johannesburg World Summit on Sustainable Development committed themselves to “achieving by 2010 a significant reduction in the current rate of biodiversity loss at the global, regional and national level” (CBD: Decision VI/26). By adopting the 2010 target, governments explicitly recognized the value of biodiversity and set goals for its conservation, whilst offering a degree of accountability for its preservation (Balmford et al 2005). As such, a range of indicators were developed to measure progress towards this goal (Walpole, Almond et al. 2009). Furthermore, attention has now turned to whether or not some of the indicators could be used in a more proactive capacity by generating forward predictions of various policy options and their likely outcomes (Jones, Collen et al. 2010). By doing so, indicators have the potential to enable decision makers to choose the right policies required to eventually meet the ambitious (Fisher 2009) ‘2010 Target’.

For this to occur, policy makers need simple, relevant measures that relate to policy priorities (Watson 2005). Two such policy realms of high importance are the need to prevent extinctions, and large - scale declines in common species (Gaston and Fuller 2007). Of the existing indicator set, the Red List Index (RLI) and the Living Planet Index (LPI) are perhaps the two most developed indicators (Walpole, Almond et al. 2009), with the former providing a quantification of relative extinction risk (Butchart, Stattersfield et al. 2005), and the latter a measurement of overall trends in abundance (Collen, Loh et al. 2008). For an indicator to capture public attention, it must appeal to public interests and concerns, with due attention to terminology and presentation (Mace and Baillie 2007). In this respect, the LPI, produced by the WWF initially as a communication tool (Collen, Loh et al. 2008), is a good example of an effective tool for public awareness that has had an impact on policy (Mace and Baillie 2007). Likewise, extinction risk is a relatively simple concept for the public to understand (Rodrigues, Pilgrim et al. 2005).

1.1 Aims and Objectives

To ascertain whether the chosen indicators are able to detect trends of interest in the underlying data

To establish whether or not the RLI and LPI are able to detect trends in biodiversity resulting from different policies, thus distinguishing between those policies and from business as usual (i.e. no action)

To assess whether or not these indicators can be used to predict outcomes that would result from the implementation of the different policies, therefore enabling a more informed selection of the desired policy

To analyse how disaggregation and scale affect the effectiveness of indicators between scenarios

1.2 Overview of Thesis Structure

Section 2 introduces the study site and presents an overview of the history, background, literature and a critical examination of previous studies regarding community conservation, placing particular emphasis on ecological economic methodologies and the various techniques for evaluating ecosystem goods and services.

Section 3 provides a detailed presentation of the methods used in the study in sequential order, from pre-appraisal dialogue to data collection in the field and statistical analysis upon return.
In section 4, the results of the study are presented in a logical manner, before being placed in the broader context of previous research in section 5. The study's strengths and limitations are then discussed, with recommendations made for future research given in conclusion.

2 Background

2.1 The Biodiversity crisis and the Convention for Biodiversity

The world is in the midst of a human induced (Mace, Cramer et al. 2010) extinction crisis - an epoch frequently referred to as the ‘sixth mass extinction’(Stork 2010). concomitant with this loss of individual species has been a steep decline in the overall abundance of biological diversity of almost 30% since 1970 (WWF 2008) (Collen, Loh et al. 2008). As species abundance declines so does the ability of ecosystems to maintain the ecological functions and ecosystem services upon which people rely (Gaston and Fuller 2007; Hector and Bagchi 2007; Collen, Loh et al. 2008).

The biodiversity crisis received its first formal global recognition with the establishment of the Convention on Biological Diversity (CBD) at the Earth Summit in 1992 (Mace, Cramer et al. 2010). By 2002 the CBD had set a target to achieve by 2010 a “significant reduction of the current rate of biodiversity loss at the global, regional and national level as a contribution to poverty alleviation and to the benefit of all life on earth”(CBD 2002). However, the 2010 targets can only catalyze effective conservation if systems are in place to tell governments, businesses, and individuals about the consequences of their actions (Balmford et al 2005).

Indicators of biodiversity, ecosystem functions and services, that are rigorous, repeatable, easily understood and widely accepted, can potentially address this issue (Balmford et al 2005). In order for such indicators to reach their full potential, they must be used to inform policy decisions. Ultimately, mature biodiversity indicators should be able to generate forward projections at the global scale, rather than just the local or single species scale, in order to compare alternative policies. Such an outcome would allow for predictive modeling and subsequent comparison of multiple policy options. Fundamentally, these models can clarify mechanisms by which activities and policies affect biodiversity and the services it provides (Balmford et al 2005), thus allowing improved projections of policy outcomes to better inform decision making.

As such, in early 2004, parties to the CBD established a framework of indicators for assessing progress on the 2010 target. However, the timescale between establishing the targets and the proposed deadline was short (8 years), and there are still significant gaps in knowledge regarding key components of biodiversity (Mace and Baillie 2007). This has necessitated the need to adopt, adapt, and strategically supplement existing data, resulting in inevitable compromises compared to what might be an ideal process (Balmford et al 2005). Evident weaknesses in the indicators can have serious consequences through an impaired ability to effectively communicate urgency, hold politicians to account, or to inform them of how best to act (Walpole et al 2009).

The headline biodiversity indicators each have their strengths and weaknesses (Brooks and Kennedy 2004). Habitat and population indices can potentially respond quickly to the changes in biodiversity that they aim to represent (Balmford, Green et al. 2003), whilst habitat indices are geographically representative but tend to sample only a coarse ecological resolution (Araújo and Williams 2001). Even for the indices that are well-developed such as the LPI and the RLI, there are significant challenges in terms of data availability, consistency and relevance (Walpole, Almond et al. 2009), with fewer data for developing countries, for non-vertebrates, and from before 1980 and after 2005 (Butchart, Walpole et al. 2010).

Despite the shortcomings of current indicators, quantification of species trends and the factors governing population and ecosystem viability are vital to forecasting, planning and managing wildlife populations, and in auditing the success of alternative conservation policies and practices (Western, Russell et al. 2009). For example, monitoring systems can provide a useful insight into the mechanisms that drive them and potentially alter their paths. Accruing knowledge in this respect can lead to decisive policy interventions, such as in the case of the international agreement to phase out chlorofluorocarbon use

(Farman, Gardiner et al. 1985), following monitoring ozone levels above the Antarctic.

Indicators can educate and inform the public and policy makers. The power of public opinion to influence political decision making has been well - documented (Hobolt and Klemmemsen 2005; Norrander and Wilcox 2005). On this basis, global indicators can catalyze international public opinion and political will towards international agreements that affect biodiversity at the international scale (Jones et al 2010). Indeed, the ban on the ivory trade, fuelled by a ground - swell of public support (Matthews 1996; Stoett 2002), offers a vivid example of the power of public opinion to shape laws, even at the global scale. However, Jones et al (2010) also suggest that public opinion naturally responds best to indicators at the national or regional scale, a point evidenced in Trillemarka in Norway (Jorstad and Skogen 2009). Here, an area of old-growth forest was at the centre of a conservation debate, whereby the public and media wanted the forest protected, and the forest industry and local government wanted to exploit the forest. At the centre of the debate was the number –of “Red List species” (i.e. those threatened as per the IUCN Red List criteria). When Central government decided to protect Trillemarka, the concept of Red List species was Central to their decision to protect the forest (Jorstad and Skogen 2009). That the Red List is communicated in an official government statement indicates that conceptually, it had entered not only the public conscious but also the highest political agenda, upon which it made an impact.

Different audiences will require different indicators (Mace and Baillie 2007). Policy makers need simple, relevant measures that relate to policy priorities (Watson 2005). Two commonly identified policy realms of high importance are the need to prevent extinctions, and large - scale declines in common species (Gaston and Fuller 2007). Of the existing indicator set, the Red List Index (RLI) and the Living Planet Index (LPI) are perhaps the two most developed indicators (Walpole, Almond et al. 2009), with the former providing a quantification of relative extinction risk (Butchart, Stattersfield et al. 2005), and the latter a measurement of overall trends in abundance (Collen, Loh et al.

2008). For an indicator to capture public attention, it must appeal to public interests and concerns, with due attention to terminology and presentation (Mace and Baillie 2007). In this respect, the LPI, produced by the WWF initially as a communication tool (Collen, Loh et al. 2008), is a good example of an effective tool for public awareness that has had an impact on policy (Mace and Baillie 2007). Likewise, extinction risk is a relatively simple concept for the public to understand (Rodrigues, Pilgrim et al. 2005), whilst the IUCN Red List itself has long been used by a broad community of conservation scientists and therefore carries with it a degree of authority (Mace and Baillie 2007).

Currently, the headline CBD indicators have been used to show past and current trends, rather than guide future policy. A key role of scientific research, from a policy perspective, is to provide prediction rather than merely just description and explanation (Jorstad and Skogen 2009). Considering future events is crucial to conservation. It can help define new strategies and test existing ones for resilience (Sutherland and Woodroof 2009). Scenario modelling can be used to assess the different implications of multiple policy options on biodiversity (Brink and Alkemade 2006), involving comparison of plausible hypothetical situations where all things are equal except for the adoption or not of a specific conservation policy (Balmford, Crane et al. 2005). Aside from Vuuren (2006), there have been no previous attempts at using the 2010 indicators to explore the outcomes of future events or policy options(Jones, Collen et al. 2010). If global indicators are to better inform policy makers when choosing between multiple policies, improved knowledge is needed of how different policies are likely to affect the different indicators (Jones, Collen et al. 2010).

Global indicators of sustainable development (ISD) are however increasingly permeating all levels of government. Local communities, national governments, and the United Nations alike see ISD as an essential evolution in the incorporation of environmental information into good government. Indeed, global indicators have already become “a key site of innovation in which people are working out new conceptual models of nature and society and new relationships among experts, citizens, and public officials for defining and measuring human well-being. More than merely technical changes in measurement protocols, ISD are important new experiments in governance."(Miller 2005).

As forms of statistical quantification (ISD) (of which the LPI is used as an example), have become symbols of a new paradigm of civic epistemologies in local, regional and global settings. Indeed, Miller adds (Miller 2005), this represents a new, post - command - and - control phase of environmentalism in efforts to decentralize and globalize public policy.

As part of this paradigm shift, nation - states no longer hold a monopoly over the definition and production of statistical knowledge. The widespread availability and independent collection and collation of statistical data have given rise to new agents who repackage and reinterpret that data for local and global actors. In the case of the existing indicator set, this would certainly appear to be borne out by the list of actors involved in producing the key indicator set, comprising as it does, a range of NGOs (e.g. Birdlife International), scientific organisations (e.g. ZSL), international organisations (UNEP), international conventions (e.g. CITES) and academic institutions (e.g. University of Queensland).

2.2 Red List and the Red List Index

The IUCN Red List of Threatened Species is the most comprehensive resource detailing the global conservation status of plants and animals (Rodrigues, Pilgrim et al. 2005; Butchart, Akc¸akaya et al. 2007; Mace, Collar et al. 2008). Red lists were intended to raise awareness and to help direct conservation actions for species(Fitter and Fitter 1987). IUCN (IUCN) states that the goals of its red list are to (1) provide a global index of the state of degeneration of biodiversity (2) identify and document those species most in need of conservation attention if global extinction rates are to be reduced.

Today's Red List criteria are based on the 2001 Categories and Criteria (version 3.1) (IUCN 2001). Within this, 3 categories represent a threat status: Critically Endangered (CR), Endangered (EN), and Vulnerable (VU). The categories are defined qualitatively by decreasing probabilities of extinction over increasing time scales and explicitly by 5 criteria (A through to E). These criteria are divided into the following sections; A: population trends, B: changes in range size and habitat, C: small population size and decline, D: Very small or restricted population and E: Quantitative analysis, such as population viability analysis. A species need only meet 1 criteria and not qualifying under any other criteria has no bearing on an assessment (Mace, Collar et al. 2008).

Despite the success and improvements of the Red List, there are still concerns regarding spatial and taxonomic coverage (Rodrigues, Pilgrim et al. 2005; Baillie, Collen et al. 2008; Jones, Collen et al. 2010). Rodrigues et al (Rodrigues, Pilgrim et al. 2005) also make reference to what they perceive as widespread misconceptions that the Red List classifications are still based solely on expert opinion, that the Red List is simply a classification of species into threat categories, or that too few species have been assessed to make it a useful tool for understanding patterns of, and threats to, biodiversity (e.g. (Possingham, Andelman et al. 2002). In some instances, these impressions have even become barriers to the use of the Red List as conservation tool (Lamoreux, Akc¸akaya et al. 2003).

Based on the Red List, the Red List Index (RLI) was developed as one of the indicators to be used to assess progress towards the 2010 target. Using information from the IUCN Red List, the RLI measures projected overall extinction risk of sets of species and tracks changes in this risk (Butchart, Akc¸akaya et al. 2007). It is based on the proportion of species in each category on the Red List, and changes in this proportion over time resulting from genuine improvement or deterioration in the status of individual species. The RLI was initially designed and tested using data on all bird species from 1988–2004 (Butchart, Stattersfield et al. 2004), and has since been applied to amphibians(Butchart, Stattersfield et al. 2005), and mammals and corals (IUCN 2010), with an RLI for cycads pending (Mace, Collar et al. 2008).

Jones et al (2010) estimate the cost of delivering the RLI to be about $1.6 million annually. The RLI is updated every 4 years Jones et al (2010). Jorstad and Skogen (Jorstad and Skogen 2009) note how easily the Red List can be understood by suggesting that, for example, "you do not need to be biologist to understand that "critically endangered" is more severe than "vulnerable". This, combined with the fact that the RLI attracts widespread media attention (957 media stories published worldwide on the 2007 launch) (Jones, Collen et al. 2010), provides the RLI with a high degree of public resonance.

2.3 Living Planet Index

As with the RLI, the Living Planet Index (LPI) was adopted as one of the indicators to address the 2010 target. The LPI is based on a proportional change in abundance measure, providing a sensitive means of assessing the status of biodiversity over the short to medium term (Balmford, Green et al. 2003; Collen, Zamin et al. 2008). As such, the index is easily related to the CBD's 2010 target to decrease the rate of decline in biodiversity, thus providing a powerful communication tool. In respect of the target, a significant change showing the slope of decline becoming less negative before 2010 would indicate the target being met (Collen, Loh et al. 2008).

Initially a communication tool for the WWF (Collen, Loh et al. 2008), the LPI is the centrepiece of WWF's global effort to reduce human pressure on the environment: the Living Planet Campaign (Miller 2005).Based on what is believed to be one of the largest time - series databases on vertebrate population trend indicators, the LPI is a simple, yet powerful way of conveying information about changing trends in biodiversity to decision makers and the public (Collen, Loh et al. 2008).

Like the RLI, the LPI must rely on data collected for other purposes (Collen, Loh et al. 2008), with biodiversity data for tropical countries generally being sparser, despite the higher levels of biodiversity (Collen, Zamin et al. 2008). Indeed, the utility of the LPI method has been questioned on the grounds of geographic coverage (Pereira and Cooper 2006). In order to counter this geographic bias, aggregated trends for tropical and temperate species' populations are given equal weight in the calculation of the index (Collen, Loh et al. 2008). However, this technique was criticised by Pereira & Cooper (Pereira and Cooper) on the basis that all decreases in population size, regardless of whether they brought a population close to extinction, were accounted for equally. From an ecosystem perspective, a decline in an abundant and widespread species is likely to be of greater importance than an equivalent decline in a rare endemic one of small population size (Gaston and Fuller 2007; Collen, Loh et al. 2008). Collen et al, (2008) counter this point by noting that a local decline in a small population of a widely distributed species is not as important as a global decline.

Collen et al, (2008) do however recognise that when all populations are given equal weighting, an extreme example of a potentially perverse outcome could be where 2 populations, one containing 90% of the global species abundance, and the other just 10%, would get equal weighting in the index.

Another potential limitation to producing robust indicators of change in population abundance is variation in the nature of underlying data. A change in type of data collected over time (e.g., from large, wide-ranging, stable populations to small populations of conservation concern) could artificially result in a declining trend toward the present (Collen, Loh et al. 2008). However, it is very difficult to separate out a change in study focus from genuine change in population status, although Collen et al, (2008) suggest one possibility is to examine publication type to try and filter out any publication bias (Collen, Loh et al. 2008).

In the case of the LPI, this problem has not arisen (Owen-Smith and Mason 2005). The time series in the LPI were not published more quickly if they showed declining species abundance, and broadly speaking, declining populations were not disproportionately documented in peer-reviewed journals (Collen, Loh et al. 2008).

Collen et al, (2008) discuss the issue of how informative population data can be, noting that in the case of long - lived vertebrate species, extinction debts can occur (Hanski and Ovaskainen 2002; Mace, Collar et al. 2008).Indeed, in theory, detailed demographic data may be more helpful in many cases (Collen, Loh et al. 2008). However, in reality, such data are seldom available (Katzner, Milner-Gulland et al. 2007). Consideration must also be given to the fact that certain species with naturally high fecundity and mortality rates can potentially exert disproportionate influence over annual time periods. Indeed, whilst certain biological characteristics can better inform population trends across taxa, alongside the dearth in such data, when it is available it is often poorly understood, with the utility of the trait highly heterogeneous depending on threat and taxa, with reducing general applicability (Collen, Loh et al. 2008). By virtue of the above, despite its limitations, population abundance does appears the only tangible option for assessing wider trends in biodiversity (Collen, Loh et al. 2008).

2.4 Protected Areas

Protected areas (PAs) are widely recognized as the cornerstone for *in situ* conservation (Chape, Harrison et al. 2005; Lovejoy 2006; Coad, Burgess et al. 2008; Gaston, Jackson et al. 2008; Jenkins and Joppa 2009).

In recognition of PA importance, PA coverage was endorsed by the seventh Conference of the Parties (CoP7) of the CBD as an indicator for immediate testing in relation to the adopted target of significantly reducing the rate of biodiversity loss by 2010. Additionally, CoP7 set a target that “at least 10% of each of the world’s ecological regions [should be] effectively conserved”(CBD 2004). The 10% target has also been widely adopted by nations to decide their own PA coverage (Coad, Burgess et al. 2008). PAs are also indicators for success in achieving MDG 7 (ensuring environmental sustainability), Target 9 (integrate the principles of sustainable development into country policies and programmes and reverse the loss of environmental resources) and Indicator 26 (land area protected to maintain biological diversity) (UN 2006).

Neither the indicator of areal extent, nor the current global PA dataset (WCMC), provide an accurate barometer of whether PAs are achieving their conservation objective of actually preserving biodiversity (Chape, Harrison et al. 2005; Gaston, Jackson et al. 2008). Indeed, establishing such an effective PA system requires long-term political and financial commitments that go far beyond simply declaring new parks. For PAs to be effective they need to be effectively managed (Hockings, Stolton et al. 2000). There are many examples of ineffective “paper parks” (Jones, Collen et al. 2010), such as those housing Ghana’s “empty forests” (Oates 1999). Indeed, a recent study by Craigie et al (Craigie, Baillie et al.) found an average 59% decline in population abundance of large mammals between 1970 and 2005 across 68 African PAs, with West Africa exhibiting the steepest declines. Perhaps more remarkably, a study by Western et al (Western, Russell et al.) showed even greater declines of large mammals inside PAs than outside PAs in Kenya, a country where PAs are managed specifically for large mammals to stimulate tourism in the country (Craigie, Baillie et al. 2010).

There are however also many instances of PAs providing better protection than non PAs. In Tanzania, country-wide aerial surveys of wildlife populations conducted between 1987 and 1994 revealed that biomass and density for nine large mammal species were significantly higher inside national parks and game reserves, where illegal hunting is effectively controlled, than in adjacent game control or open areas, where illegal hunting is less controlled (Caro, Pelkey et al. 1998). In the Katavi ecosystem, in Western Tanzania, ground surveys in the mid-1990s revealed an even greater decline in large mammal biomass across a protected area gradient. Relative to Katavi National Park, large mammal biomass in an adjacent game control area, open area, and forest reserve was found to be 68%, 97%, and 99% less, respectively (Caro 1999). Setsass et al. (Setsaas, Holmern et al.) showed that densities of Impala inside the Serengeti were 15.3 ind/km^2 compared to 4.3 ind/km^2. Similarly, a questionnaire survey by Struhsaker et al(Struhsaker, Struhsaker et al. 2005), involving 23 scientists and 13 managers, whose collective experience of rainforest PAs in Africa (predominantly located Central and Western Africa) exceeded 567 years, yielded a unanimous response that fauna were far better protected inside than outside PAs.

Even with the apparent heterogeneity in PA performance and the need to quantify conservation programmes (Western, Russell et al. 2009), there are remarkably few studies contrasting PAs with non - PAs (Gaston 1996; Ervin 2003; Western, Russell et al. 2009; Craigie, Baillie et al. 2010).

Western et al (2009) suggest a number of factors that contribute to the paucity of PA - related conservation audits in Africa:

the required level of monitoring is expensive and requires long term commitment and planning; research priorities have tended to focus on charismatic species and urgent conservation threats, with long - term ecological monitoring receiving little attention as a consequence; Exceptions include some the large - mammal ecosystems of Eastern and Southern Africa (predominantly ungulate counts) (Ottichilo, Leeuw et al. 2000; Toit, Rogers et al. 2003; Estes, Atwood et al. 2006; Craigie, Baillie et al. 2010), although these do not compare PAs to non - PAs; A lack of research co-ordination between different organisations creates methodological problems in comparing discontinuous data; Data are also hard to locate by virtue of their general dispersal around different agency reports, private files and journals; Complex ecological interactions make such as rainfall - ungulate and predator - prey oscillations make it difficult to distinguish human - induced from background ecological changes (Owen-Smith and Mason 2005).

Indeed, even when data can be compiled, the complex ecological and social interaction driving the results, often make them difficult to interpret (Gaston, Jackson et al. 2008). Likewise, greater richness or abundance outside than within PAs could arise because (*a*) PAs were originally designated in areas of lower richness or abundance, because of competing interests (e.g., for resource exploitation; (Edgar, Bustamante et al. 2004)), insufficient consideration of biological criteria (Western, Russell et al. 2009), or if there were significant spatial mismatches in the richness and abundance of features for which PAs were and were not primarily designated (Caro 2002); (*b*) habitats and management outside protected areas are favoured by species (Rannestad, Danielsen et al. 2006); or (*c*) species interactions (most obviously predation) cause greater abundance of some species within compared with outside protected areas, resulting in the lower abundance of others (Micheli, Benedetti-Cecchi et al. 2005). Which of these scenarios best explains a given outcome can be difficult to determine.

However, where there has been a lack of systematic monitoring, but where a large number of individual wildlife counts exists, it is still possible to statistically combine such disparate counts and methodologies to compare PA systems to non-PA systems (Western, Russell et al. 2009).

Despite this high level of high variance and unpredictability, Strushaker et al (Struhsaker, Struhsaker et al.) found the following factors to be the most important correlates of a successful PA (table 1):

Table 1: First-order correlations & hypothesized causal relations

Determinant	Score (r)
Positive public attitude	0.83
Effective law enforcement	0.81
Large size	0.59
Low human populations	0.55
NGO presence	0.48
Ecological continuity	0.45

(p-values for Pearson product–moment correlation coefficients: for $r = 0.426$, $p = 0.10$; $r = 0.497$, $p = 0.05$; $r = 0.575$, $p = 0.02$; $r = 0.623$, $p = 0.01$; $r = 0.742$, $p = 0.001$)

2.5 Coverage of PAs

Management issues aside, the distribution of PAs is uneven, particularly with respect to those areas with the strictest protection levels (Brooks, Bakarr et al. 2004; Chape, Harrison et al. 2005; Hoekstra, Boucher et al. 2005). Likewise, there is incomplete coverage of the global system of PAs for mammals, birds and amphibians (Rodrigues 2004).

With regard to the CBD targets, of the 236 nations assessed by Coad et al (Coad, Burgess et al.), the mean protection of their terrestrial environment was 12.2% (+/- 0.86 s.e., n=236), and 5.1% (+/- 0.81 s.e., n=194) for their marine environment. Although mean coverage per nation was above the 10% target for terrestrial area, there was a great deal of variation in protection among nations, and only 45% (106 of 236) of nations had over 10% coverage of their terrestrial area (table 3). Marine protection was much lower with only 14% (28 of 194) of nations reaching the 10% protected area coverage for their marine environments. 9 out of the 15 global biomes have achieved the 10% target.

Further expansion is therefore required to satisfy the 2010 targets. However, this is likely to clash with the immediate needs of socioeconomic development in our increasingly overcrowded world, with real-world examples coming from Africa (Musters, Graaf et al. 2000; Balmford, Moore et al. 2001).

Following on from the inefficiency (Rondinini, Stuart et al. 2004) of ad hoc reservation (Pressey 1994), and the anticlimax of the “silver bullet” strategy of biodiversity hotspots(Myers, Mittermeier et al. 2000), contemporary methods for PA expansion focus on systematic conservation planning(Margules and 2000 2000). Systematic conservation planning is a more versatile(Mace 2000), relies on systematic area-selection algorithms to identify a network of protected areas complementing each other, such that each target species is represented somewhere(Rondinini, Stuart et al. 2004). All these strategies have been shaped by the common constraint of optimization due to the paucity of expendable resources. This carries an intrinsic risk: that coarse or incomplete data on the distribution of biodiversity may lead to an underestimation of the actual minimum target to be achieved(Rondinini, Stuart et al. 2004).

3 Methods

In this section I explain the various steps that were required to test both of the indicators' sensitivity to various policy outputs. Firstly, I describe the selection of policy scenarios, species and any other parameters that provide the framework for the study; secondly, how baseline data was collected and estimated for the populations of the selected species; thirdly, how the species data was manipulated in accordance with the various policies to provide an estimate of its impacts; lastly, how the impacts of each scenario were interpreted by the RLI to provide an estimate of extinction risk, and the LPI to provide an estimate of overall trends in abundance.

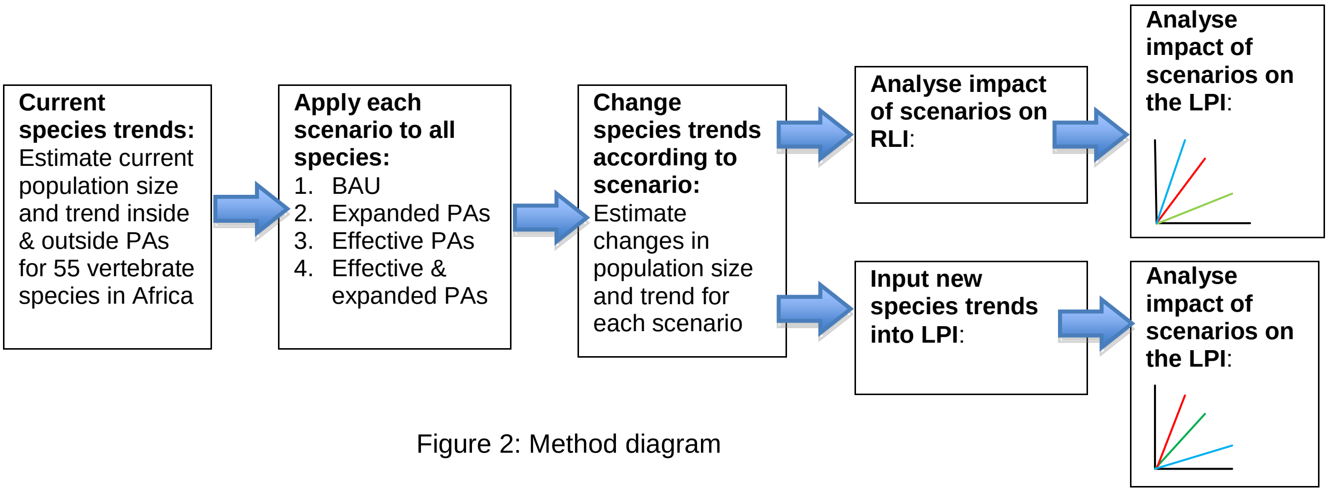

Figure 2: Method diagram

3.1 Policy Scenarios

In addition to a 'business as usual' scenario, three additional policy scenarios were chosen to test the sensitivity and predictive power of the indicators. Multiple scenarios enable comparative analysis and increase the relevance of research to policy makers. The four scenarios are as follows:

I. Business as usual (no policy interference)

II. Expand total terrestrial PA coverage up to 10% of total land cover in each country across all selected regions of Africa:

III. Increased management effectiveness of all existing terrestrial PAs in Africa

IV. Expand total terrestrial PA coverage up to 10% of total land cover in every country across all selected regions of Africa; increase management effectiveness across the new network of PAs.

3.2 Case study & species data

The African continent was chosen due to its relatively well- conserved levels of biodiversity within a global context (Hoekstra, Boucher et al. 2005) with high levels of data comparable to many other biodiversity rich countries located within the tropics (UNEP 2007). The above policy scenarios were applied to East, Southern, West, and Central Africa, with the regions defined according to the United Nations Geoscheme (appendix 3). Ecosystems in Africa are broadly arranged in a latitudinal pattern (UNEP 2007), so whilst the chosen regions have no biological basis to their conception, each does contain within it similar ecological components. The use of internationally recognised regions comprising defined nation states, again allows the research to maintain a greater relevance to policy makers.

North Africa was not included in the study due to a paucity of data on: population numbers, trends, and locations of the chosen species. African islands were excluded as the habitat suitability maps used for scenario 2 (Rondinini, Stuart et al. 2004) are for mainland Africa only.

3.3 Selection of species

Research conducted by Craigie et al (Craigie, Baillie et al. 2010) provided long – term population data for 55 vertebrate species (appendix 1) within continental African PAs, from which population trends could be obtained. The data was collected from published scientific literature, online databases (NERC and Biology 1999; SANParks 2009), park management reports, the Living Planet Index database (IOZ 2009) and personal communications of unpublished data: see Craigie et al (2010) for further details. This data comprised population abundance estimates, density estimates and encounter rates. Data were only included if: population measures were available for at least 2 years; information was available on how the data were collected and what the units of measurement were; the same data collection method was used throughout and the method was reputable; the geographic location of the population was provided and was consistent throughout; the majority of each population was located within an African PA listed in the UNEP/IUCN World Database of Protected Areas (WDPA 2009); and the data source was traceable to a reputable individual or organisation.

3.4 Estimates of population size of each species per country, inside and outside PAs

Country level data of populations of each species inside and outside of protected areas in every study country were collected or estimated from IUCN SSC publications (appendix 1), with the exception of data for Lions *Panthera leo* (Bauer and Merwe 2004). Where specific figures were not given, estimates were calculated using available information on overall numbers, densities, extent of occurrence and correction factors for aerial estimates. In line with East (1999), private lands were not considered to be protected areas.

In order to calculate the Red List status, generation length was also obtained for each species from the Pantheria database (Jones, Bielby et al. 2009). Where generation length was not available, the closest related species was used as a surrogate. Surrogate preference was given in the following descending order:

Tribe
Genus
Family
Sub-family
Order

Where no suitable surrogate was available, age at sexual maturity (Jones, Bielby et al. 2009) was used instead. Information was collected from the IUCN Red List (2010) on the status and threats for each species.

3.5 Estimates of current regional population trends for individual species

For each of the species, trends in population size within protected areas were available in most regions but with differing numbers of observations and over different time frames. The time series were used to estimate trends in population size within PAs for all species and in each region for which data were available.

In cases where there were gaps in the time series, the missing values were calculated using log-linear interpolation. Where t is the year for which the value is interpolated, *t-j* a preceding year with a measured value and *t+i* is a subsequent year with a measured value:

$$N_t = N_{t-j}\left(\frac{N_{t+i}}{N_{t-j}}\right)^{\frac{j}{i+j}} \qquad \text{Equation 1}$$

The inter-annual proportional change in population size, r_t, was calculated for each available pair of consecutive years in a given time series (between t and t-1). Where N is the population measure and t is the year:

$$r_t = \frac{N_t}{N_{t-1}} \qquad \text{Equation 2}$$

From this, the average percentage annual rate of increase or decrease (A) was then calculated using the geometric mean of the inter-annual proportional changes r_t. Where T is the length of the time series:

$$A = \left(\prod_{i=1}^{t} r_i\right)^{1/T} - 1 \qquad \text{Equation 3}$$

3.6 Calculation of regional trends

In order to calculate overall regional index values the rates of inter-annual population change from time series of the same species within the same region were aggregated using geometric means:

$$r_t = \frac{1}{n_t}\sum_{i=1}^{n_t} r_{it} \qquad \text{Equation 4}$$

Following Collen et al. (Collen, Loh et al. 2008), two different methods were used to generate aggregated index values for each species in each region: a log-linear chain method (Loh, Green et al. 2005) and a generalized additive modelling (GAM) technique (Buckland, Magurran et al. 2005). For time series with six or more population measures a GAM was implemented, following Collen et al. (2009), which was specified with the mgcv package framework in R (Team 2009). The GAMs predicted values in time series, including the missing values, prior to the inter-annual population changes being calculated using Eq. (1). Those time series with five or fewer population measures were put directly through the log-linear chain method Eqs. (1) and (2). In cases where the estimate of population size N was zero in any year the minimum-value of all non zero values in the time series was added to all values in the time series. In those time series where GAMs were not implemented the missing values were imputed with log-linear interpolation (Eq.2). After the interpolation stage used for non-GAM treated series, all time series were treated similarly. Next, for species with multiple nt time series contributing data for year t, the mean value of r was calculated across all time series for that species. Species- specific values for r were combined:

$$r_t = \frac{1}{n_t}\sum_{i=1}^{n_t} r_{it} \qquad (4)$$

Multi-species values for r were combined, also using Eq. (3), to yield a single r value for each transition between consecutive years. Finally, the regional index value (I) was then calculated using Eq. 3, with the index value again set to 1 in 1970.

Using the above methods an overall index value was calculated for population change in African PAs between 1970 and 2005, and regional sub-indices for Eastern, Southern, Western and Central Africa. There were insufficient data to produce indices beyond 2005, or for northern regions.

3.7 Calculating population projections per scenario

For each scenario, the appropriate average annual trend *At* was applied to the given populations in order to provide a population projection using the following equation, where T is the time period over which the average annual trend (At) is applied:

$$N_t = N_0(1+A)^T \qquad \text{Equation 5}$$

For the LPI, all species populations were projected forward for a time period of 30 years. For the RLI the populations were projected forward at either 10 year intervals up to 30 years, or 3 times the generation length, whichever were greater, up to 100 years, as per the IUCN Red List guidelines (IUCN 2001). See section below for further details on the use of generation lengths when calculating the RLI.

3.7.1 Scenario 1: Business as usual

Population trends inside PAs: Trends inside protected areas were those calculated for each species in each region at stage 3 using Eq.5.

Population trends outside PAs: In the absence of suitable trend data for species outside of protected areas, it was assumed that population trends for all species would be 25% worse outside protected areas than they were inside protected areas. Therefore, positive trends would be decelerated by 25% and negative trends would be accelerated by 25%. For example: If the average annual rate of change A in a PA was 0.25 (ie a 25% increase), the the rate of change outside protected areas would be A = 0.1875. If the average annual rate of change within PAs was -0.25 (ie a 25% decrease), the rate of change outside of PAs would be -0.3125.

The assumption of a 25% decrease is supported by several studies. In East African savannah ecosystems, Caro et al. (1998) showed biomass and density for nine large mammal species were significantly higher inside PAs (where illegal hunting was effectively controlled) than in adjacent ‘open’ areas in Tanzania, while Caro et al (1999) showed that, relative to Katavi National Park in Tanzania, large mammal biomass in an adjacent game control area, open area, and forest reserve were found to be 68%, 97% and 99% less, respectively. Setsass et al. (2007) showed that densities of Impala inside the Serengeti were 15.3 ind/km^2 compared to 4.3 ind/km^2. Similarly, a questionnaire survey by Struhsaker et al (2005), involving 23 scientists and 13 managers, whose collective experience of rainforest PAs in Africa exceeded 567 years, yielded a unanimous response that fauna were far better protected inside than outside PAs. On the assumption that the superior performance of PAs (in protecting large mammals) illustrated above must be the consequence of a temporal trend, albeit an unknown one, the negative difference of 25% for trends outside PAs is considered to be a suitably conservative estimate.

The effect of this assumption on the results was assessed with sensitivity analyses (see below).

3.7.2 Scenario 2: Expand total terrestrial PA coverage up to 10% of total land cover in each country across all selected regions of Africa

Table 3 shows those countries where expansion was required in line with the above policy. We used the reserve design study by Rondinini et al (2004) as the basis for selecting suitable sites for PA expansion or creation. Rondinini et al used Marxan (Ball and Possingham 2000) to identify reserve systems based on habitat suitability models for 1223 species of African mainland vertebrates - including all of the species listed in appendix 1.

The planning units used by Rondinini et al (2005) were based on protected areas in 2003, using the World Database on Protected Areas (WDPA 2003) and the IUCN SSC elephant specialist group database (Blanc, Thouless et al. 2003), while all non-protected areas were divided into planning units on the basis of water basics. In many countries however, incorporating all of the suitable planning units would have resulted in expansion above and beyond the 10% required of the policy scenario. It was therefore necessary to select a high-value subset of planning units for the reserve expansion scenario. Therefore sites with high irreplaceability values were considered for reserve expansion (the precise irreplaceability values used were country specific to ensure at least 10% of each country was covered by PAs or potential PAs, meaning that some potential PAs were of lower irreplaceability). Areas of low irreplaceability (e.g. 0) were left out of the selection process.

Rondinini et al’s PAs were compared with the current WDPA database for the target countries, in any large obvious anomalies were corrected e.g. Chad and Mali. However, in some cases this was not feasible (e.g. multiple very small reserves in South Africa were ignored). As a result there were some small differences in the estimate of PA coverage between the current WDPA estimate and those in the maps used.

The PU shapefile was converted to a raster, and the 'pick' function (Spatial Analyst Toolbox, ArcGIS 9.3) was used to extract the number of 1km² grid cells of suitable habitat (category 1) for each species in each planning unit. Exceptions were the klipspringer, where category 2 was used, as the map contained only suitability classes 2 and 3, and the impala (*Aepyceros melampus*), for which we did not have a habitat suitability map, so the IUCN range map was used instead.

The added value of each potential PA was calculated by summing the number of suitable cells of habitat across the species: the potential PAs with the highest added value were selected within the budget. This is a simple richness algorithm, and does not include any measure of complementarity or rareness, which may have made the results more robust across species. However, we sought to maximise the chance of increasing population sizes, particularly of species that were not represented; therefore in some cases less species rich sites were chosen where they had high complementary of species not currently in PAs. Less species rich sites were also chosen if all the highest value sites were too large to fit within the budget.

Population trends inside PAs: the same as those used for scenario 1.

Population trends outside of PAs: the same as those used for scenario 1.

Population distribution: Population distribution inside and outside of PAs calculated at stage 3.6 was recalculated in order to factor in any increases in PA coverage by, firstly, calculating the existing density of each species within existing PAs in those countries where expansion of suitable habitat had occurred; Secondly, this population density was then applied to the new PAs in order to provide an estimate of the number of individuals captured by the expansion, with this figure being capped at the number of individuals residing outside PAs before any expansion had occurred.

3.7.3 Scenario 3: Increased management effectiveness of all terrestrial PAs in Africa

Population trends inside PAs: No data exists on the impact of effective management within PAs across Africa. However, PAs is Southern Africa are considered to be most effectively managed (James et al., 1999); Craigie et al., 2010), and provide a benchmark for management in the rest of Africa. Therefore in the absence of better information, we assumed that species in effectively managed PAs would experience the same annual rate of increase in population size as the average across all species in Southern African PAs (A=1.8%), except for those that already had a more positive annual trend, which was assumed to stay constant. The effect of this assumption on the results was assessed with sensitivity analyses (see below).

Population trends outside PAs: Population trends outside of PAs are the same as those used for scenario 1.

3.7.4 Scenario 4: Expand total terrestrial PA coverage up to 10% of total land cover across all selected regions of Africa; increase management effectiveness of all PAs

Population trends inside PAs: the same as those used for scenario 2.

Population trends outside PAs: the same as those used for scenario 2.

Population distribution: Population distribution inside and outside of PAs is the same as that calculated for scenario 3.

3.8 Calculating the biodiversity indicators (LPI and RLI)

3.8.1 Calculating the LPI

Using the population projections created for each scenario as outlined above, index values for the LPI were then created for each scenario and each region for every annual interval up to 2070, following the same methods that were used to calculate regional index values from 1970 - 2005 from the raw data, shown at 3.2.

Following Collen et al (2008) and Craigie et al (Craigie, Baillie et al. 2010) a bootstrap resampling technique was used to generate confidence limits around index values. For each interval, t-1 to t, a sample of n_t species-specific values of population rate of change r_t was selected at random with replacement from the n_t observed values. The bootstrap procedure was then implemented 1000 times and used the bounds of the Central 9500 index values for each year to represent the 95% confidence interval for the index in that year.

3.8.2 Calculating the RLI

Using the population projections calculated for each species in each scenario as described above - each species was assigned to a Red List category at decadal intervals using the IUCN (IUCN 2001) Red List Criteria version 3.1, for categories A and C only.

Following on the work of Butchart et al (Butchart, Akc¸akaya et al. 2007), the overall Red List index value was calculated for each scenario at decadal intervals, using the following formula:

$RLI_t = (M-T_t)/M$

where M is the ‘maximum threat score’, i.e. the number of species multiplied by the maximum category weight (WEX, which is the weight assigned to extinct species; this equals 5 using the recommended ‘equal steps’ weights, with Critically Endangered = 4, Endangered = 3, Vulnerable = 2, Near threatened = 1, Least Concern= 0;). Thus,

$M= W_{EX}.N$

where N is the total number of assessed species, excluding those considered Data Deficient and those assessed as Extinct in the year the set of species was first assessed. (Alternatively, if RLIs for different sets of species are being compared, species that have gone extinct prior to the earliest year of assessment for any group would be excluded.)

The 'current threat score' (T) is defined $T_{t=} \sum_{c} W_C.Nc_{(t)} = \sum_{S} W_c\,(t,s)$

3.9 Sensitivity Analysis

3.9.1 Sensitivity analyses: Trends outside protected areas

In order to test the sensitivity of both indices to the assumptions being made regarding trends outside of PAs, for scenario 1, the trend outside PAs was systematically altered from being assumed the same as the trends within PAs to a 75% increase at 25% intervals. Therefore trends were assumed to be, respectively, 0%, 25%, 50% and 75% worse than trends within PAs.

3.9.2 Sensitivity Analyses: trends in effectively managed PAs

In order to test the sensitivity of both indicators to the assumptions being made regarding trends inside PAs, for scenario 2, the trend outside PAs was systematically altered from the same as average trend within Southern African PAs to worse than within Southern Africa PAs, again at 25% intervals (Therefore trends were assumed to be respectively, the same, 25%, and 50% worse than trends within Southern African PAs)

4 Results

4.1 Time series data

We obtained time series data for 485 mammal populations of 56 species from 77 PAs from Craigie et al (Craigie, Baillie et al. 2010). Time series had a mean length of 19.84 years (SD 9.9; n = 485), with population measures reported for a mean of 10.22 years (SD 8.39), giving a mean fullness of 0.51 population measures per year of time series. 194 (40%) of the time series were sourced from peer-reviewed journals, 58 (12%) were from published reports or secondary sources and 233 (48%) were from unpublished reports. 296 (61%) time series were from aerial surveys and 189 (39%) were from ground surveys on foot or vehicle. 359 (74%) were absolute abundance estimates, 68 (14%) were encounter rates and 58 (10%) were density estimates.

The majority of species in the data set were large (>5 kg) herbivores. The PAs which contributed population time series had a median size of 151,000 ha (mean 574,043 ha; SD 970,472; n = 77), and included 51 PAs in IUCN category I or II (strict nature reserve or national park), 15 in category IV (habitat/species management area), and the remaining 11 in categories V, VI (protected landscape or managed resource PA) or unassigned.

Table 2: Number and percentage of species populations in each region; population trends calculated from abundance data; surrogate trends applied to a population where no abundance data were obtainable.

Region	Number of populations	Data trend	Surrogate trend
Central	32 (23%)	5 (16%)	27 (84%)
West	30 (22%)	13 (43%)	17 (57%)
Southern	34 (25%)	24 (71%)	10 (29%)
East	42 (30%)	35 (83%)	7 (17%)
All regions	138	77 (56%)	61 (44%)

A greater number of population trends were in East Africa, with Central, West and Southern Africa containing smaller and similar numbers. Considerably more trends were obtainable directly from time series data for populations in East and Southern Africa. Central Africa required the use of more surrogate trends than any other region. Just over half of all trend estimates applied to species were obtained directly from time series data for that species in the same region (Table 2).

East Africa contains the largest number of individuals of the study species and the largest proportion of individuals within PAs. Despite having below average for overall study species richness, East Africa has the highest species richness outside PAs. Southern and West Africa harbor the greatest density of species overall but with significantly higher numbers inside PAs than outside. Central Africa has the lowest levels of species richness overall as well as inside and outside of PAs. On average, species richness is 22% greater inside than outside PAs (**Error! Reference source not found.**3).

Table 3: Total abundance of all species (% of total), study species richness per km² and proportion of individuals in PAs.

Region	Total abundance	% in PAs	Spp per km² (SD)	Spp per km² PA (SD)	Spp per km² NPA (SD)
East	7338083 (41%)	67%	0.96 (1.05)	6.75 (6.80)	0.73 (1.07)
Southern	4987661 (28%)	25%	1.40 (0.92)	17.19 (15.11)	0.51 (0.57)
West	3017092 (17%)	39%	1.27 (1.26)	17.34 (32.06)	0.48 (0.61)
Central	2574414 (14%)	42%	0.66 (0.66)	5.54 (6.23)	0.28 (0.27)
All regions	17917249	52%	1.06 (1.21)	11.66 (20.85)	0.52 (0.73)

Abbreviations: PA, protected areas; NPA, non - protected areas; spp, species.

Twelve species (22%) in this study are classified as threatened according to the IUCN Red List – similar to the global mean of 25% of land mammals (Schipper, Chanson et al. 2008), with a mean proportion of 53% (SD: 31%) found in PAs - slightly higher than the mean of 52% (SD: 22%) for all species within the study. However, these threatened species represent only 51 (11%) of the time series in this study.

4.2 Expansion of protected areas

A total of 15 countries saw PA coverage expanded under scenarios 1 and 3, with this figure comprising 40% of the total number of countries included within the study (table 3). However, only a further 3.19% of total terrestrial land cover was incorporated by the expansion. Within expansion countries, average species richness is 0.44 spp per km², just under half the mean average of 1.06 spp per km². 79.9% of total land cover incorporated by expansion occurs in the following countries: Chad, Djibitou, Eritrea, Mali, Mauritania, Niger, and Somalia. Average species density within these countries is only 0.07 spp per km².

Almost half of all species saw some proportion of their population shift from NPAs to PAs There was however considerable variation between species in proportional population change. Most species experienced relatively marginal change, although a few species saw significant changes, with the mean proportional change of 9.32% (SD 18.07%). 22% of species with more than a 1% proportional change are currently listed as under threat under the IUCN Red List criteria, with this figure being the same as the overall mean number of threatened study species. Of the threatened study species, there was a higher than average proportional increase in protected populations of 18.2% (SD 27.07%), largely driven by the Critically Endangered African wild ass (*Equus africanus*) which benefits from a 71% proportional population shift to PAs.

Table 1: Protected Area Expansion per country

Country	*Country Area km^2*	*PA area km²*	*Current PA %*	*% Required*	*Added PA area km²*	*% PA Added*	*Total PA %*
Burundi	26,949	980	3.64%	6.36%	2,045	7.59%	11.22%
Chad	1,269,963	119,664	9.42%	0.58%	16,707	1.32%	10.74%
Djibouti	21,679	0	0.00%	10%	2,181	10.06%	10.06%
Eritrea	122,099	6,041	4.95%	5.05%	19,870	16.27%	21.22%
Gambia	10,797	181	1.68%	8.32%	3,048	28.23%	29.91%
Guinea	244,872	9,019	3.68%	6.32%	16,338	6.67%	10.35%

Lesotho	30,454	151	0.49%	9.51%	6,057	19.89%	20.38%
Mali	1,251,575	32,378	2.59%	7.41%	103,720	8.29%	10.87%
Mauritania	1,040,736	5,125	0.49%	9.51%	108,681	10.44%	10.94%
Niger	1,183,765	83,455	7.05%	2.95%	36,055	3.05%	10.10%
Sierra Leone	72,322	3,074	4.25%	5.75%	5,783	8.00%	12.25%
Somalia	637,661	0	0.00%	10%	76,272	11.96%	11.96%
South Africa	1,220,394	71,501	5.86%	4.14%	56,915	4.66%	10.52%
Swaziland	17,290	473	2.73%	7.27%	2,409	13.93%	16.67%

4.3 Red list index

The following section shows the RLI generated as a consequence of each policy scenario, alongside sensitivity analysis of these results:

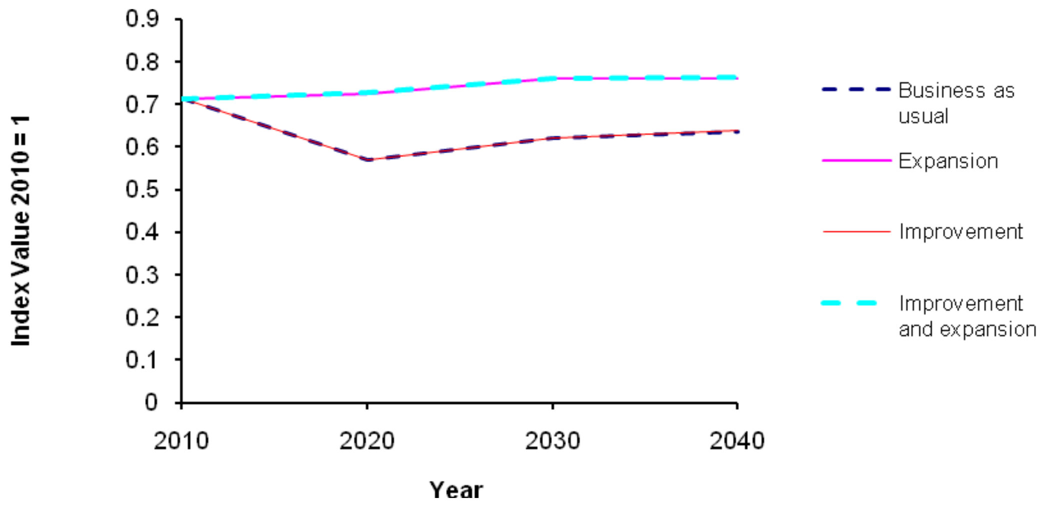

Figure 3: Aggregated Red List Index for all species under all scenarios

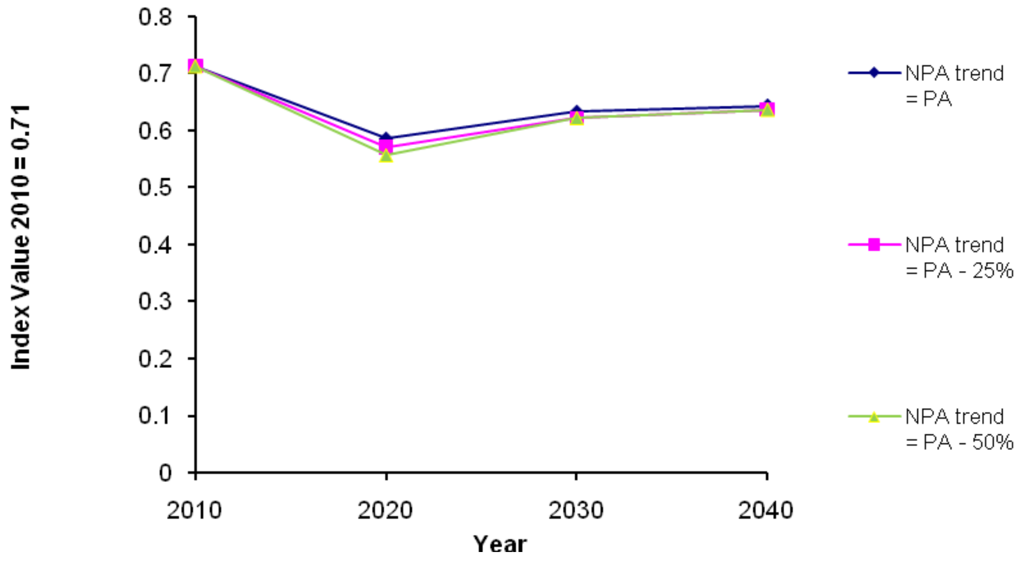

Figure 4: Aggregated Red List Index for all species under scenario 1. Blue line represents same trends being inside PAs as outside; pink line represents trends outside PAs being 25% worse; green line represents trends outside PAs being 50% worse.

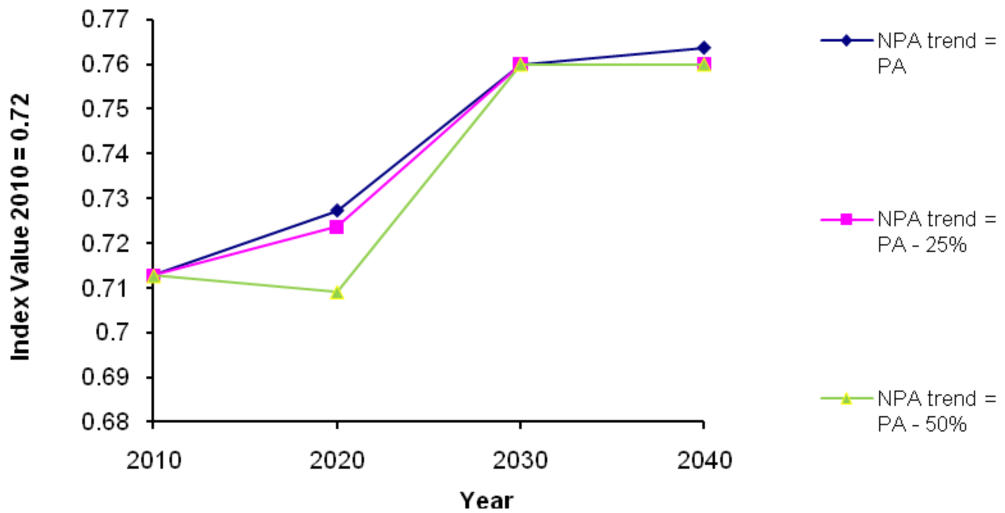

Figure 5: Aggregated Red List Index for all species under scenario 3. Blue line represents same trends being inside PAs as outside; pink line represents trends outside PAs being 25% worse; green line represents trends outside PAs being 50% worse.

The RLI clearly shows that increasing management effectiveness to PAs can significantly reduce the relative risk of extinction for aggregated species. The two scenarios with improved management effectiveness within PAs showed similar results, with overall a significant improvement in the index value. By contrast, expansion of PAs alone showed little impact, and in fact was indistinguishable from business as usual. Scenarios 3 and 4, both of which incorporate improved management effectiveness, show improvements in the overall index value, whereas scenarios 1 and 2 both show continued declines. Over all time periods, scenario 3 performs considerably better than scenario 1, while scenario 4 performs better than scenario 2.

Under scenario 1, business as usual, the RLI predicts a continuing increase in aggregated extinction risk for all species, with a particularly sharp decline over the first decade, followed by a recovery in the index value, the rate of which decreases over time.

Scenario 2, expansion of PAs to 10% of each country, follows an identical trajectory to scenario 1 over the first 20 years. In the final 10 years, there is however a slight improvement due to the common chimpanzee (*Pan troglodytes*) population being approximately 400 individuals greater, thus pushing the overall population to 10,095 - just above the 10,000 population threshold between V and NT, thereby allowing the species to be listed as the latter for scenario 3.

For scenario 3, under increased management effectiveness of PAs, the index shows an overall improvement of 6.63%, with a small increase of 1.53% over the first ten years followed by a larger increase of 5.03% up to 2025, from where the index plateaus up until 2035. Although there is only a small change in the index value over the first 10 years, there is still considerable change between categories, with 6 species being uplisted and 7 downlisted; the similar levels of uplisting and downlisting serve to cancel out each other’s impact on the overall index value. Of the species initially seeing an uplisting, all are subsequently downlisted in the following time-period.

Scenario 4, with PA expansion and effective management, exhibits very similar temporal trends to scenario 2, with the same index values over the first

30 years. Scenario 4 does however see a slight improvement over scenario 3 in the final 10 years, with this a consequence of the African wild ass (*Equus africanus*) population moving from CE to E.

Comparison of scenarios 1 & 3 and 2 & 4, shows that increasing PA coverage has relatively little bearing on the RLI. This illustrates that increasing management effectiveness to all PAs has a greater impact in reducing the overall risk of extinction, compared with increasing protected area coverage in those countries that are currently below the 10% threshold - with this appearing to have a comparatively marginal impact.

Table 4: Percentage of species changing Red List status over time under different scenarios

Scenario	Status change	2020	2030	2040
1	Uplisted	0	0	0
	Downlisted	9.09	18.18	5.45
	Total Change	52.73	18.18	5.45
2	Uplisted	10.91	0	0
	Downlisting	12.73	16.36	0
	Total Change	23.64	16.36	0
3	Uplisted	43.64	0	0
	Downlisting	9.09	18.18	7.27
	Total Change	52.73	18.18	7.27
4	Uplisted	10.91	0	0
	Downlisting	12.73	16.36	3.64
	Total Change	23.64	16.36	3.64

Under all scenarios the number of species changing Red List status decreases over time, with an initial surge in the first 10 years (table 4). Scenarios 1 and 2 show greater mean rates of change over the three time periods than scenarios 3 and 4 – with all changes a result of uplisting. For scenarios 3 and 4 however, whilst all changes post 2020 are the result of downlistings, in the first time period, a number of the same species are uplisted under both scenarios before being downlisted in the subsequent ten years. Fluctuations in status are a result of changing ratios of population abundance to different trends. This occurs when a negative trend diminishes a previously larger population over time, to the point where it becomes smaller than a previously smaller population that is increasing by virtue of a positive trend, creating a switching point where the overall population trend shifts from negative to positive, with this determining status under IUCN Red List Criteria A.

It is also the changing ratio of population to trend that accounts for the pronounced drop in the index value in the first time period, followed by a subsequent increase over the next two for scenarios 1 and 2. This trend is driven by certain species such as the tsessebe (*Damascus lunatus*), for whom a proportionately large regional population - in this case East Africa, with a particularly strong negative annual trend (13.4%), drives an overall population decline sufficiently for it to qualify as Critically Endangered under Criteria A due to 87% decline. As time progresses however, the population in East Africa drops and commensurately so does its proportional share of the overall tsessebe population. This allows previously smaller regional populations with positive trends, in this case Southern Africa (1.3%), to become proportionately larger to the point where they can drive the overall trend, albeit for what becomes a heavily depleted overall population. Figure6 shows changes in regional and overall abundance for the tsessebe. and figure7 shows the associated RLI for the tsessebe over the same time frame.

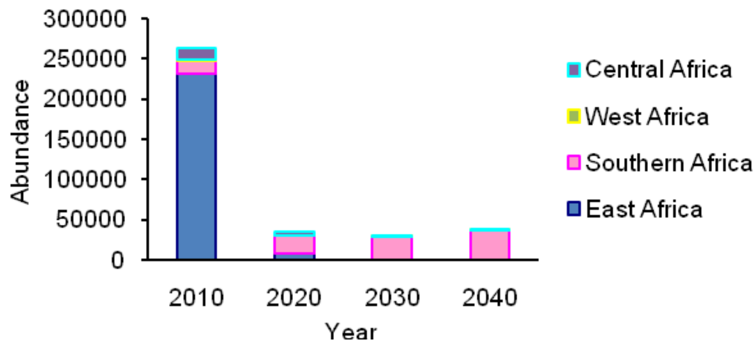

Figure 6: Regional abundance for the tsessebe

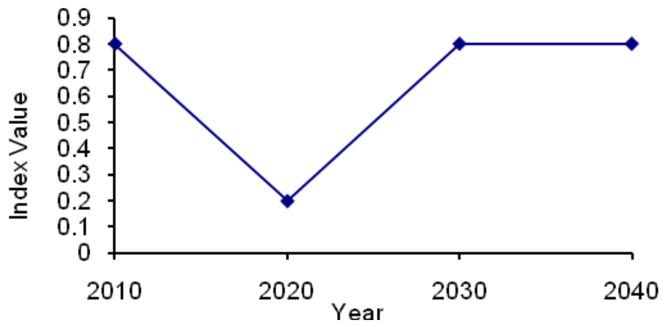

Figure 7: Red List Index for the Tsessebe

4.4 Sensitivity analysis

The RLI results were relatively insensitive to the assumptions being made regarding trends outside PAs. Although these assumptions did exert an influence on the overall trend over the first time period, over the course of the 30 year time horizon results were very similar. This sensitivity analysis therefore illustrates that over time, the positive trend inside PAs become the dominant driver of the index value.

Under scenario 1 the index is relatively insensitive to changes in trends outside PAs. Keeping the trend outside PAs the same as those inside, resulted in only a marginal improvement of 1.14%, compared with a reducing the trend by 25% and 50% - both of which show a convergence in index value in 2030 up to 2040, and only a slight difference in 2020. Under all variations trends assumptions, there is a decline in the index value in the first ten years, suggesting that the relative influence of the assumed trends outside PAs, diminishes over time, whilst the trend inside PAs increases in relative influence (figures 3 and 4). Scenario 2 followed a similar pattern to scenario 1.

Under scenario 3 adjustments in trends outside PAs still affect the index in earlier time periods with the more negative NPA trends affecting more negative RLI values. However, by 2030 there is a convergence in index values before the scenario with equal trends, shows a further improvement. As per the main analysis, scenario 4 exhibited the same trends as scenario 3.

4.5 Living Planet Index

The following section shows the RLI generated as a consequence of each policy scenario, alongside sensitivity analysis of these results:

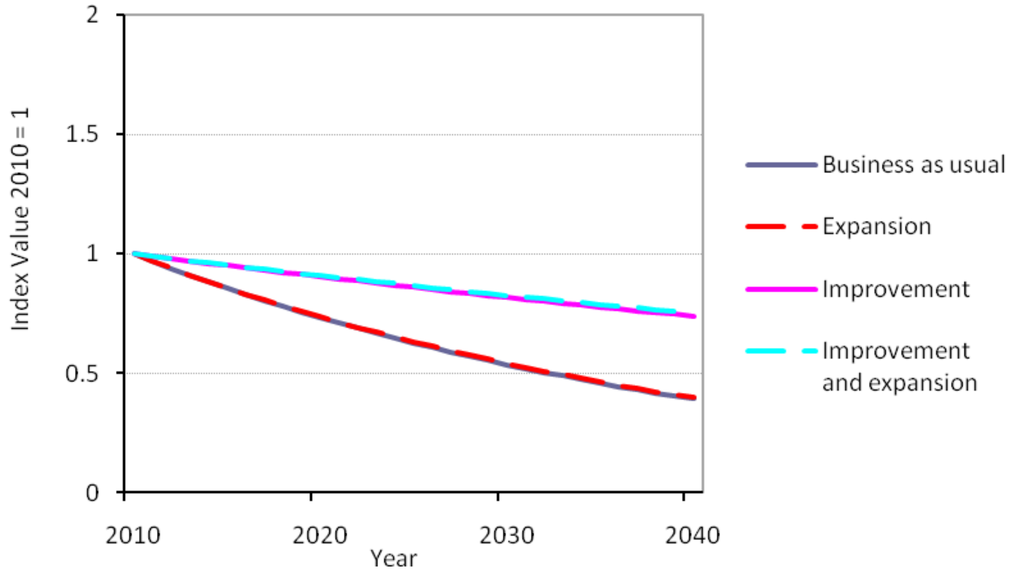

Figure 8: Regionally aggregated population index for all four scenarios

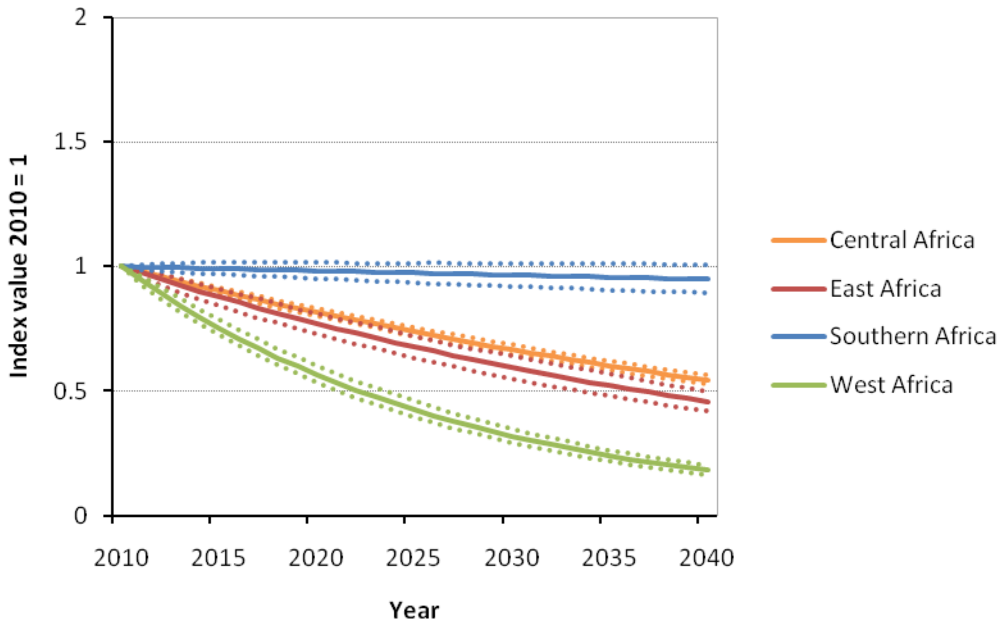

Figure 9: Regionally disaggregated LPI for scenario 1: Broken lines are the 95% confidence intervals produced from bootstrapping with 10,000 replicates

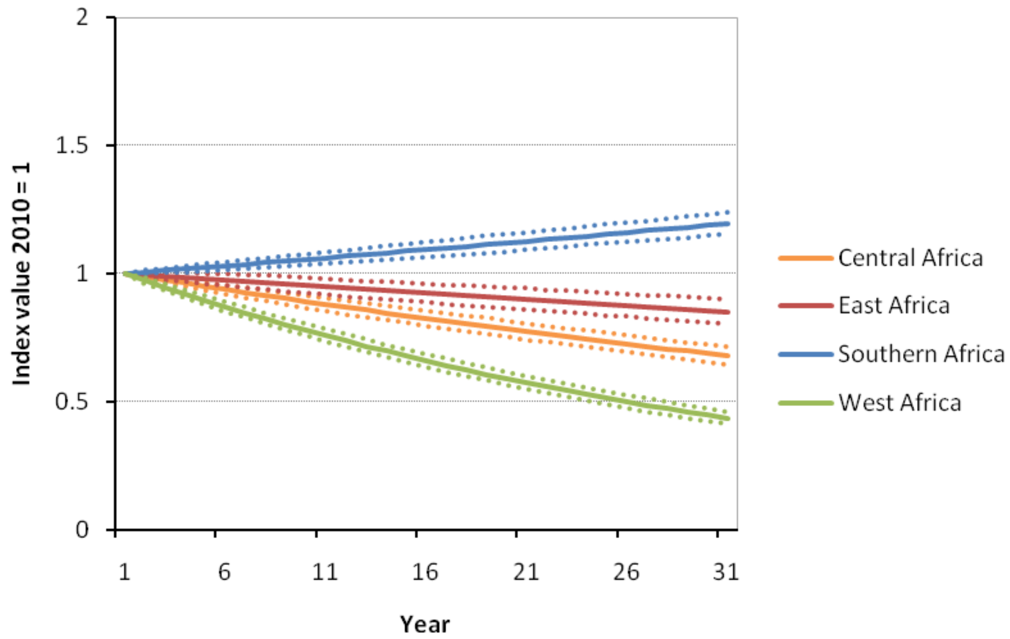

Figure 10: LPI for each regions under scenario 3: Broken lines are the 95% confidence intervals produced from bootstrapping with 10,000 replicates

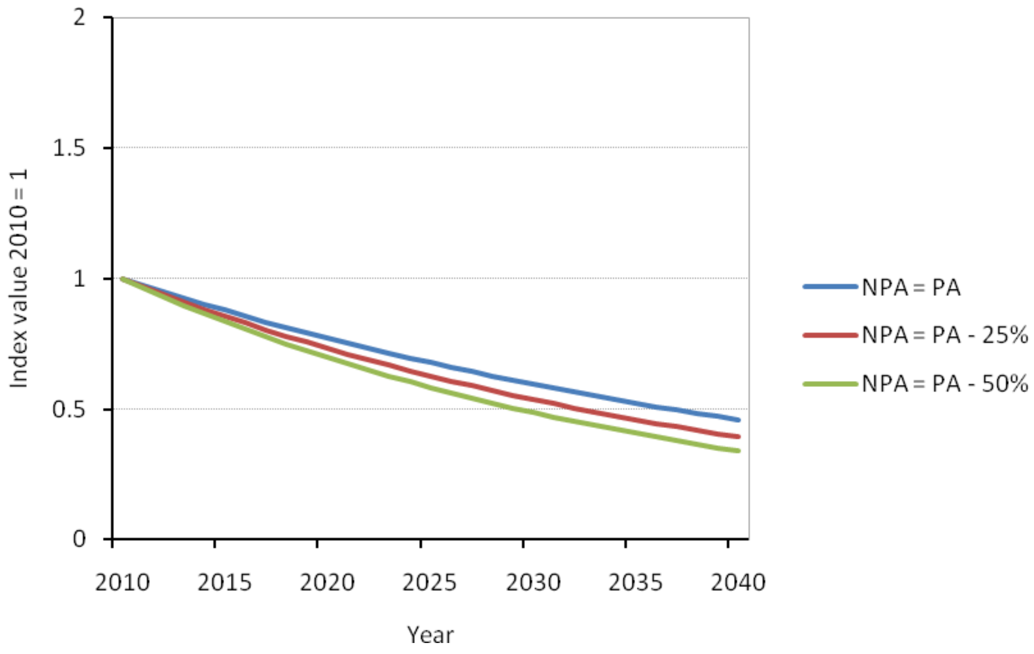

Figure 11: Regionally agreggated population trends for scenario 1: Blue line represents the same population trends used inside PAs as outside; red line represents the trends outside PAs being 25% less positive than those inside; green line represents trends outside PAs being 50% less positive than those inside PAs

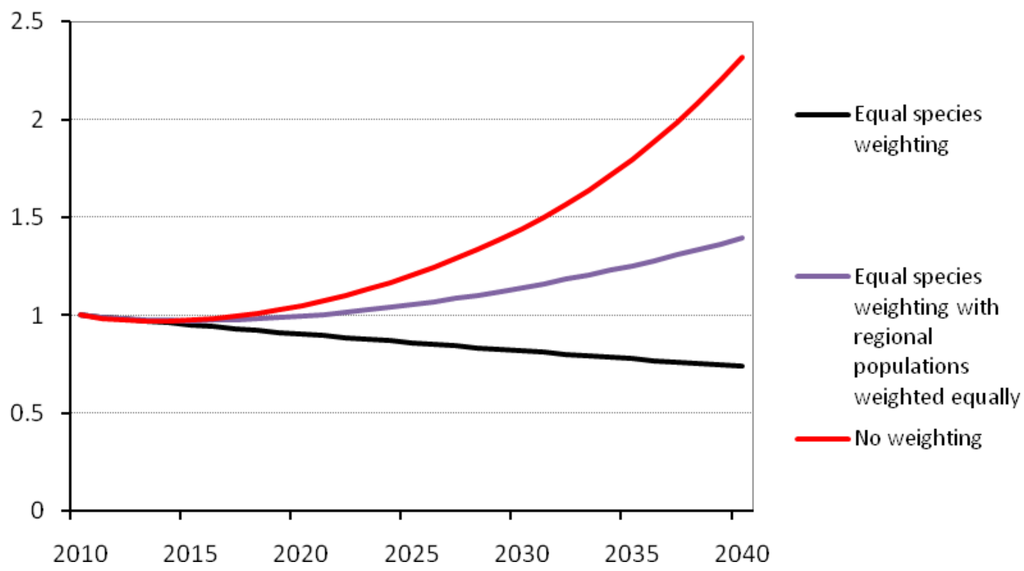

Figure 12: Regionally aggregated population index for scenario 3: Red line represents aggregated trend for all species; purple line represents aggregated trend for all species, with each species weighted equally; black line represents aggregated trend for all species, with each species weighted equally and their regional populations weighted equally.

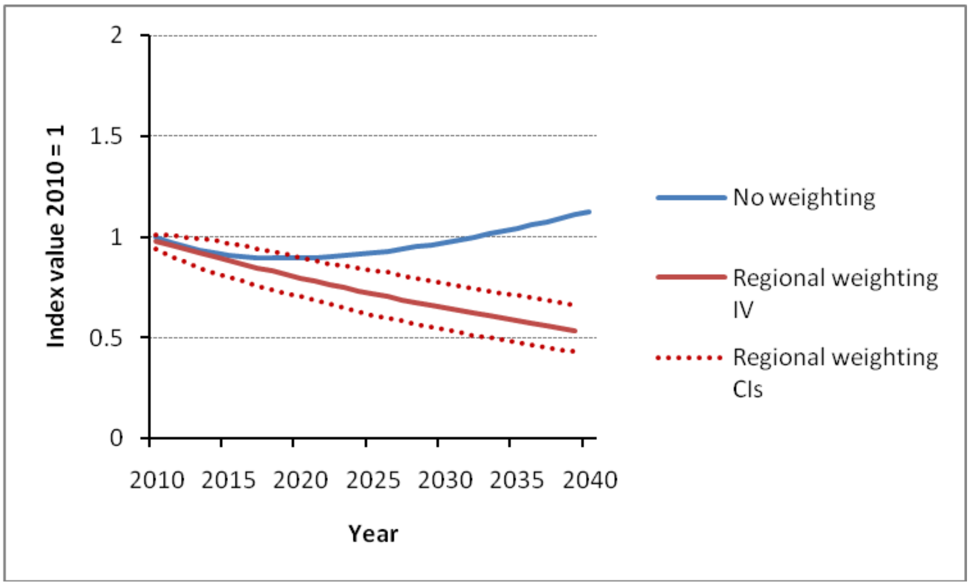

Figure 13: Population index for the African wild dog under scenario 1: Blue line represents regionally aggregated population; red line represents regionally weighted population index

Under all scenarios the LPI shows a decline in population trends, but with a clear difference between the expansion and improvement scenarios (figure 7). For the business as usual scenario the LPI shows a pronounced and consistent decline in the index value over time with it dropping to 0.393 by 2040. Scenario 2 exhibits a similar trend, with a steady but slightly less severe decline in the index value to 0.399 in 2040.

In contrast, under scenarios 3 and 4, the decline is much less aggressive, with the index value comparatively declining only moderately to 7.40 and 7.52 respectively. Again, expansion of PAs appears to have only a marginal impact in reducing the rate of decline.

Scenarios 1 and 3 show considerable regional variation (figures 8 & 9). Southern Africa experiences only a moderate decline in the index value to 0.94 in 2040. Central and East Africa show similar trends to each other, with both experiencing more significant declines with index values in 2040 of 0.54 and 0.45, respectively. West Africa suffers the greatest decline with the index value dropping to 0.18 in 2040, with this being 0.74 lower than the index value for Southern Africa. Scenarios 3 and 4, show broadly similar regional variation, although East Africa appears to benefit most from increasing management effectiveness (figure 9).

Analysis shows that the LPI is moderately sensitive to the assumptions being made regarding trends outside PAs, but that this does not prejudice the overall picture (figure 11). The influence exerted on the overall trend by accelerating declines outside PAs, is not linear, with its relative influence diminishing as the rate of decline grows.

Altering the weighting method employed by the LPI did however make a considerable difference to the results (figure 10). Under scenario 3, when weighting the regional population of each species equally and then weighting each species equally, the LPI shows a moderate decline in the index value to 0.74. When regional populations are aggregated but species are weighted equally, the index shows a considerable improvement to 1.39. Furthermore, when all species are aggregated, with no weighting at all, the LPI shows the overall trend in abundance rising to 1.58.

The difference in how potential methods manifest themselves can also be seen on a species level. For example, figure 12 shows that when regional populations of the African wild dog (*Lycaon pictus*) are weighted equally, the LPI shows a downward trend in abundance for the species. However, when regional populations are aggregated, following an initial decline, the LPI shows a positive trend.

5 Discussion

5.1 Sensitivity of the indicators

Before an indicator can point decision makers towards the right policy, it must first be able to discriminate between the modelled outputs of different policy options. Taken within a pressure – state – response framework (Hoole and Berkes 2010), indicators should be able to predict the future state of biodiversity that may result from different policies (pressures), thus informing decision makers of the right policies required to engineer desired states of biodiversity in the future.

Within the context of this study, the RLI and the LPI achieved this goal. Both clearly distinguished between the four policy options in terms of their likely impacts on biodiversity, providing clear guidance in the process. Although neither of the indices has hitherto been used in a predictive capacity, both are well – developed (Walpole, Almond et al. 2009) and have been successfully deployed to gauge states of biodiversity by interpreting relevant data ((Butchart, Stattersfield et al. 2004; Baillie, Collen et al. 2008; Collen, Loh et al. 2008). The output data of the modelled scenarios was inputted in the same tried and trusted format, thus it is relatively expected that the indicators were similarly responsive and sensitive to the data generated for this study.

Of the two indicators, the LPI was the more sensitive of the two, with both telling a similar story of how successful the policies might be relative to one another. I don't quite understand why this is the case- but presumably your results make it clear Although the RLI and LPI are designed to express different measures of the state of biodiversity, with the former relative extinction risk and latter trends in abundance, one of the key determinants of extinction risk is indeed a high decline rate in abundance (Mace, Collar et al. 2008). As such, Criteria A of the Red List relates to proportional declines in population size. The majority of study species are relatively abundant, meaning the negative outcomes of the business as usual scenario and expansion scenarios were being picked up under Category A, as opposed to

Category C (small population size and decline) in the RLI. Therefore both indicators were effectively distinguishing between policies by virtue of their impacts on population trends.

That the LPI was the more sensitive of the two indicators can partially be explained by the threshold level at which the RLI operates. Changes in the RLI will only occur when a species moves between threat levels, with these threat levels being determined by very precise thresholds. On the other hand, the LPI is responsive to all changing population trends above a certain magnitude, and was therefore able to pick up population changes as they moved between, through and outside of the thresholds required to trigger changes in the RLI.

What the LPI is sensitive to is to some extent determined by the methods used to generate its outcome. LPI methodology has previously been criticised on the grounds that all decreases in population sizes are weighted equally, regardless of the absolute numbers involved in the decline (Pereira and Cooper 2006). Collen et al (2009) acknowledge the potential for perverse outcomes, citing an extreme example of possible impacts (could be) where two populations, one containing 90% of the global species abundance, and one containing just 10% would get equal weighting in the index.' Such concerns did indeed manifest themselves within this study. For example, the African wild dog under scenario 2 experienced steep declines of smaller regional populations but more moderate increases in larger populations meaning that the actual overall trend in abundance was increasing. However, because the downward trend was stronger than the upward trend, the overall trend was presented as a decline by the LPI. This was not an extreme case either. When using equal weighting for all populations under scenario 2, the LPI showed a downward trend, whereas without regional weighting, the trend was positive, with the latter almost certainly being a better reflection on *overall* trends in abundance - and of the policy itself.

From a policy perspective, that the methodology and results of the LPI are open to interpretation has important implications. Although it was still possible to choose between policies using the equal weighting method, if, for example, the policy was presented to politicians in isolation, the underrepresentation of the potentially positive benefits may provide an underwhelming incentive to adopt the policy. On a species level, use of the equal weighting method would indicate that increasing management effectiveness of PAs would have a negative effect on the African wild dog, when the opposite is actually true.

Weighting species equally undoubtedly has its merits, particularly when data is biased towards well studied species. In this respect, the LPI methodology can easily be altered depending on the context with which it is being used. However, careful consideration would need to be given to how this may affect the resonance of the indicator to policy makers and the public alike, if varying methods can alter the meaning of the outcome.

The sheer complexity of biological diversity means that a single measure cannot describe it or track its change (Collen, Loh et al. 2008), or be expected to inform the different policies required to remedy those changes. For instance, Gaston et al (2007) suggest that in addition to threatened species, conservationists need to pay more attention to the depletion of common species, especially given their importance to ecosystem function (Collen, Loh et al. 2008). Using the tsessebe as an example, (your example, I'm assuming) once the rate of decline for the species was at a level above the threshold values for criteria A and the decline was a historical event, it dropped off the threatened list. Used in conjunction however, the LPI is able to draw attention to the historical decline and heavily diminished population size. Used in tandem the RLI and LPI can therefore provide a more holistic prediction of the future state of biodiversity than if used in isolation. Such complementarity is important given that any successful indicator should be designed with their primary role in mind, and recognizing this, it is important to avoid reliance on indicators for purposes other than that for which they were designed (Mace and Baillie 2007). It is important to note however, that to effectively communicate broader trends in biodiversity, a small number of simple and meaningful aggregated indices are likely to be more influential (Balmford, Bennun et al. 2005). Taken as a pair, the RLI and LPI are again able to fit this profile.

Islands aside, risk of localised extirpations are relatively difficult to quantify through the Red List (Gardenfors 2001) and attempts at generating predictions through the RLI were not possible in the time allocated to this study. As per (Collen, Loh et al. 2008), the LPI was, however, easily disaggregated geographically and articulated great variation between regions, with this showing to be one of its greatest strengths.

On a more technical note, the use of the RLI for generating forward predictions highlighted some issues that have previously been raised regarding its application - and that of the Red List more generally - in quantifying current extinction risk. For instance, a large number of species changed categories within the first ten years based on a projection of past trends, potentially contradicting their current Red List status. Whether or not the data used to make my projections was better, worse, or just different, might be open to question. This question is crucial in determining whether a status change should occur, and whether or not this change is genuine (Lamoreux, Akc¸akaya et al. 2003). The requirement for expert judgement in such circumstances has long been noted in the rules for applying Red List criteria (IUCN 2001), and given the inherent uncertainty in making predictions, authoritative opinion would be at lEast as necessary within a predictive context in future assessments.

5.2 Differences between scenarios

The results of the study clearly show that increasing management effectiveness of existing PAs would be a preferred policy option to expanding PA coverage by up to 10% of each study country. Moreover, expansion made little difference at all when compared to the non - expansion policies. That the improvement scenario performed better, is not surprising. The PA trends being applied to the expansion only scenario and the business as usual scenario, were the same as those used in a study by Craigie et al (2010), which showed a 59% decline in regional population abundance between 1970 and 2005, demonstrating large regional differences, with Southern African PAs typically maintaining their populations and West Africa showing the greatest declines. By using the average trend for Southern Africa as the basis for my improved management scenario, projections for the two scenarios were always likely to broadly reflect the regional differences shown in the initial study by Craigie et al, 2010 (Craigie, Baillie et al. 2010).

What is more surprising however, was the poor performance of the expansion scenario relative to business as usual. That the RLI was insensitive to the expansion scenario might have been expected within the context of previous studies that suggest threatened species tend to be found at higher densities inside than outside PAs in comparison to less threatened species (Devictor, Godet et al. 2007). However, in respect of this study, the mean proportion of threatened study species within PAs is only slightly higher than the overall average.

The poor performance can be explained by the relatively few individuals currently residing outside PAs in countries where expansion was required to meet the 10% target, as well as the relatively small area of PA that was added. Therefore few species benefitted from a proportional increase in the number of individuals to whom PA trends were being applied.

There are several possible explanations for the lower species richness outside PAs in expansion countries. For instance, it may be possible that it is the greater PA coverage itself that has lead to higher species richness outside of PAs in countries where expansion was not required. As noted by Gaston et al (2008) PAs often act as sources for the wider landscape, particularly when they constitute substantial components of the remaining or better quality patches of habitat.

From a purely ecological perspective however, species richness in Africa is heavily determined by rainfall (Mutke, Kier et al. 2001), which in turn is largely determined by topography and proximity to the equator (Mutke and Barthlott

2005). Chad, Djibitou, Eritrea, Mali, Mauritania, North and Central Niger, and Somalia account for 79.9% of total land incorporated by expansion, and all experience considerably lower than average rainfall (FAO 2007) and have a much lower average species density outside PAs. These arguments would suggest that the lack of PAs is not the driving factor behind the low levels of species richness in countries where expansion was required. Instead, ecological factors are more likely to be the determining factor.

The above findings would suggest that a lack of PAs is not the driving factor behind the low levels of species richness in countries where expansion was required. Instead, ecological factors are more likely to be the determining factor.

5.3 Implications for conservation

The findings of this study support the assertions of Jenkins and Joppa (2009), who suggest that protection of yet more land may not always be the best conservation strategy for some regions, and that shifting effort toward implementation and enforcement of already declared PAs may best serve conservation. In this respect, whilst expanding each country up to 10% of its terrestrial coverage might still have its merits for national policy and biodiversity at the local scale, on a global scale however its impact is clearly limited. This has important implications for the allocation of globally fungible conservation resources. Although most conservation decisions are made at the national level (Mace and Baillie 2007), the majority of conservation resources available to developing countries, such as those in Africa, stem from international sources (Balmford, Moore et al. 2001; Bruner, Balmford et al. 2004; Brooks, Mittermeier et al. 2006). Expanding PA coverage and improving management effectiveness would incur additional cost to just improving management effectiveness (Bruner, Balmford et al. 2004), but with little extra gain, meaning international agencies would see a considerably greater return by investing in existing PAs. This is not to say however that more intelligent expansion, ignoring political but considering ecological boundaries, would not be beneficial.

Expansion of PAs, targeted at species richness or other specific aspects of biodiversity, can undoubtedly prove beneficial even in the absence of increased management effectiveness. For example, this study showed that expansion was of great benefit to the African wild ass, highlighting the potential of targeted expansion policies for rare, endemic species. Global warming may also necessitate expansion or movement of PAs (Lovejoy 2006), given that current reserves are unlikely to be effective in buffering at global climate impacts as climate and habitat types shift in space (Lee and Jetz 2008). Indeed, recent warming has already affected some species' geographical or altitudinal ranges with clear consequences for species protection (Walther, Post et al. 2002).

Although this study highlights the need for better management of PAs, such rhetoric is not easily translated into action. Effective management of PAs is complicated by a plethora of social (Chan, Pringle et al. 2007; Hoole and Berkes 2010) and ecological (Hansen and DeFries 2007) factors, many of which also require the management of wider landscapes surrounding PAs (Gaston, Jackson et al. 2008; Western, Russell et al. 2009). Indeed, although this study uses average trends obtained from Southern African, the management techniques used there may not easily transcend regional boundaries. Areas where a high percentage of underweight children - used as a proxy for poverty - coincide with a high occurrence of amphibian species and endemic bird areas - a proxy for biodiversity - may indicate areas in which poor people likely have no other choice than to extract some of the natural resources inside PAs (FAO 2007). In Africa, the areas of highest poverty and biodiversity congruence are in East, West and Central Africa, whereas Southern Africa exhibits relatively lower levels congruence (FAO 2007) In this respect, whilst the fines and fences approach typically adopted in Southern Africa (Hoole and Berkes 2010)-(Newmark 2008) may generally benefit biodiversity inside PAs (Craigie, Baillie et al. 2010) such a strategy may have unduly negative impacts on human welfare (Fisher, Maginnis et al. 2005; Hoole and Berkes 2010) in other regions of Africa.

From a regional perspective, this study also highlighted the need for an urgent mobilisation of funds in order to try and stave off what may be irreversible declines in biodiversity, particularly in West Africa Such drastic declines as those predicted under the business as usual scenario, not only threaten biodiversity in the near term, they also compromise the potential longevity of PAs, which are in themselves a likely prerequisite of any future recovery of biodiversity in the long - term. “The first requirement so that individual protected areas can provide a positive contribution to the condition of biodiversity features is that those areas must retain their protected status (Gaston 2005).” In the face of growing populations, crippling poverty and rising opportunity costs (Bruner, Balmford et al. 2004), ‘empty’ forests (Oates 1999) in West and Central Africa are even less likely to be afforded the future protection needed to reverse current trendsThe short time frame for implementing this goal is critical; inadequate management results in environmental degradation and in situations that are ever more complicated and expensive to manage (Bruner, Balmford et al. 2004).

That the RLI shows steep declines under the business as usual scenario, followed by a relative recovery in the index, again highlights the need for an urgent mobilisation of funds in the near - term, whilst also raising the undesirable (Balmford 1999) spectre of baselines shifting yet further in the future. In this respect, it will be important for conservationists to be sure that any future ‘success’ communicated by indicators is framed within the right context, and that expectations are not downgraded in respect of targets for biological recovery (Mace, Collar et al. 2008).

5.4 Limitations of research

The major limitation to this study was the lack of available trend data for species outside of PAs, requiring assumptions to be made regarding those trends. That this these data were not available reflects a wider problem of data paucity in biodiversity rich but economically poor regions such as Africa (Collen, Zamin et al. 2008). It also raises serious concerns about conservationist’s ability to monitor PAs effectively, given that data scarcity makes comparison between inside and outside PAs so difficult (Hansen and DeFries 2007; Gaston, Jackson et al. 2008; Western, Russell et al. 2009). However, our results were relatively insensitive to assumptions about trends, though we did not test the underlying trend data from Craigie et al (Craigie, Baillie et al. 2010). Sensitivity analysis showed that the major assumption of trends outside PAs did not define the overall outcome of the research, with the trends inside PAs being the main driver of both indicators over time. This can again be explained by trend to abundance ratios and the fact that over time the PA trend is being applied to a growing population whilst the NPA trend is being applied to a declining population.

The time series data used to generate population trends was subject to a degree of spatial bias, with Central Africa in particular receiving poor coverage. This necessitated the use of surrogate trends in many instances, with the surrogate trends being applied always the more positive if multiple options were available. Consequently, it is likely that projected declines for Central Africa were underestimated. For all regions, the time series data did not come from a random sample of PAs but from sites with the resources and skills to carry out long-term monitoring programmes (Craigie, Baillie et al. 2010). Assuming that this increased capacity is associated with a greater capacity to deal with threat processes (Hockings, Stolton et al. 2006), then the results projected here are likely to underestimate future declines in general.

This study is exclusively biased towards mammals and does implicitly consider other taxa. How vertebrate trends relate to that of wider biodiversity however, is largely unknown (Collen, Loh et al. 2008). Given the disparities in data availability between taxa this has wider implications for the future application of indicators. However, all species are either large herbivores or carnivores with the majority having large ranges, and can therefore be considered keystone or umbrella species.

The importance of underrepresenting threatened species in the study is debatable (Craigie, Baillie et al. 2010), as it could be argued that major declines in common species potentially have greater ecological consequences than declines in rare species (Gaston and Fuller 2007). A lack of species currently under threat will however have limited to some extent the likely sensitivity of the RLI.

More generally, whereas this research assumed that expanded PAs would carry with them existing trends, in reality, for many species and many PAs, the trend would improve purely as a consequence of the expansion itself, given that many PAs are ineffective by virtue of inadequate size (Struhsaker, Struhsaker et al. 2005; Hansen and DeFries 2007; Gaston, Jackson et al. 2008).

No species level consideration of rmax probably lead to some overestimation of increased abundance (Hone, Duncan et al. 2010) where pronounced population growth occurred, although the LPI code used for R does exclude biologically improbable growth rates, on a generic level.

Assuming that expanded PAs would capture new individuals at the same densities may also have lead to overestimates of population abundance for expansion scenarios, although these changes were only marginal. For ungulates, relative densities close to the edge of PAs could be quite similar, particularly in East Africa (Caro, Pelkey et al. 1998), but perhaps less so in more isolated forest PAs in West Africa (Struhsaker, Struhsaker et al. 2005). For Carnivores, it may also be less true due to larger ranges and greater persecution (Woodroffe 2000).

5.5 Future research

Within the context of this study, future research would look to break down the results of this study by bio-geographic realm, ecosystem and habitat type, and species groups to explore further the specific spatial implications of the policies. In particular, excluding species poor countries from the expansion scenarios would help to elucidate the extent to which species density was a determining factor in its failure.

From a broader perspective, further work is required to explore the applicability of not just the RLI and the LPI, but other indicators, to act as predictors within different policy contexts. Modelling the impacts of pressing contemporary issues such as the potential implementation of REDD could be enormously beneficial in motivating policy makers. Likewise, in the face of growing human populations (Meffe 1993) and increasing human consumption (Baltz 1999), predicting the impacts of likely urban and agricultural expansion on biodiversity could have similar benefits. Within the context of conservation planning, it would be useful to see whether some of the sophisticated computer software used in that field, could somehow factor in biodiversity indicators.

Such research should also look to take explicit consideration of social and economic issues, perhaps exploring in parallel the impact of policy scenarios on social indicators such as the Human Development Index, and economic indicators such as Gross Domestic Produce. Outside of a policy context, indicators could also be used to communicate the likely outcomes of species interactions, for example predator prey relationships.

5.6 Conclusion

“If you aren’t at the table, you’re often on the menu” (Portor and Reindhardt 2007).

Both the RLI and LPI were sensitive to the underlying data resulting from the modelled output of each scenario. That the indicators were so responsive enabled a clear discrimination between which policy option, in this case improving management, would likely prove to be the most successful.

Indicators cannot in themselves however predict what might happen to biodiversity. Such predictions will instead likely come from a plethora of different and complex methods and models that will vary depending on context. The potentially influential role of indicators will come from them acting as vehicles with which to communicate the outcome of any such predictions, in a manner that resonates with decision makers and the public alike.

The ability of indicators to communicate effectively will therefore depend largely on the degree of their exposure, and how well their chosen format can articulate the prediction. The veracity of those predictions will depend on the quality, applicability and availability of data that can be used to inform any modelling. A better understanding of past and current trends and their contextual drivers will allow for more informed estimates of future changes in the state of biodiversity.

Indicator development has necessarily taken an approach of adopting, adapting and strategically supplementing indicators with existing data (Collen, Loh et al. 2008). However, for indicators to fulfil their potential, lessons must be taken from theories of optimal monitoring (Jones, Collen et al. 2010), whereby the right data should be collected in the right format for use by indicators. Given that budgets for biodiversity are thinly stretched (Ferraro and Pattenayak 2006) targeted and purposeful data collection can avoid unnecessary and costly data collection. Indeed, sophisticated models can help to overcome inevitable data gaps (IEEP 2009), and ideally models will be developed in line with indicators. Synchronising data collection, model and indicator development, and their subsequent use within a policy context should form part of a continual cycle of improvement.

Despite the inherent difficulties of making ecological predictions (Carpenter 2002), there is a need to make long – term predictions that reflect the temporal scale at which changes in biodiversity may occur – and the time it may take for policies to influence these changes. Although indicators will inevitably represent a relatively crude prediction tool – science is necessarily a simplification of a complex subjects (Jorstad and Skogen 2009). Indeed, simplification is a requirement of indicators becoming the boundary object (Star 1983) that can enable science to communicate complex changes in biodiversity to policy makers and the public alike.

There has been some debate about whether policy/ public involvement undermines scientific integrity (Morris 2006). However, as Lubchenco (1998) articulates in her article on ‘A new social contract for science’, “Some of the most pressing needs for science include communicating the certainties and uncertainties and seriousness of different environmental or social problems, providing alternatives to address them, and educating citizens about the issues.”

In this respect, communication is vital and conservationists will have to stick their heads above the parapet. Too often biodiversity has been marginalised and unrepresented in the decision making arena. Just as business as usual is not an option for biodiversity, neither is it an option for conservationists. In the same way as business and politicians may use GDP to illustrate economic opportunity costs of protecting biodiversity, so conservationists can use indicators to articulate the biodiversity costs of business and politics.

The deficit model (Owens 2005) of simply providing requisite information to policy makers will not however necessarily translate into the required action being taken. Fisheries were identified as a problem, yet existing policies persisted and some fisheries collapsed (Gezelius and Raakjær 2008). For this reason conservationists need to enter the political arena with eyes open. It is a messy, complex and iterative business, with many players vying for position and attention (Lawton 2007).

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Appendices

Appendix 1: List of all species and key references for population data

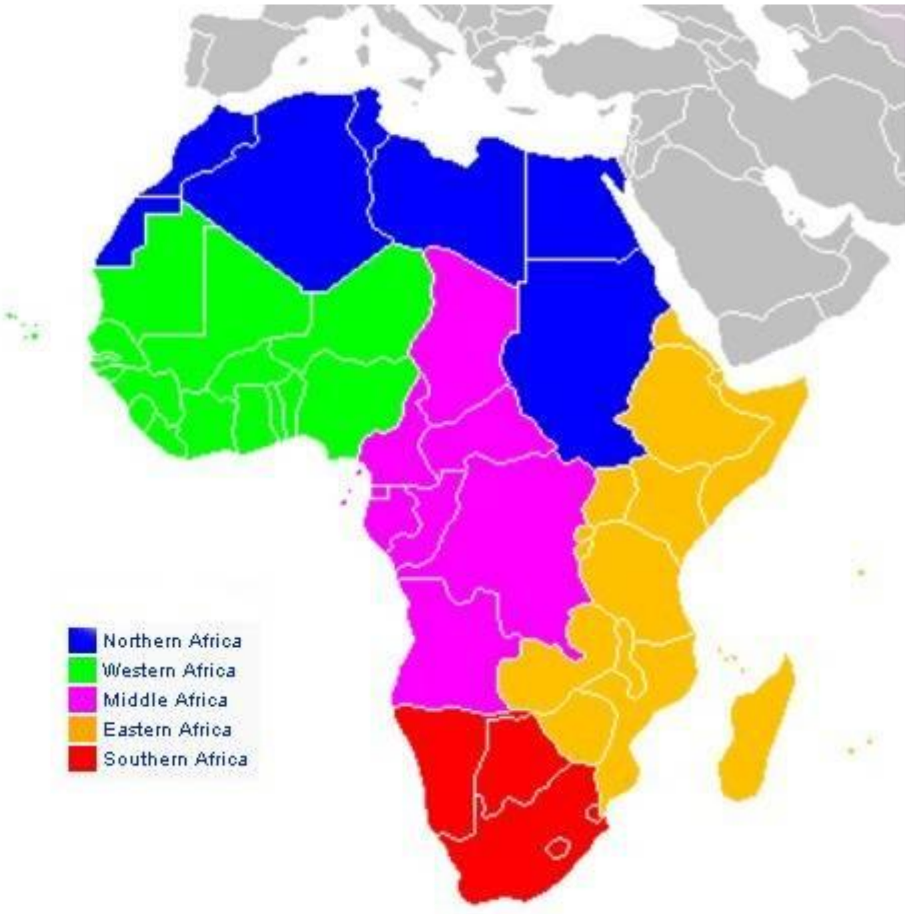
Binomial	Common Name	Key references
Acinonyx jubatus	Cheetah	(IUCN, Group et al. 2007)
Aepyceros melampus	Impala	(East 1999)
Alcelaphus buselaphus	Hartebeest	(East 1999)
Antidorcas marsupialis	Springbok	(East 1999)
Canis simensis	Ethiopian wolf	(Group, ZSL et al. 2004)
Cephalophus dorsalis	Bay duiker	(East 1999)
Cephalophus ogilbyi	Ogilby's duiker	(East 1999)
Cephalophus rufilatus	Red-flanked duiker	(East 1999)
Cephalophus silvicultor	Yellow-backed duiker	(East 1999)
Ceratotherium simum	White rhinoceros	(Esmile and Brooks 1999)
Connochaetes gnou	Black wildebeest	(East 1999)
Connochaetes taurinus	Common wildebeest	(East 1999)
Damaliscus lunatus	Tsessebe	(East 1999)
Damaliscus pygargus	Blesbok/bontebok	(East 1999)
Diceros bicornis	Black rhinoceros	(Esmile and Brooks 1999)
Equus africanus	African wild ass	(Hack, East et al. 2004)
Equus burchellii	Plains zebra	(Hack, East et al. 2004)
Equus grevyi	Grevy's zebra	(Hack, East et al. 2004)
Equus zebra	Mountain zebra	(Hack, East et al. 2004)
Eudorcas rufifrons	Red-fronted gazelle	(East 1999)
Eudorcas thomsonii	Thomson's gazelle	(East 1999)
Giraffa camelopardalis	Giraffe	(East 1999)
Gorilla beringei	Eastern gorilla	(UNEP 2003)
Hippotragus equinus	Roan antelope	(East 1999)
Hippotragus niger	Sable antelope	(East 1999)
Kobus ellipsiprymnus	Waterbuck	(East 1999)
Kobus kob	Kob	(East 1999)
Kobus leche	Southern lechwe	(East 1999)
Kobus vardonii	Puku	(East 1999)
Litocranius walleri	Gerenuk	(East 1999)
Lycaon pictus	African wild dog	(IUCN, Group et al. 2007
Madoqua guentheri	Guenther's dik-dik	(East 1999)
Nanger granti	Grant's gazelle	(East 1999)
Neotragus pygmaeus	Royal antelope	(East 1999)
Oreotragus oreotragus	Klipspringer	(East 1999)
Oryx gazelle	Gemsbok	(East 1999)

Ourebia ourebi	Oribi	(East 1999)
Pan troglodytes	Common chimpanzee	(Kormos, Boesch et al. 2003)
Panthera leo	Lion	(Bauer and Merwe 2004)
Pelea capreolus	Grey rhebok	(East 1999)
Philantomba maxwellii	Maxwell's duiker	(East 1999)
Raphicerus campestris	Steenbok	(East 1999)
Raphicerus melanotis	Cape grysbok	(East 1999)
Redunca arundinum	Southern reedbuck	(East 1999)
Redunca fulvorufula	Mountain reedbuck	(East 1999)
Redunca redunca	Bohor reedbuck	(East 1999)
Sylvicapra grimmia	Common duiker	(East 1999)
Tragelaphus angasii	Nyala	(East 1999)
Tragelaphus buxtoni	Mountain nyala	(East 1999)
Tragelaphus derbianus	Giant eland	(East 1999)
Tragelaphus eurycerus	Bongo	(East 1999)
Tragelaphus oryx	Common eland	(East 1999)
Tragelaphus scriptus	Bushbuck	(East 1999)
Tragelaphus spekii	Sitatunga	(East 1999)
Tragelaphus strepsiceros	Greater kudu	(East 1999)

Appendix 2: List of countries by region

Eastern Africa	*Southern Africa*	*Western Africa*	*Middle Africa*
Burundi	Botwana	Benin	Angola
Djibitou	Lesotho	Burkina Faso	Cameroon
Eritrea	Namibia	Cote d'Ivoire	Central African Republic
Ethiopia	South Africa	Gambia	Chad
Kenya	Swaziland	Ghana	Congo
Malawi		Guinea	Democratic Republic of the Congo
Mozambique		Guinea-Bissau	Equatorial Guinea
Rwanda		Liberia	Gabon
Somalia		Mali	
Uganda		Mauritania	
Tanzania		Niger	
Zimbabwe		Nigeria	
		Senegal	
		Sierra Leone	
		Togo	

Appendix 3: Map of countries per region



Appendix 4: List of species experiencing proportional shift in population residing inside PAs

Species name	IUCN Red List Status	% Increase inside PAs
Equus africanus	4	71.4
Redunca fulvorufula	1	68.0
Eudorcas rufifrons	2	18.6
Connochaetes gnou	1	15.4
Pan troglodytes	4	14.2
Aepyceros melampus	1	12.4
Tragelaphus strepsiceros	1	7.0
Neotragus pygmaeus	1	6.2
Redunca arundinum	1	4.6
Kobus ellipsiprymnus	1	2.9
Damaliscus pygargus	1	2.7
Equus burchellii	1	2.6
Raphicerus campestris	1	2.4
Cephalophus rufilatus	1	2.2
Alcelaphus buselaphus	1	2.1
Philantomba maxwellii	1	2.0
Hippotragus niger	1	2.0
Lycaon pictus	3	1.9
Antidorcas marsupialis	1	1.9
Panthera leo	2	1.7
Raphicerus melanotis	1	1.6
Oryx gazelle	1	1.6
Acinonyx jubatus	2	1.4
Tragelaphus oryx	1	1.3
Sylvicapra grimmia	1	1.2
Giraffa camelopardalis	1	1.2
Tragelaphus angasii	1	1.1