The IUCN Red List Index (RLI)

The IUCN Red List Index (RLI) measures trends over time in overall extinction risk of species, as measured by their category of extinction risk on the IUCN Red List of Threatened Species. The RLI methodology is elaborated upon in detail by Butchart et al. (2004, 2007). It is calculated as the number of species in each Red List category weighted according to their Red List category (0 for Least Concern, LC; 1 for Near Threatened, NT; 2 for Vulnerable, VU; 3 for Endangered, EN; 4 for Critically Endangered CR; and 5 for Extinct in the Wild, EW, and Extinct, EX, as well as species flagged as Possibly Extinct and Possibly Extinct in the Wild under Critically Endangered, CR(PE) and CR(PEW)), divided by the value obtained if all species were Extinct (i.e. the total number of species multiplied by 5), and subtracted from one. This produces a value from 1 to 0, with an RLI value of 1 equating to all species being categorized as Least Concern, and hence that none are expected to go extinct in the near future. An RLI value of 0 indicates that all species have gone Extinct. Ergo, a downwards trend in the graph line between two assessment points (i.e. declining RLI values) means that the expected rate of species extinctions is increasing.

The formula for calculating the RLI requires that (i) exactly the same set of species is included in all time steps, and (ii) the only category changes included are those resulting from genuine improvement or deterioration in status. In other words, changes resulting from improved knowledge, taxonomic revision, or the 2001 change in criteria are excluded (such changes are termed “non-genuine”). In practice, species lists will often change slightly from one assessment to the next (e.g. owing to taxonomic revisions), and many species change category between assessments owing to improved knowledge of their population size, trends, distribution, threats etc. The RLI is only calculated after the earlier Red List categorizations are retrospectively corrected using the current information and taxonomy, to ensure that the same species are considered throughout and that only genuine changes in threat status are included. It is assumed that the current Red List Categories for the taxa have applied in previous assessments, unless there is information to the contrary that a genuine change in status has occurred. Such information is often contextual, e.g. relating to the known history of habitat loss within the range of the species (see Butchart et al. 2007; Hoffmann et al. 2011).

Data Deficient species are not included in the calculation of the RLI. Indeed, any Data Deficient species subsequently reassigned to a category of threat will be for a “non-genuine” reason. For these species, past threat status is then inferred using the best current knowledge, with the default assumption being that unless there are reasons to think otherwise, past status was the same as current.

To calculate the trend in the RLI over time, at least two data-points are needed corresponding to complete assessments of all species according to the IUCN Red List. Mammals were first assessed globally in 1996 (Baillie & Groombridge 1996). A reassessment of all mammals was published in 2008 (Schipper et al. 2008). As part of this study, the 1996 Red List categories assigned by Baillie & Groombridge were retrospectively corrected in line with the methodology outlined above thereby enabling calculation of an RLI for mammals between 1996 and 2008 (see Hoffmann et al. 2011 for further details).

Brief remarks on the 1996 and 2008 observed Red List categories
While the Red List strives to be rigorous and authoritative, it is not without error. In six species, we amended the 1996 and 2008 observed Red List categories from Hoffmann et al. (2010, 2011) where we considered that not doing so would have an impact on the counterfactual. For example, previous assessments for the fallow deer (*Dama dama*) failed to take into account that the vast majority of wild-living populations even within the native range are descendants of domesticated (selectively bred) stock and so are ineligible for inclusion in the IUCN Red List assessment; consequently, the species should correctly be assigned a category of Critically Endangered in both 1996 and 2008. Such changes will need to be reflected on future updates of the Red List.

Some species’ generation lengths assigned in 2008 may be incorrect, but for our purposes we consider them valid to foster consistency in determining the counterfactual. While some species analysed are too poorly known to be sure whether the generation lengths are correct, consistent use within species means that this uncertainty is unlikely to induce spurious conclusions. Where the generation length used in 2008 was not available, we determined generation length ourselves based on available information either for the species directly or from conspecifics and applying the IUCN Red List guidelines for calculating generation length, and cross-checking with Pacifici et al. (2013).

**Defining conservation action**

*How do we define protected areas, and interpret the effects of their cessation?*

Habitat protection is a particularly cumbersome action to consider, because the motivations or intentions driving said protection are often unclear and seldom stated. Consequently, when it comes to habitat protection we considered motive *per se* to be irrelevant (i.e., we did not distinguish cases where habitat protection is driven by watershed protection rather than for biodiversity conservation reasons) as long as the resulting nature of the habitat protection (i.e., the protected area designation) conforms to the widely accepted IUCN definition of a protected area (see below). For example, in some cases, military installations may have a positive conservation impact on a species. We considered military-grade interventions at the habitat scale that conform to the IUCN definition of a protected area as conservation, but otherwise we did not consider military interventions as conservation action even where it may result in positive outcomes for biodiversity.

We use the standard IUCN definition of protected area, namely “A protected area is a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long term conservation of nature with associated ecosystem services and cultural values” (Dudley 2008). Since protected areas submitted to the World Database on Protected Areas (WDPA) must conform to the IUCN definition to be given their designation, any protected areas formally catalogued in the World Database on Protected Areas are, by default, included in our study.

There is clearly some uncertainty surrounding the consequences of a protected area designation ceasing, which may be entirely context dependent. For example, in southern Africa, many PAs would probably have been converted to private land and used in the ecotourism or game-ranching industry, and consequently the loss of PAs may not have led to major declines (in Scenario B, where conservation in private land is included in the counterfactual). Our species-by-species rationales explain our interpretation of the circumstances and the thought process.
How to define private land?

We define private land as land managed for trophy hunting, meat production, and or game-viewing / ecotourism (i.e., land managed exclusively for the purpose of the sustainable production of wild species) and not supported out of state, provincial or NGO conservation budgets. For example, in the case of the black wildebeest (*Connochaetes gnou*), ~80% of the global population is on private land (where the populations are believed either stable or increasing). If only PAs were considered conservation action, it seems unlikely that the species’ status would have changed much from the observed status in 2008 (in part, because many PAs in South Africa would have become private, and because such a small proportion of the population is in PAs in the first place); however, in the absence of conservation in private land, 80% of the population would have been entirely unprotected from hunting, and a deterioration in status seems unavoidable. Even if hunting pressures are lower in southern Africa today than formerly, without PAs (and, in particular, private reserves) this species would probably not be increasing as it is today. For this species then, including conservation in private lands in the counterfactual is the difference between the species being assigned to categories Least Concern or Vulnerable. As noted in the main text, separate scenarios have been constructed to either consider (Scenario A) or not (Scenario B) conservation in private lands as conservation action. For our purposes, we set a threshold of 10% of the global population estimated as occurring on private land (drawing in particular on East 1999) for a species to be considered as potentially affected by conservation in private land and thereby considered when constructing the counterfactual under Scenario B.

We predict that in southern Africa the cessation of PAs would probably result in the conversion of much protected land into private land, some of which would then continue to be dedicated to conservation. In such cases, we assume that the proportion that would have been converted to private land is in the same proportion as current. In such cases, the resulting loss of PAs may not have been so dramatic in terms of changes to counterfactual categories under Scenario A (where conservation action ceased, but private land remained in its current form); however, resulting losses certainly would have greater under Scenario B, where private land is included as a conservation action terminated in 1996.

How to account for introduced populations?

Introduced populations do not count towards assessment of extinction risk (with the exception of whether or not a species is considered EW) unless they conform to an established set of conditions as laid out in the Guidelines for Using the IUCN Red List Categories and Criteria. Such “conservation translocations” (or “benign introductions”) require that “the known or likely intent of the introduction was to reduce the extinction risk of the taxon being introduced” and that “the introduced subpopulation is geographically close to the natural range of the taxon” (see Standards and Petitions Sub-committee 2014; pp 7). We used evidence of intent (e.g., a management or conservation plan) to inform our determination of whether or not introduced populations qualified for inclusion. To construct our counterfactuals, we assume that any introductions that took place post-1996 would not have existed.

How to define ex situ conservation programmes?

In defining *ex situ* conservation programmes, we aimed to distinguish those where conservation was a primary motive (given that ultimately we aim to estimate the impact of
the cessation of the collective conservation budget). For example, *ex situ* populations of the scimitar-horned oryx (*Oryx dammah*) currently includes individuals in both conservation-breeding programmes as well as those in private collections (where the underlying motivation is not straightforward to determine). If in constructing our scenarios we assumed that all *ex situ* populations were "conservation actions", and therefore discontinued in 1996, then this species would have been considered Extinct in the 2008 counterfactuals. However, because *ex situ* programmes were distinguished according to motive (specifically, by considering only active conservation breeding programmes as conservation actions) then a reasonable assumption would be that, in 1996, all formal conservation breeding programmes only would have ceased, but animals in private collections would have remained alive, and the species would then be Extinct in the Wild (as indeed it is currently listed). This distinction is important to a broader group of species, because for the purposes of our study the assumption that private collections are not conservation actions means that any species known to exist in such collections would not be listed as EX in scenarios where conservation actions (including formal captive-breeding programmes) would cease to exist. The species would survive in these private collections from whence it could, theoretically, be restored to the wild. In reality, distinguishing EX and EW is not straightforward as it is impossible to know fully the extent to which some species are kept in private collections. However, for the purposes of calculating the counterfactual RLIs here, the distinction matters not as EX and EW are weighted equally in the methodology. Rather, such a distinction serves mainly to highlight that, in some cases, animals held in private collections may have tremendous conservation value for helping to restore species to the wild, even if that was not necessarily the original intention.

Distinguishing motive requires detailed understanding of every collection (private or not), which is invariably lacking. Consequently, we use listing on the International Species Information System (ISIS; www.isis.org) as a proxy, considering any population in this database as being in formal conservation breeding programmes. We emphasize that this does not mean that captive private populations do not contribute to conservation (per above); however, it could be argued that populations not listed in ISIS are less likely to be incorporated into studbook planning and, therefore, wider management plans for the species.

To estimate the counterfactual Red List categories we assume that when conservation actions end in 1996, all individuals held in captivity are killed and not allowed to escape to the wild.

**Estimating the 2008 counterfactual Red List categories**

We used the following logic in applying the IUCN Red List Categories and Criteria in determining our best estimate for the 2008 counterfactual Red List categories under Scenarios A and B:

- In applying the A2 criterion (population decline in the past), estimation was done based on the likely decline being measured over three full generations back from 2008, but noting that in the 12-year period from 1996 – 2008 there would have been no conservation action. In other words, if a species has a generation length of 8 years, the time period under consideration would be 24 years (1990 – 2008), of which only 12 years (1996-2008) involves zero conservation interventions.
- For species with a 2008 observed Red List category meeting the A criterion, we considered all species to meet exactly the baseline thresholds (30%, 50% and 80%), unless otherwise stated. For example, if species X was assessed as VU in 2008 under
the A2 criterion, we assume that decline to be exactly 30%. For determining the 2008 counterfactual Red List category, the thought experiment requires estimating whether declines would then have been exacerbated to such a degree that species X could have crossed the EN threshold (50%). While it may seem that this rough-and-ready application (necessary because the actual position of most A criterion-listed species within their Red List Category is not defensibly assignable) will obscure the patterns, it is in fact conservative (underestimating change): it guarantees that category changes are only attributed to species for which there is a very high likelihood that these would have taken place. In practice, species might not be at the baseline thresholds for their category but closer to the threshold for the next category (e.g. if VU species X had a 40% decline rather than 30%), in which case they would need a smaller conservation impact to change categories (10% rather than 20%).

- For the purposes of determining the 2008 counterfactual Red List category, we ignored the A3/4 criterion. Our thought experiment was based on the premise that conservation actions ceased between 1996 and 2008, but we made no assumptions as to whether this would have continued thereafter (in the case of A3, in particular, speculating forward three-generations from 2008 based on an improbable scenario is likely to produce results with wide error margins). This approach is again conservative, as application of A3/4 under a no-conservation scenario would be very likely to result in several species having projected 2008 categories in an even higher category of extinction risk.

**Testing the sensitivity of the counterfactual analysis**

To test the sensitivity of our results to our estimates of extinction risk under Scenario A and Scenario B, we recorded – besides the best estimate for the 2008 counterfactual Red List category, as described above – an optimistic and a pessimistic estimate of this counterfactual. We obtained these by asking: how much better/worse could it realistically have been in the absence of conservation? These were generally obtained from the Red List categories adjacent to the best estimate, with the following exceptions:

- There cannot be a more optimistic category than LC, so whenever the best estimate was LC, so was the optimistic estimate. Similarly, whenever the best estimate was EX/EW, so was the pessimistic estimate.
- We assumed it was not plausible that species could have done better without conservation action than with it. Consequently, if a species’ observed status improved from 1996 to 2008, then the optimistic estimate was taken as the same as the 2008 observed Red List category; if a species’ best estimate for 2008 was the same as the observed 2008 category, then we assumed that the optimistic estimate was the same as these; if a species’ observed status in 1996 was CR(PE) or EW, then the optimistic and pessimistic estimates for 2008 were taken as the same as the best estimate in 2008.

Some examples:
- For the impala (*Aepyceros melampus*), with 1996 and 2008 observed Red List categories of Least Concern, our best estimate was that it would have deteriorated to Near Threatened under Scenario A. Given uncertainty about possible rates of declines in the absence of conservation, we also considered the pessimistic possibility that the species would have been deteriorated even further (more than 50%) to Vulnerable; similarly, we considered the optimistic possibility that the species would not have
exceeded a 20-25% decline (sufficient to warrant listing as NT under criterion A), and therefore would have remained listed as LC.

- The addax (*Addax nasomaculatus*) deteriorated in conservation status from EN in 1996 to CR in 2008. Our best estimate of the 2008 counterfactual Red List category was also CR. The pessimistic estimate was that it would become EX, one step above CR, whereas the optimistic estimate was that it would be CR (same as the 2008 observed Red List category).

We calculated a counterfactual RLI under Scenarios A and B based on best, optimistic and pessimistic estimates of the 2008 counterfactual Red List category.

**Comparing the counter-factual in this study with those from previous studies**

To test how the counterfactual RLI developed in the current study compares with counterfactual RLIs using the methodologies of Butchart et al. (2006) and Hoffmann et al. (2010), we constructed two additional counterfactual RLIs (Fig. S2): one where ungulate extinctions would not have been avoided (*sensu* Butchart et al. 2006), and a second where ungulates that we observed improving in status sufficiently to qualify for a lower Red List category of extinction risk between 1996 and 2008 did not do so (*sensu* Hoffmann et al. 2010).

Following the methodology of Butchart et al. (2006), we identified one ungulate listed as Critically Endangered in 1996, the Javan rhinoceros (*Rhinoceros sondaicus*), which we estimate would have gone Extinct (or Possibly Extinct) during the 1996-2008 period without conservation. We then used this 2008 counterfactual Red List category to calculate a counterfactual RLI corresponding to a world where this species extinction would have occurred (because the conservation efforts that avoided the extinction would not have happened). Had the extinction of the Javan rhinoceros actually taken place, the RLI would have declined by 2.6% over the 12 years (0.22%/year), equivalent to 24 species deteriorating by one Red List Category.

Following the methodology of Hoffmann et al. (2010), we identified seven ungulate species whose Red List categories improved over the 1996-2008 period due to conservation action (Table S1), including the vicuna (*Vicugna vicugna*) and Przewalski’s horse (*Equus ferus*). No ungulate species improved in status without conservation action. We then estimated 2008 counterfactual Red List categories assuming that these ungulate species that we observed improving in conservation status to qualify for a lower Red List category of extinction risk over the 1996-2008 time period because of conservation efforts instead retained their 1996 status (i.e., they did not improve in status). Had these improvements not taken place (and their categories instead remained unchanged between 1996 and 2008), RLI values would have declined by 3.3% (0.28%/year), equivalent to 28 species deteriorating by one Red List Category.
Supplementary Figures

Fig. S1. Number of species estimated that would have undergone one or more category changes between 1996 and 2008 in the absence of conservation actions, according to Scenarios A (conservation in private land not considered) and B (including conservation in private land) compared with those changes that did take place (observed). Positive values correspond to improvements, negative values to deteriorations.
Fig. S2. Observed and counterfactual trends in Red List Indices for ungulates during the 1996–2008 period. The observed trend corresponds to actual change (sensu Hoffmann et al. 2010, 2011). In addition to the counterfactual RLIs obtained for Scenarios A and B, two additional scenarios based on previously published methodologies are presented: sensu Butchart et al. (2006), where the conservation actions that prevented one species from becoming Extinct did not take place; and sensu Hoffmann et al. (2010), where the conservation actions that resulted in observed improvements in the conservation status of seven species of ungulates did not take place.
Supplementary References


