# A biodiversity intactness index

**R. J. Scholes & R. Biggs**

*CSIR Environmentek, PO Box 395, Pretoria, 0001, South Africa*

**The nations of the world have set themselves a target of reducing the rate of biodiversity loss by 2010. Here, we propose a biodiversity intactness index (BII) for assessing progress towards this target that is simple and practical—but sensitive to important factors that influence biodiversity status—and which satisfies the criteria for policy relevance set by the Convention on Biological Diversity. Application of the BII is demonstrated on a large region ($4 \times 10^6$ km$^2$) of southern Africa. The BII score in the year 2000 is about 84%: in other words, averaged across all plant and vertebrate species in the region, populations have declined to 84% of their presumed pre-modern levels. The taxonomic group with the greatest loss is mammals, at 71% of pre-modern levels, and the ecosystem type with the greatest loss is grassland, with 74% of its former populations remaining. During the 1990s, a population decline of 0.8% is estimated to have occurred.**

The loss of biodiversity in the modern era, at rates unequalled since the major extinction events in the distant geological past[1,2], is a matter of considerable policy concern. The Convention on Biological Diversity (CBD) has adopted a target of reducing the rate of biodiversity loss by 2010 (ref. 3). For this target to be met, a method of measuring biodiversity status must be agreed on and implemented. At present no scientific consensus measure exists[4], although several candidates have been proposed[5]. The CBD has agreed on a partial set of indicators[6]. Difficulties in establishing operational indicators stem largely from the complex, multi-dimensional nature of biodiversity, which can be defined in terms of composition, structure and function at multiple scales[7].

The CBD developed a set of criteria that an indicator of biodiversity change should satisfy[8]. The salient ones are that it should be scientifically sound, be sensitive to changes at policy-relevant spatial and temporal scales, allow for comparison with a baseline situation and policy target, be useable in models of future projections, and be amenable to aggregation and disaggregation at ecosystem, national and international levels. It should also be simple and easily understood, broadly accepted and measurable with sufficient accuracy at affordable cost.

Whereas there is no shortage of ways to express biodiversity[8,9], none adequately meets the criteria set by the CBD. Most indices require essentially complete knowledge of the biota or the population sizes of individual species, neither of which are achievable conditions at regional to global scales for the next several decades. Many methods are scale-dependent and thus hard to interpret in a comparative context. The most widely used indicators are based either on risk of extinction[10], or on land area under conservation protection[11]. Several indices have been offered that combine either a sparse and selective set of population estimates for indicator species[12], or combine a number of factors that are thought to relate to biodiversity status[13].

A sensitive, realistic and useful measure of biodiversity loss needs to be based on changes in population abundance across a wide range of species, and must consider the entire landscape. At a global scale, habitat loss, including reductions in both quality and quantity of suitable environment, is the main factor responsible for declines in species abundance[1,14]. Other important causes, such as excessive harvest pressure or the effects of pollutants, can also be expressed on the basis of area affected and intensity of impact. This paper introduces an index that takes into account these factors and meets the CBD criteria for policy relevance.

## Attributes of the proposed index

The BII is an indicator of the average abundance of a large and diverse set of organisms in a given geographical area, relative to their reference populations. We recommend calculating the BII across all species within the broad taxonomic groups that are reasonably well described. For most parts of the world this includes plants and vertebrates, and excludes invertebrates and microbes, which are diverse but poorly documented. We exclude alien species in the calculation of the index.

An easily grasped reference population for large parts of the world is that which occurred in the landscape before alteration by modern industrial society. Because accurate data on pre-modern populations are seldom available, contemporary populations in large protected areas serve as a practical reference. Alternative reference points can be defined for parts of the world that had already been highly transformed by the beginning of the modern period; for example, by selecting a specific baseline year within records or reliable memory.

The BII can in principle be calculated exactly by 'bottom-up' aggregation of population data for individual species. However, especially in the highly biodiverse, but poorly studied parts of the world, this will not be a practical option for the next several decades. The proposed strategy is therefore to initially calculate the BII 'top-down'. This is analogous to the top-down/bottom-up distinction that has permitted progress in the assessment of global greenhouse gas emissions[15]. In the greenhouse gas example, the collective national emissions from thousands of individual sources are estimated on an activity basis, rather than source-by-source summation. In the case of biodiversity, we estimate the impacts of a set of land use activities on the population sizes of groups of ecologically similar species ('functional types'). The chosen land use activities range from complete protection to extreme transformation, such as urbanization. All activities are expressed on the basis of the area affected. The index is aggregated by weighting by the area subject to each activity and the number of species occurring in the particular area.

The BII is an aggregate index, intended to provide an intuitive, high-level synthetic overview for the public and policy makers. It can be disaggregated in several ways to meet the information needs of particular users: by ecosystem or political units, taxonomic group, functional type, or land use activity. This provides transparency and credibility. The BII has the same meaning at all spatial scales. It is possible to estimate the value of BII for the past, and project it into the future under various situations. An error bar can be associated with the BII, allowing a monitoring goal to be defined in terms of shrinking the uncertainty range.

## The biodiversity intactness index algorithm

The BII gives the average richness- and area-weighted impact of a set of activities on the populations of a given group of organisms in a specific area. The population impact ($I_{ijk}$) is defined as the population of species group $i$ under land use activity $k$ in ecosystem $j$, relative to a reference population in the same ecosystem type. The BII is calculated as:

$$\mathrm{BII} = (\Sigma_i \Sigma_j \Sigma_k R_{ij} A_{jk} I_{ijk}) / (\Sigma_i \Sigma_j \Sigma_k R_{ij} A_{jk})$$

where $R_{ij}$ = richness (number of species) of taxon $i$ in ecosystem $j$, and $A_{jk}$ = area of land use $k$ in ecosystem $j$

The BII can be disaggregated along different axes. For instance, the intactness of a particular taxonomic group $i$ is given by:

$$\mathrm{BII}_i = (\Sigma_j \Sigma_k R_{ij} A_{jk} I_{ijk}) / (\Sigma_j \Sigma_k R_{ij} A_{jk})$$

For ease of understanding, we suggest expressing BII as a percentage rather than a proportion.

Species-by-species population data for estimating $I_{ijk}$ are seldom available for more than a few species, in a few locations. We used expert judgement to generate a matrix of values of $I_{ijk}$ for southern Africa. Three or more highly experienced specialists in each broad taxonomic group (plants, mammals, birds, reptiles and amphibians) were independently asked to estimate the reduction in the populations of their speciality group caused by a predefined set of land use activities (Table 1). Their estimates were made relative to populations in a large protected area in the same ecosystem type, of which six were defined: forest, savanna, grassland, shrubland, *fynbos* (a South African sclerophyllous thicket) and wetland. To assist in the estimation process, each taxonomic group was divided into five to ten functional types that respond in similar ways to human activities. Across all groups, the functional types were defined primarily by body size, trophic niche and reproductive strategy. $I_{ijk}$ typically assumed values between 0% and 100%, but exceeded 100% in situations where certain activities benefited particular functional types. Estimates of $I_{ijk}$ were aggregated up to the broad taxonomic level by weighting the estimates for each functional type by the number of species in that group in the particular ecosystem type.

A total of 4,650 estimates of $I_{ijk}$ were made, comprising five broad taxonomic groups, each with at least three experts, six ecosystem types, an average of six land use activities and eight functional types. This works out at approximately 300 estimates per expert, a process that took about five hours per interview. Estimating the BII for southern Africa therefore took a few weeks of effort, rather than the decades needed for detailed population surveys. The range of estimates from different experts was used to construct an uncertainty bar around $I_{ijk}$ (Fig. 1). Expert-derived estimates of $I_{ijk}$ were validated against measurements available in the literature[16–26] (Fig. 2). The paucity of field studies, and the fact that the variation between comparable field studies is substantially larger than between the expert estimates, supports the use of expert-based approaches at this time.

Species richness data ($R_{ij}$) is typically available as total species counts per ecosystem type. In this study, species richness data[27] associated with the WWF ecoregions[28] were used. Using such data is equivalent to assuming that every species occurs throughout the extent of the particular ecosystem type. The BII can also be calculated using the potential geographical distributions for individual species, where such data are available.

The area of a particular land use within a specific ecosystem type, $A_{jk}$, is determined by overlaying land use and ecosystem maps. In this study, broad classes of land use were inferred from land cover and land tenure boundaries. We suggest limiting the number of land use classes to below ten, to keep the number of $I_{ijk}$ estimates required manageable. We defined and mapped six levels of land use intensity (Table 1). Where classes derived from different data sources overlapped, the highest impact land use was assigned. The resolution of the land use classification affects the estimation of $I_{ijk}$. In regional-to-global studies, land cover is typically mapped at a resolution of about 1 × 1 km, and an area classified as, for example, 'cultivated' almost always has inclusions of uncultivated land. Experts were instructed to take this into account in making their estimates of $I_{ijk}$.

How finely the taxonomic groups need to be divided into functional types, the broad biomes into particular ecosystem

Table 1 **Classes and data sources used to compile a land use map**

| Land use class | Description | Examples | Data source |
|---|---|---|---|
| Protected | Minimal recent human impact on structure, composition or function of the ecosystem. Biotic populations inferred to be near their potential. | Large protected areas, national, provincial and private nature reserves, 'wilderness' areas. | World Database on Protected Areas[11]. All designated protected areas of IUCN categories I–V. |
| Moderate use | Extractive use of populations and associated disturbance, but not enough to cause continuing or irreversible declines in populations. Processes, communities and populations largely intact. | Forest areas used by indigenous peoples or under sustainable, low-impact forestry; grasslands grazed within their sustainable carrying capacity. | All remaining areas not classified into one of the other five categories. |
| Degraded | Extractive use at a rate exceeding replenishment and widespread disturbance. Often associated with high human population densities and poverty in rural areas. Productive capacity reduced to approximately 60% of 'natural' state. | Clear-cut logging, areas subject to intense harvesting, hunting, fishing or overgrazing, areas invaded by alien vegetation. | All areas falling below 75% (forest, grassland and savanna) or 50% (shrublands) of expected production as estimated by nonlinear regression (Michaelis–Menten function) of maximum annual NDVI on growth days. Degraded areas not estimated for desert, wetland and *fynbos*. |
| Cultivated | Natural land cover replaced by planted crops. Most processes persist, but are significantly disrupted by ploughing and harvesting activities. Residual biodiversity persists in the landscape, mainly in set-asides and in strips between fields (matrix), assumed to constitute approximately 20% of class. | Commercial and subsistence crop agriculture, both irrigated and dryland, including planted pastures and fallow, or recently abandoned cultivated areas. Orchards and vineyards. | SADC Landcover Data set[36], filled with GLC2000 (ref. 37) for Namibia and Botswana. |
| Plantation | Natural land cover permanently replaced by dense plantations of trees. Unplanted areas assumed to constitute approximately 25% of class. | Plantation forestry, typically *Pinus* and *Eucalyptus* species. | SADC Landcover Data set[36]. |
| Urban | Land cover replaced by hard surfaces such as roads and buildings. Dense populations of people. Most ecological processes are highly modified. Remnant semi-natural cover assumed to constitute 10% of class. | Dense human settlements, industrial areas, transport infrastructure, mines and quarries. | Urban extents[38]. |

types, and how many land use activities are defined is largely a pragmatic question. Our experience is that the patterns of impact of a given land use activity is markedly similar between ecosystem types (confirmed by analysis of variance (ANOVA), which showed no significant biome by land use interaction). This suggests that defining one set of impacts per functional type per biome (high-level ecosystem classification) is sufficient, despite the fact that even the relatively coarse-level ecoregion classification we used for species richness[28] has several ecosystem types per biome.

The uncertainty in BII can be calculated from the standard errors of each component in the BII equation. We found that by far the largest error was associated with $I_{ijk}$. We used this error to calculate a 95% confidence interval for BII. We tested the sensitivity of BII to different resolutions and sources of data for $A_{jk}$ and $R_{ij}$ and found it to be small in the southern African case.

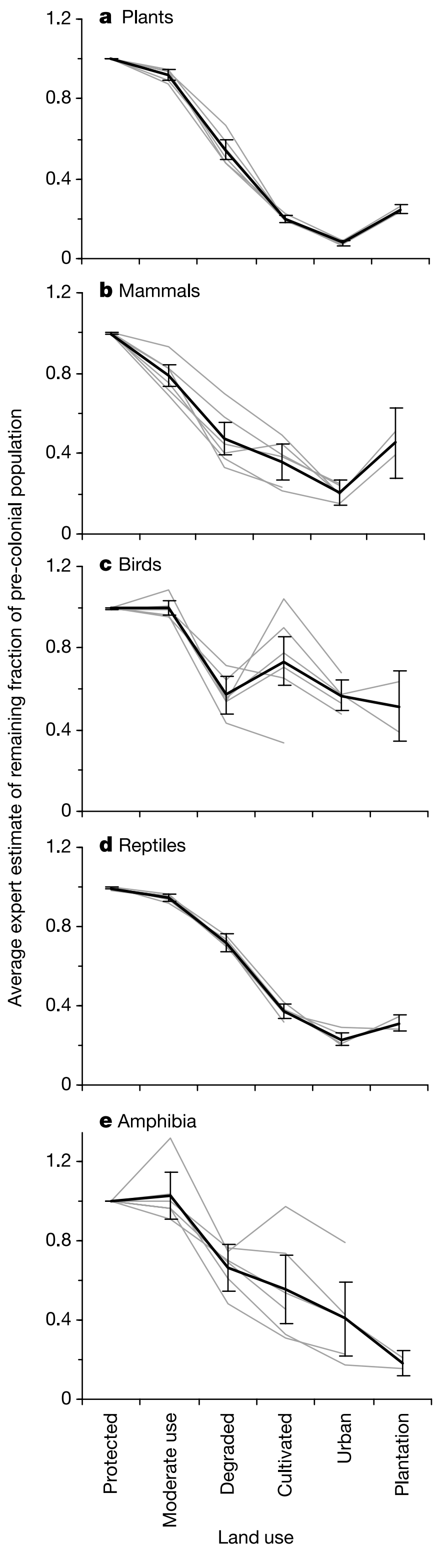


**Figure 1** Average fraction of original populations remaining under a range of land use activities. Data are estimates by three or more taxon experts for each broad taxonomic group (**a**–**e**). The grey lines reflect the average estimates for each of six biomes or ecosystem types and the bold line reflects the estimated impact averaged across all biomes. The error bar reflects the 95% confidence interval around the mean estimate.

## The biodiversity intactness of southern Africa

We applied the BII to the region of southern Africa comprising South Africa, Namibia, Lesotho, Swaziland, Botswana, Zimbabwe and Mozambique. This region was selected because the experts we interviewed felt comfortable extrapolating their experience within this biogeographical domain, but not beyond. Overall, we estimate that 84 $\pm$ 7% of the pre-colonial number of wild organisms persist in present-day southern Africa (Table 2), despite greatly increased human demands on ecosystems over the past 300 years. In contrast, over 99% of the species persist[1], illustrating the insensitivity of indices based on extinction (changes in richness alone). Protected areas currently constitute 8% of the region, moderate use areas 78%, degraded areas 2%, cultivated areas 9%, plantations 0.5% and urban areas 2% (Table 1). The persistence of populations of wild organisms is therefore substantially greater than that indicated by the area under formal conservation, and points to the importance of areas outside of reserves in the preservation of biodiversity. The contemporary distribution of wild organisms in southern Africa is 85% in moderate use areas, 10% in protected areas, and the remaining 5% in cultivated, degraded, urban and plantation areas.

The impact of humans on biodiversity is expressed very selectively. Large-bodied mammal, bird and reptile species that are easy to hunt or harvest are most affected, especially if they are valuable or in direct conflict with human well-being. These species are only a small portion of the total biodiversity. The vast majority of species are affected mainly through loss of habitat to cultivation or urban settlement, both of which make up relatively small fractions of the southern African landscape.

The greatest impact on biodiversity in southern Africa has occurred in the grassland biome, followed closely by *fynbos*. In both cases, the main cause is conversion to cultivated land, followed

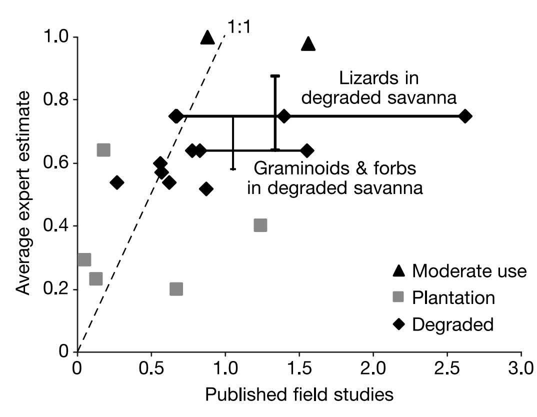


**Figure 2** Comparison of estimates of $I_{ijk}$, the ratio of current populations to reference populations under various land use activities. This ratio was determined by expert estimation versus field studies published in the literature ($n = 19$, $r = 0.47$, $P = 0.04$). For the two functional groups (lizards and graminoids and forbs) where several comparable field studies were available, the range (bar) reported by field studies was substantially larger than the range of estimates between experts.

by urban sprawl and plantation forestry. The arid shrublands and savannas that make up most of the land area of southern Africa are overall less affected. The main cause of biodiversity loss in these systems is land degradation, defined here as land uses that do not alter the cover type, but lead to a persistent loss in ecosystem productivity. Aggregated at a national level, the most densely populated countries in the region, Lesotho and Swaziland, have lower BII scores than the sparsely inhabited countries of Botswana and Namibia (Table 3).

We estimated the rate of change of BII in southern Africa during the decade of the 1990s as an absolute decline of 0.8%. We used data on average rates of land cover change in the region for this calculation. Over the period 1991 to 2000, the relative increase in protected areas was 3.3% (ref. 11), cultivated areas 10.1% (ref. 29), urban areas 5.1% (ref. 29), and plantation forestry 9.2% (ref. 30). Degradation was assumed to have kept pace with the growth of the rural population[29] (12.9%), and the moderate use category was calculated by difference (−1.9%). Assuming that the $I_{ijk}$ values remained constant over the decade, we calculated the overall BII for the year 1990 as 85.2%.

Our results suggest that the policy action with the greatest potential to prevent further loss of biodiversity in southern Africa is to prevent the extensive areas currently under moderate extractive use from becoming degraded. Moderately used land (for example, grazed within stocking norms) has almost the same level of biodiversity as protected areas. Degradation, in the form of overgrazing, forest clearing or dense invasion by alien plants, on average reduces species populations by 40–60%. Cultivation and urbanization have a higher impact per unit land area than degradation, but a much smaller fraction of southern Africa is at risk.

## Scalability and sensitivity of the index

A central feature of the BII is that it can be compared directly within and across scales. This was tested in practice by applying the BII to the three levels of environmental decision-making in South Africa: national ($1.2 \times 10^6\,\mathrm{km}^2$), provincial (average area $1.35 \times 10^5\,\mathrm{km}^2$) and local government (average area $4.6 \times 10^4\,\mathrm{km}^2$), using biome-level species richness data compiled for South Africa[31] in place of the WWF ecoregion richness data[27]. The BII was found to deliver intuitively meaningful results at all scales.

Sensitivity to the resolution of species richness data was examined by comparing the values of BII for mammals obtained using individual mammal species distribution maps[32], to that obtained using aggregated biome-level mammal richness data[31]. For South Africa overall, the BII for mammals was 67.5% using the biome-level data and 66.1% using the individual species data. At the level of individual biomes, the largest absolute difference between the two methods was 0.6%; the largest difference at both the provincial and municipal levels was 3.0%. No correlation was found between the calculated difference and the size of the units of aggregation.

Sensitivity to the type of species richness data was further explored by calculating 'functional biodiversity intactness' (as opposed to compositional biodiversity intactness[7]), in which the aggregated taxon-level $I_{ijk}$ estimates and $R_{ij}$ were derived by weighting by functional group, instead of by species richness. Conceptually, this assigns greater weight to heavily affected, but relatively species-poor groups, such as megaherbivores, and negates the overwhelming effect of plants on the BII. The functional intactness (FII) of South Africa is 81.0%, and therefore slightly higher than the species-weighted BII (79.6%). Mammals showed by far the largest change (−9.7%) in comparison to the species-based estimate, reflecting the high impact of human activities on a few relatively species-poor functional groups. All other taxa showed small changes. These results suggest that in regions lacking species-level richness data, FII could serve as a first approximation for BII: $I_{ijk}$ would be estimated on the basis of a sample of species per functional group, and $R_{ij}$ would be a count of the number of functional groups present in a particular ecosystem type.

Sensitivity of the BII to the source of land use information was explored in two ways. Increasing or decreasing the total area of any single land use category by 5% of its current area resulted in an absolute change of less than 0.3% in the value of BII. The maximum compounded change across all categories was 0.7%. Second, the BII results based on the merged regional land use map (Table 1) were compared with those derived from the South African national land cover[33] and protected areas databases[34]. The BII score derived from the national-level data is 81.2%, compared with a value of 79.6% using the regional land use map.

In both the case of species richness ($R_{ij}$) and land use ($A_{jk}$), the use of reasonable alternative data sources therefore resulted in a difference in the estimated BII (1.4% absolute for species richness and 1.6% for land use) of about twice the magnitude of the estimated decadal change in BII (0.8%), but five times smaller than the uncertainty associated with the impact factors $I_{ijk}$ (6.4%). Thus, for purposes of documenting change, a single type of information source for species richness and land cover should be used for the beginning and end of the period, and the impact factors should be held constant.

## Discussion

The BII is an indicator of the overall state of biodiversity in a given area, synthesizing land use, ecosystem extent, species richness and population abundance data. It is sensitive to the drivers and changes in the populations of species that typify the process of biodiversity loss, and robust to typical variations in data quality.

It is clear that a single index of biodiversity is not sufficient for all purposes. The BII is not intended to highlight individual species that are under threat, and should be used together with indicators such as the IUCN red list of threatened species. Conceptually, the BII is very similar to the natural capital index (NCI), which has been implemented in the Netherlands[35]. However, the method for estimating BII does not require actual population data, and it can therefore complement the NCI in data-sparse regions.

The principal disadvantage of the BII is that it may be insensitive to slow acting, diffuse impacts on biodiversity, for instance the long-term effects of habitat fragmentation, climate change or pollution. However, an indicator does not have to be flawless to be useful. The gross domestic product is a good example: it is a universal standard

Table 2 **BII (%) for southern Africa, per biome and taxonomic group**

| | Area ($\mathrm{km}^2$) | Plants | Mammals | Birds | Reptiles | Amphibia | All taxa |
|---|---|---|---|---|---|---|---|
| Richness* | | 23,420 | 258 | 694 | 363 | 111 | 24,846 |
| Forest | 176,893 | 75.5 | 74.9 | 92.0 | 85.7 | 84.8 | 78.0 |
| Savanna | 2,329,550 | 85.5 | 73.2 | 95.5 | 88.9 | 95.9 | 87.0 |
| Grassland | 408,874 | 72.5 | 55.1 | 90.0 | 75.6 | 81.1 | 74.1 |
| Shrubland | 750,217 | 86.0 | 72.2 | 105.6 | 93.4 | 126.5 | 88.6 |
| *Fynbos* | 78,533 | 75.5 | 78.1 | 91.0 | 76.5 | 79.4 | 76.4 |
| Wetland† | 95,166 | 90.7 | 83.3 | 94.3 | 91.7 | 94.6 | 91.3 |
| All biomes | 3,839,233 | 82.4 | 71.3 | 96.0 | 88.1 | 95.1 | 84.4 |

*Or number of species. Data are for South Africa[31], presented for indicative purposes only.
†Refers only to large wetlands, such as the Okavango Delta.

Table 3 **BII calculated per country for southern Africa**

| Country | BII (%) | 95% CI* |
|---|---|---|
| Botswana | 88.5 | ±7.2 |
| Lesotho | 69.0 | ±7.1 |
| Mozambique | 89.5 | ±8.4 |
| Namibia | 91.4 | ±7.1 |
| Swaziland | 72.1 | ±7.1 |
| South Africa | 79.9 | ±6.5 |
| Zimbabwe | 76.2 | ±7.7 |
| Region | 84.4 | ±7.3 |

*The confidence interval reflects the uncertainty in the expert estimates, $I_{ijk}$.

for comparison, despite its many, well-known problems. Environmental policy could benefit from a 'macro-ecological' indicator of a similar level of generality. □

Received 17 November; accepted 15 December 2004; doi:10.1038/nature03289.


1. WCMC *Global Biodiversity: Earth's Living Resources in the 21st Century* (World Conservation Press, Cambridge, UK, 2000).
2. Anderson, J. A. (ed.) *Towards Gondwana Alive: Promoting Biodiversity and Stemming the Sixth Extinction* (Gondwana Alive Society, Pretoria, South Africa, 2001).
3. UN *World Summit on Sustainable Development: Johannesburg Plan of Implementation* (United Nations, New York, 2002).
4. The Royal Society *Measuring Biodiversity for Conservation* (The Royal Society, London, 2003).
5. Reid, W. V., McNeely, J. A., Tunstall, D. B., Bryant, D. A. & Winograd, M. *Biodiversity Indicators for Policy Makers* (World Resources Institute, Washington DC, 1993).
6. CBD *Proposed Biodiversity Indicators Relevant to the 2010 Target* (Convention on Biological Diversity, Montreal, 2003).
7. Noss, R. F. Indicators for monitoring biodiversity: A hierarchical approach. *Conserv. Biol.* **4,** 355–364 (1990).
8. CBD *Monitoring and Indicators: Designing National-Level Monitoring Programmes and Indicators* (Convention on Biological Diversity, Montreal, 2003).
9. Magurran, A. E. *Measuring Biological Diversity* (Blackwell, Oxford, 2004).
10. Hilton-Taylor, C. (ed.) *2000 IUCN Red List of Threatened Species* (IUCN, Gland, Switzerland and Cambridge, UK, 2000).
11. IUCN & UNEP *World Database on Protected Areas* (IUCN & UNEP, Cambridge, UK, 2003).
12. Loh, J. (ed.) *Living Planet Report 2002* (WWF International, Gland, Switzerland, 2002).
13. Sanderson, E. W. *et al.* The human footprint and the last of the wild. *Bioscience* **52,** 891–904 (2002).
14. Jenkins, M. Prospects for biodiversity. *Science* **302,** 1175–1177 (2003).
15. Houghton, J. T. *et al.* (eds) *Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories* (UK Meteorological Office, Bracknell, 1997).
16. Armstrong, A. J. & van Hensbergen, H. J. Comparison of avifaunas in *Pinus radiata* habitats and indigenous riparian habitat at Jonkershoek, Stellenbosch. *S. Afr. J. Wildl. Res.* **24,** 48–55 (1994).
17. Dean, W. R. J., Anderson, M. D., Milton, S. J. & Anderson, T. A. Avian assemblages in native *Acacia* and alien *Prosopis* drainage line woodland in the Kalahari, South Africa. *J. Arid Environ.* **51,** 1–19 (2002).
18. Fabricius, C., Burger, M. & Hockey, P. A. R. Comparing biodiversity between protected areas and adjacent rangeland in xeric succulent thicket, South Africa: arthropods and reptiles. *J. Appl. Ecol.* **40,** 392–403 (2003).
19. Herremans, M. Conservation status of birds in Botswana in relation to land use. *Biol. Conserv.* **86,** 139–160 (1998).
20. Johnson, R., Ferguson, J. W. H., van Jaarsveld, A. S., Bronner, G. N. & Chimimba, C. T. Delayed responses of small-mammal assemblages subject to afforestation-induced grassland fragmentation. *J. Mamm.* **83,** 290–300 (2002).
21. Joubert, D. F. & Ryan, P. G. Differences in mammal and bird assemblages between commercial and communal rangelands in the Succulent Karoo, South Africa. *J. Arid Environ.* **43,** 287–299 (1999).
22. Meik, J. M., Jeo, R. M., Mendelson, J. R. III & Jenks, K. E. Effects of bush encroachment on an assemblage of diurnal lizard species in central Namibia. *Biol. Conserv.* **106,** 29–36 (2002).
23. Parsons, D. A. B., Shackleton, C. M. & Scholes, R. J. Changes in herbaceous layer condition under contrasting land use systems in the semi-arid lowveld, South Africa. *J. Arid Environ.* **37,** 319–329 (1997).
24. Richardson, D. M. & van Wilgen, B. W. Effects of thirty-five years of afforestation with *Pinus radiata* on the composition of mesic mountain fynbos near Stellenbosch. *S. Afr. J. Bot.* **52,** 309–315 (1986).
25. Shackleton, C. M. Comparison of plant diversity in protected and communal lands in the Bushbuckridge lowveld savanna, South Africa. *Biol. Conserv.* **94,** 273–285 (2000).
26. Smart, R., Whiting, M. J. & Twine, W. Lizards and landscapes: Integrating field surveys and interviews to assess the impact of human disturbance on lizard assemblages and selected reptiles in a savanna in South Africa. *Biol. Conserv.* **122,** 23–31 (2005).
27. WWF-US *Species Database for African Ecoregions* (WWF Conservation Science Programme, Washington DC, 2003).
28. Olson, D. M. *et al.* Terrestrial ecoregions of the world: A new map of life on earth. *Bioscience* **51,** 933–938 (2001).
29. FAO *FAOSTAT Agricultural Database* ⟨http://apps.fao.org⟩ (2004).
30. FAO *Global Forest Resources Assessment 2000* (Food and Agriculture Organisation of the United Nations, Rome, 2001).
31. Le Roux, J. (ed.) *The Biodiversity of South Africa 2002: Indicators, Trends and Human Impacts* (Struik, Cape Town, 2002).
32. Friedmann, Y. & Daly, D. (eds) *Red Data Book of the Mammals of South Africa: a Conservation Assessment* (IUCN Conservation Breeding Specialist Group & Endangered Wildlife Trust, Johannesburg, South Africa, 2004).
33. CSIR *National Land Cover of South Africa* (Council for Scientific and Industrial Research, Division of Water, Environment and Forestry Technology, Pretoria, South Africa, 1996).
34. CSIR *Protected Areas of South Africa* (Department of Environmental Affairs and Tourism, Pretoria, South Africa, 2003).
35. Ten Brink, B. J. E. *et al. Technisch ontwerp Natuurwaarde 1.0 en toepassing in Natuurverkenning 2 [Technical design Natural Capital Index framework and implementation for the Nature Outlook 2]* (RIVM, Bilthoven, The Netherlands, 2002).
36. CSIR *SADC Landcover Dataset* (Council for Scientific and Industrial Research, Division of Water, Environment and Forestry Technology, Pretoria, South Africa, 2002).
37. EC *Global Land Cover 2000 (GLC2000) Database* (European Commission Joint Research Centre, Ispra, Italy, 2003).
38. CIESIN IFPRI & CIAT *Global Rural-Urban Mapping Project (GRUMP): Urban Extents* (CIESIN, Columbia Univ., Palisades, 2004).


**Acknowledgements** We thank the following experts whom we interviewed: G. Alexander, G. Bredenkamp, D. Biggs, B. Branch, V. Carruthers, A. Channing, C. Chimimba, J. du Toit, W. Haacke, J. Harrison, M. Keith, L. Minter, M. Rutherford, W. Tarboton and M. Whiting. R. Dean, C. Geldenhuys, B. Reyers, H. Simons, B. van Wilgen and M. Whiting are thanked for providing links to literature studies for validation of the expert estimates. We thank N. Burgess, J. D'Amico and the Conservation Science Programme at WWF, and G. Kier and the Biomap Working Group at the University of Bonn, Germany, for species richness data. J. Harrison is thanked for data used to derive the richness of each taxon functional type per biome. Thank you to D. van Zyl and T. Newby at the Institute for Soil, Climate and Water in Pretoria for the maximum annual normalized difference vegetation index (NDVI) data that were used to map degradation. B. Reyers at the University of Stellenbosch is thanked for the summation of the mammal distribution grids, which were provided by M. Keith. R.J.S. conceived the original idea for the index. It was then jointly developed further, and the paper was co-written by the authors. R.B. did the BII calculations, GIS application and literature validation.

**Competing interests statement** The authors declare that they have no competing financial interests.

**Correspondence** and requests for materials should be addressed to R.J.S. (bscholes@csir.co.za).