A metric driving local action for global biodiversity conservation

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A metric driving local action for global biodiversity conservation

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Abstract

Difficulties identifying appropriate biodiversity impact metrics remain a major barrier to inclusion of biodiversity considerations in environmentally responsible investment. We propose and analyse a simple science-based local metric: the sum of the proportions that all local populations contribute to their global species abundances, with a correction for species close to extinction. It links mathematically to a widely used global biodiversity indicator, the Living Planet Index, for which we propose an improved formula that directly addresses the known problem of singularities caused by extinctions. We show that, in an ideal market, trade in our metric would lead to near-optimal conservation resource allocation, emphasising support of extinction-threatened species. Barriers to adoption are low, as use of the metric does not require an institutional certification system. Used in conjunction with other metrics, potential areas of application include biodiversity related financial disclosures and voluntary or legislated no net biodiversity loss policies.

Main

The rapid recent growth of markets for responsible investments considering Environmental, Social and Governance (ESG) concerns¹, and the growing attention biodiversity receives in this context^{2,3}, highlight the need for metrics of biodiversity impacts for use by businesses and in financial markets^{2,4,5}.

Requirements on such metrics are different from those for metrics used in the policy context, where some flexibility is desired and a balanced representation of multiple objectives sought⁶. In the business context, metric simplicity is of the essence. Non-expert market participants must understand the impacts their decisions have on a metric⁴. Metric definitions should be rigorous and robust enough to withstand strong financial incentives to game them⁴. Metric values should be comparable across a wide range of business activities⁷. Importantly, biodiversity metrics should support a simple narrative about biodiversity

conservation in communication between business, customers, and the general public⁴. To reduce exposure of market participants to risks from redefinition of metrics as flaws become apparent⁸, metrics should be ‘science based’ in the sense that all choices made in their definition have a scientific rationale. Metrics should be consistent across the spatial scales on which business activity occurs. Ideally, they should also be marketable and the resulting market forces trigger local action that adds up to protection and recovery of global biodiversity.

The Taskforce on Nature-related Financial Disclosures (TNFD), which aims to develop by 2023 a framework, suitable for the financial sector, “for organisations to report and act on evolving nature-related risks”⁹, builds on metric selection criteria developed by UNEP-WCMC’s Aligning Biodiversity Measures for Business^{5,10}, however, with a stronger view on comparability within and across business sectors. These emphasise relevance, transparency, consistency and rigour of metrics, alongside the need for completeness of the metric set.

Here we propose a new metric, ‘Biodiversity Stewardship Credits’ (BSC), designed to address this need with regards to the protection of biodiversity for its ‘intrinsic value’ and as a matter of intergenerational justice. Specifically, it is designed to link local action mathematically to its impact on the Living Planet Index (LPI)^{11,12}, one of the most widely cited global diversity indicators currently in use.

To overcome technical and conceptual problems arising when populations of species entering the LPI approach zero^{12,13}, we begin by introducing a variant of this indicator that avoids these problems. For this variant we demonstrate that, under certain simplifying assumptions, it quantifies long-term extinction risk, thus providing a solid scientific basis for the marketable metric (BSC) that we subsequently derive.

Extinction risk and regularisation of the Living Plant Index

The LPI quantifies the change in the geometric mean of the abundances of species in a given taxonomic or functional group (below: group of species), relative to a baseline year¹². Geometric mean abundance

biodiversity indices of this type stand out by their combined simplicity, favourable statistical properties¹⁴, intuitive accessibility and ecological plausibility¹⁴. Not all species are regularly surveyed at sufficient accuracy to compute the LPI directly, but the index can be estimated as an appropriately weighted geometric average from existing survey time series¹². These estimates are now regularly determined and published under the leadership of the World Wildlife Fund¹¹. The LPI is amongst the most frequently cited biodiversity status indicators in the media, evidencing its wide acceptance by policy makers and the public. It can be interpreted as the typical amount by which species abundances have increased or decreased since the baseline. Because geometric mean abundance is highly sensitive to changes in the abundances of rare species¹³, LPI is sensitive to pressures on rare species even when the total biomass of the group of species represented changes only little.

Mathematically, the LPI is often studied in terms of the sum $\mathcal{L}$ of the natural logarithms (notation ‘ $\log x$ ’) of the global abundances N_i of all S species i in the group of interest:

$$\mathcal{L} = \sum_i^S \log N_i. \quad (1)$$

In place of abundance N_i , other measures of population size can be used, such as population biomass or area covered, whichever approximates total reproductive value¹⁵ well.

If we denote by $\mathcal{L}_0$ the value of $\mathcal{L}$ in the baseline year, current LPI can be computed from $\mathcal{L}$ as

$$\text{LPI} = \exp\left(\frac{\mathcal{L} - \mathcal{L}_0}{S}\right). \quad (2)$$

If one of the S species went extinct since the baseline year, $\mathcal{L}$ attains a value of negative infinity and LPI becomes zero, irrespective of all other species. Practical calculations of the LPI avoid this mathematical singularity by introducing *ad hoc* rules to handle rare or extinct species¹⁶. Satisfactory resolution of this issue is difficult because the interpretation of geometric mean abundance metrics in terms of ecological first

principles remains incomplete. In the following we propose such an interpretation that naturally overcomes the singularity.

It has been shown on long-term time series data, e.g., for trees^{17,18}, fish¹⁸, herbaceous plants¹⁸, and birds¹⁷ that the populations of most species exhibit random walks on the $\log(N)$ axis with mean-squared increments that are largely independent of N , a phenomenon known as ‘environmental stochasticity’ (caused by variability in both the abiotic environment and other, co-occurring species). For a given population, we denote the mean square of this increment during a time interval of length Δt , divided Δt , by v_e . These studies also reveals little if any detectable drift towards smaller population sizes or any preferred value.

On top of environmental stochasticity, the discrete nature of life-history events (e.g. birth, germination, death etc.) generates additional variability in population sizes, known as ‘demographic stochasticity’, for which mean-squared increments in N over a given time interval Δt scale proportionally to N . We denote the corresponding proportionality constant, divided by Δt , by v_d . Demographic stochasticity is widely understood to be negligible compared to environmental stochasticity, except for small population sizes¹⁹. The ratio $N^* = v_d/v_e$ specifies the population size below which demographic stochasticity dominates over environmental stochasticity.

In Box 1 we show for populations driven by environmental and demographic stochasticity that the quantity

$$\mathcal{L}_{\text{reg}} = \sum_i^S \log(1 + N_i/N_i^*) \quad (7)$$

is, for a given group of S species, proportional to the expected number of surviving species after a long time T . In view of this intuitive ecological interpretation, we propose use of the regularised quantity $\mathcal{L}_{\text{reg}}$ in place of $\mathcal{L}$ and correspondingly define the *regularised Living Planet Index* as

$$\text{LPI}_{\text{reg}} = \exp\left(\frac{\mathcal{L}_{\text{reg}} - \mathcal{L}_{\text{reg},0}}{S}\right), \quad (8)$$

Box 1 The regularised Living Planet Index as a measure for extinction risk

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We show that with a small modification the LPI can be understood as quantifying long-term extinction risk. We model the change in the population size of a species over a time interval Δt as

$$N(t + \Delta t) = \exp \left[\xi(t) \sqrt{v_e \Delta t} \right] N(t) + \xi'(t) \sqrt{v_d \Delta t} N(t), \quad (3)$$

where $\xi(t)$ and $\xi'(t)$ denote independent standard normal random numbers. Parameters v_e and v_d represent the strengths of environmental and demographic stochasticity, respectively. Formally taking the limit $\Delta t \rightarrow 0$, standard procedures lead to an approximation of this process by the Ito stochastic differential equation

$$dN = \frac{v_e}{2} N dt + \sqrt{v_d N + v_e N^2} dW_t, \quad (4)$$

where W_t represents a Wiener process (Brownian motion). The first term on the right-hand-side describes drift to larger values. It goes back to the fact that the expectation value of the log-normal distribution $\exp[\xi(t) \sqrt{v_e \Delta t}]$ in Eq. (3) is $\exp(v_e \Delta t / 2)$. If one were to formulate this process in terms of $\log(N)$ rather than N , this term would disappear. However, demographic stochasticity would then generate another drift term instead. To eliminate drift altogether, we express population sizes in terms of $x = \log(1 + N/N^*)$, where $N^* = v_d/v_e$ is the population size at which environmental and demographic stochasticity have the same strength (see Eq. (4)). Note that x becomes zero when a species goes extinct ($N = 0$). Applying Ito's lemma, this change of variables simplifies Eq. (4) to

$$dx = \sqrt{(1 - e^{-x}) v_e} dW_t. \quad (5)$$

That is, x performs a Brownian motion, represented by $dx = \sqrt{v_e} dW_t$, except when x is of the order of one or smaller.

For x approaching zero from above, the factor $1 - e^{-x}$ reduces the magnitude of fluctuations in x , slowing down the random walk. As a result, x can get trapped in the region of low x , and the vicinity of 0 acts similar to an absorbing boundary^{20,21}. This effect is reinforced by the breakdown of the diffusion approximation underlying Eqs. (4) and (5) for small x ²². In reality N and so x reach zero eventually, implying global extinction of that species. We therefore approximate the dynamics of x by simple Brownian motion with an absorbing boundary at $x = 0$ ²³.

Now consider the probability that a species starting from $x = x_0$ will still exist after a time T , i.e., the probability for x to never reach 0 before T . Textbook methods evaluate this to $\text{erf}(x_0 / \sqrt{2T v_e})$ ²⁴, where erf denotes the error function²⁵. For T not too near in the future ($T \gg v_e^{-1} x_0^2$), this simplifies to

$$(\text{probability of survival until } T) = \sqrt{\frac{2}{\pi T v_e}} x_0. \quad (6)$$

That is, for any sufficiently large, fixed observation time T , the current value of $x_0 = x = \log(1 + N/N^*)$ is directly proportional to the probability of species survival. In Supplementary Information (S1) we demonstrate validity of Eq. (6) for a model with discrete population sizes N .

Data suggest that v_e does not usually vary strongly within taxonomic groups^{17,18}. When this is so, $\sum_i^S \log(1 + N_i/N_i^*)$ is, for a given group of S species, proportional to the expected number of species surviving after a long time T .

with $\mathcal{L}_{\text{reg},0}$ denoting the value of $\mathcal{L}_{\text{reg}}$ at the baseline year. These regularised quantities naturally avoid the singularity that arises for extinct species ($N_i = 0$). Yet, LPI and LPI_{reg} are nearly identical when all species populations N_i are much larger than the corresponding N_i^* .

Quantifying local biodiversity impact

With global biodiversity quantified in terms of $\mathcal{L}_{\text{reg}}$, the biodiversity impact of any intervention can be quantified as the resulting change $\Delta \mathcal{L}_{\text{reg}}$ in this metric. If the changes in population sizes ΔN_i resulting from a local impact are just small fractions of population sizes N_i , as will often be the case, $\Delta \mathcal{L}_{\text{reg}}$ is well

104 approximated to linear order in ΔN_i as

$$\Delta \mathcal{L}_{\text{reg}} \approx \sum_i^S \frac{\partial \mathcal{L}}{\partial N_i} \Delta N_i = \sum_i^S \frac{\Delta N_i}{N_i^* + N_i}. \quad (9)$$

105 From Eq. (9) it is clear that $\Delta \mathcal{L}_{\text{reg}}$ weights impacts on globally rare species higher than impacts on common
106 species, thus plausibly providing a measure of pressure on ($\Delta \mathcal{L}_{\text{reg}} < 0$) or relief to ($\Delta \mathcal{L}_{\text{reg}} > 0$) global
107 biodiversity.

108 Any practical determination of $\Delta \mathcal{L}_{\text{reg}}$ would require that the changes in abundances ΔN_i entering Eq. (9)
109 are calculated by comparing measured abundances before and not long after an intervention has occurred.
110 This might raise concerns that the long-term ecological reverberations of an intervention are yet to unfold and
111 so missed by Eq. (9). Since populations are spatially connected through dispersal, local interventions might
112 also impact ecological communities outside the area considered. Indeed, we demonstrated such complex,
113 far-reaching long-term effects of press perturbations in recent simulation of a species-rich metacommunity
114 model²⁶. Remarkably, however, these simulations also revealed that, at least in the case of complete erad-
115 ication of all species on a site (as when building a warehouse in a meadow), the *average* long-term change
116 in the total population of each species across the metacommunity is nearly identical to the size of the local
117 population removed by the intervention. This suggests that use of Eq. (9) with the short term values of ΔN_i
118 provides a good estimate of the expectation value of the long-term impact, even when ecological complexity
119 makes actual long-term impacts less predictable.

120 The Biodiversity Stewardship Credit metric

121 A disadvantage of using impact metrics in the management of biodiversity is that inaccuracies in the quan-
122 tification of positive and negative impacts can accumulate, potentially leading to leakage with systematic,
123 unaccounted for biodiversity loss^{7,27}. Impact metrics are therefore usefully defined in terms of change in a

124 local measures of biodiversity state⁸. However, this approach is not directly applicable here, since LPI and
 125 LPI_{reg} are difficult to apportion over space. Indices permitting easy spatial disaggregation²⁸, on the other
 126 hand, tend to have difficulties incentivising the protection of rare species²⁹.

127 As a proposal to overcome this conundrum, we define, for a given area of land (or water), associated
 128 Biodiversity Stewardship Credits as

$$BSC = \sum_i^S \frac{n_i}{N_i^* + N_i}, \quad (10)$$

129 with S , N_i and N_i^* defined as above and n_i being the local population size of species i that this area sustains
 130 (for migratory species scaled by the proportion of time spent in the area).

131 Comparison of Eqs. (9) and (10) shows that, when all n_i are small relative to N_i , e.g. because the area is
 132 small, BSC quantify the short-term impact that would result from removing all species of the group in that
 133 area. The metric supports a narrative of biodiversity stewardship because it gives high weight to locally
 134 sustained populations of globally rare species.

135 We note the following four properties of BSC that follow directly from Eq. (10):

- 136 1. BSC are additive: the BSC of a combination of non-overlapping areas is simply the sum of their BSC.
- 137 2. BSC represent a strictly finite resources: since n_i is local and N_i global population size, the global
 138 sum of all BSC for a given group of species of size S is close to but never larger than S .
- 139 3. The numerical value of BSC can be interpreted as the effective number of species that the associated
 140 area sustains, while accounting for extinction risks through the N_i^* parameters.
- 141 4. While there will always be some uncertainty in measuring BSC, and standards and certification
 142 systems help reducing this uncertainty, such external authority is not required. Land holders hold the
 143 associated BSC without a need for certification or expert assessment.

144 Computation of BSC is simpler than it might first appear. Of those species that occur at a site, just
145 a few will typically dominate the sum in Eq. (10), either because they dominate the local community or
146 because they are globally rare (Box 2). Compared to values of N_i larger than about 100-1000, N_i^* is generally
147 sufficiently small that it can be disregarded in the evaluation of Eq. (10)^{17,19}. Usually, BSC can therefore
148 be computed by simply estimating the proportions local populations of the dominant species contribute to
149 their global abundance, and adding up these proportions. Box 2 provides an example. To account for the
150 small, non-dominant contributions from other species, generic statistical estimates can be used.

151 The largest hurdle in this calculation might be estimation of the global abundances of species. Since
152 nearly all IUCN Red List assessment criteria³³ involve population size or relevant alternative measures (e.g.
153 number of extant sub-populations), Red List assessments are a valuable source of relevant global abundance
154 data, especially for rare species. Furthermore, a general statistical approach to global abundance estimation
155 has recently been developed and demonstrated for the majority of known birds species³⁰. For many other
156 taxonomic and functional groups survey databases with global coverage are now being built, which should
157 permit replication of this result. For land plants, for example, relative global abundances have been estimated
158 from the Botanical Information and Ecology Network (BIEN, <http://bien.nceas.ucsb.edu/bien/>)³⁴.

159 Where required, N^* values can be computed as $N^* = v_d/v_e$, with v_d and v_e estimated from population
160 time series data as used in calculations of the global LPI^{17,18}, life-history data^{32,35,36}, or combinations
161 thereof³¹.

162 We are unaware of direct alternatives to BSC that would quantify local contributions to extinction risk.
163 Comparable is the recently proposed Species Threat Abatement and Recovery (STAR) metric⁴⁴. However,
164 STAR is neither a measure of biodiversity state nor has it been designed for marketisation. In Supplementary
165 Information (S2) we show that the mathematical structure underlying STAR and BSC is similar, suggesting
166 that these metrics support broadly consistent conservation priorities.

167 Incentive structure generated by BSC

168 We envision that organisations can usefully include regular BSC accounts in environmental impact reports.
 169 The fact that BSC are always positive encourages reporting as complete as logistically possible. The bonuses
 170 that environmentally friendly assets currently attract on markets then imply that BSC on their own have
 171 monetary value. For example, an asset classification scheme based on BSC per hectare of land held would
 172 encourage organisations to acquire high-BSC land and to increase BSC on the land they hold. A market
 173 for BSC could emerge as a result. However, it is not *a priori* clear what the overall effect of the resulting
 174 trade in BSC would be. Could it create perverse incentives running contrary to the original objective of
 175 biodiversity protection? We study this question in the following, assuming a perfect market.

176 Let p be the price of one BSC and $C_\alpha = C_\alpha(n_{\alpha,1}, \dots, n_{\alpha,S})$ the net present value of the costs of sustaining
 177 local population sizes $n_{\alpha,1}, \dots, n_{\alpha,S} \geq 0$ in areas of land held by market participant α , including costs related
 178 to lost opportunities for other uses, minus the net present value of the resulting ecosystem services. The
 179 total value of that area is then $V_\alpha = -C_\alpha(n_1, \dots, n_S) + p \text{BSC}_\alpha$ plus value unrelated to $n_1 \dots n_S$. In a
 180 perfect market, market participant α will take measures to maximise V_α , at which point

$$0 = \frac{\partial V_\alpha}{\partial n_{\alpha,i}} = -\frac{\partial C_\alpha}{\partial n_{\alpha,i}} + \frac{p}{N_i^* + N_i} - \frac{p n_{\alpha,i}}{(N_i^* + N_i)^2} \quad (\text{or } n_{\alpha,i} = 0) \quad (11)$$

181 for all species $1 \leq i \leq S$. The last, negative term in Eq. (11) results because $n_{\alpha,i}$ contributes to total
 182 population size N_i (so $\partial N_i / \partial n_{\alpha,i} = 1$). The $n_{\alpha,i} = 0$ case arises when there are no net benefits at all in
 183 sustaining species i . In such a case one typically finds that even small populations of $n_{\alpha,i} > 0$ do not add
 184 value, implying that $\partial C_\alpha / \partial n_{\alpha,i} > p / (N_i^* + N_i)$ at $n_{\alpha,i} = 0$.

185 Now, consider the problem of minimising the global costs of sustaining $\mathcal{L}_{\text{reg}}$ at a given level by ap-
 186 propriately choosing the population sizes $n_{\alpha,i} \geq 0$ sustained by each land holder α . The solution of this
 187 constrained non-linear optimisation problem satisfied the Karush–Kuhn–Tucker conditions³⁷ derived from

188 the Lagrangian

$$\sum_{\alpha} C_{\alpha} - \sum_{\alpha, i} \mu_{\alpha, i} n_{\alpha, i} - \lambda \mathcal{L}_{\text{reg}}, \quad (12)$$

189 where $\mu_{\alpha, i}$ and λ are the Karush-Kuhn-Tucker multipliers. It leads to the stationarity condition that

$$-\frac{\partial C_{\alpha}}{\partial n_{\alpha, i}} - \mu_{\alpha, i} + \frac{\lambda}{N_i^* + N_i} = 0 \quad (13)$$

190 for all α and i , where either $\mu_{\alpha, i} = 0$ and $n_{\alpha, i} \geq 0$ or $\mu_{\alpha, i} > 0$ and $n_{\alpha, i} = 0$. Comparison of Eqs. (11) and
 191 (13) and of the considerations for $n_{\alpha, i} = 0$ shows that with $p = \lambda$ the two conditions are identical, except
 192 for the term $pn_{\alpha, i}/(N_i^* + N_i)^2$ in Eq. (11). However, this term makes a sizeable contribution compared to
 193 the term $p/(N_i^* + N_i)$ only for when α holds a large proportion of the population of species i (so $n_{\alpha, i}$ and
 194 N_i are of comparable size). Absent such dominant market participants, an ideal BSC market leads to near
 195 optimal allocation of resources to sustain $\mathcal{L}_{\text{reg}}$, and so LPI_{reg} , at a given level. Larger LPI_{reg} correspond to
 196 higher BSC prices p .

197 To address the special case of dominant market participants, we now show the following: Firstly, even
 198 when rare species are dominantly held by a single market participants, BSC still incentivise protection of
 199 these species and growing their populations. Secondly, BSC disincentivise dominance. Consider a market
 200 participant whose land is home to the entire global population of some species i . The participant's BSC
 201 associated with species i are then

$$\text{bsc}_i = \frac{n_i}{N_i^* + n_i}. \quad (14)$$

202 A local population size of $n_i = N_i^*$, for example, earns this market participant 0.5 BSC (Fig. 1). This is
 203 plenty and worth preserving, considering that populations of size N_i^* tend to be comparatively small and

Figure 1: **Accumulation of BSC when rebuilding a species at the brink of extinction.** The solid line gives the BSC associated with this species, Eq. (14). As illustrated by the dashed lines, BSC gained by increasing the species' population by a single individual are largest when it is closest to extinction. However, to achieve full BSC, the species' population must be lifted well above N^* , the population size below which random birth and death events cause large population fluctuations.

that the global sum of available BSC is limited to S . In fact, market incentives to rebuild the population of i are highest at the lowest population size (Fig. 1).

On the second point, note that market participants that sustain a smaller population of a species i on their land benefit more from increasing this population than those holding a larger population. The reason is that the latter incur a higher penalty from reducing the BSC of their current holding by increasing N_i in the denominator Eq. (10). Hence BSC disincentivise dominant holdings of species. We therefore expect that market misalignment due to the difference between conditions Eq. (11) and (13) will be harmless in most cases, implying optimal resource allocation as explained above.

212 The role of BSC in biodiversity conservation

213 To illustrate the role BSC can play in biodiversity conservation and its limitations, we consider a hypothetical
214 use case. Assume that Organisation X has committed to a net gain in BSC on the land it holds. Conversion
215 of large areas of low productivity agricultural land into solar farms by X is estimated to reduce the BSC
216 for plants on this land by a total of 0.1 units. To compensate this biodiversity impact, X works with
217 a conservation NGO, such as the Tree Conservation Fund, to acquire an area of land suitable for the
218 reintroduction of Chinese watermelon trees (*Artocarpus nanchuanensis*), which are critically endangered,
219 with perhaps only 100 individuals remaining in the wild. Over 1000 saplings of *A. nanchuanensis* are
220 planted in the area acquired by X. Assume that, after about five years, when the trees mature, over 200
221 saplings survive. The majority of mature individuals of *A. nanchuanensis* are now on the land owned by
222 X, thus contributing over 0.5 BSC to X's holdings and clearly compensating for the loss of plant BSC from
223 construction of the solar farms. The Tree Conservation Fund estimates the total cost of such measures to
224 be around USD 250,000³⁸, which suggests an opening price for a BSC market of around USD 500,000 per
225 species.

226 Critics might argue that a low-cost, small-scale conservation project cannot compensate the large-scale
227 biodiversity loss due to solar farm construction. Yet, X's use of BSC was as intended by the design of the
228 metric. Capital was directed to a conservation cause where it made a substantial difference at moderate cost.
229 *A. nanchuanensis* was saved from extinction. Overall, global LPI for plants has increased as a result of X's
230 action. Other early adopters would find similar opportunities to acquire BSC at low cost. As the market for
231 BSC matures, organisations committed to net biodiversity gain would seek more efficient schemes to raise
232 BSC (e.g. by restoring entire ecosystems), thus further improving resource allocation to conservation.

233 To address the concern noted above, consider, for example, the Mean Species Abundances (MSA) metric,
234 originally called the Biodiversity Intactness Index²⁸, a well-established metric which is based on similar data.
235 MSA is defined as the average over all native species in the area of interest and over space of the ratio of

236 a species' population to what it would be in an undisturbed ecosystem (capped to be ≤ 1). Assuming the
237 area of the solar farms is much larger than the area of the *A. nanchuanensis* sanctuary, the combined MSA
238 for solar farms and sanctuary would most likely significantly decrease in the scenario above. To achieve net
239 MSA gain, restoration or rewilding of an area comparable to that of the solar farms would likely be required.

240 Thus, MSA puts more emphasis on the size of the area, contrasting BSC's emphasis on species con-
241 servation. Georgina Mace²⁹, in a commentary on MSA, defended this property by arguing that (A) the
242 diversity amongst species as "Irreplaceable, unique units with combinations of traits from long and inde-
243 pendent evolution" with "Intrinsic value" is different from "the components of biodiversity that underlie
244 key functions and benefits of an ecosystem", i.e., what we now call ecosystem services; (B) the latter, so
245 the premise, is more important than the former; and (C) MSA's strength lies in capturing this ecosystem
246 service provision. While we agree with Steps A and C of this argument, we argue that the choice made in
247 Step B is not required for a principled approach to conservation.

248 Building on the international consensus about sustainable development documented in the Brundtland
249 Report³⁹, one can formulate a general principle for the sustainable use of ecosystems:

250 Sustainable use of ecosystems means use [i] to meet the needs of the present [ii] without com-
251 promising the ability of future generations to meet their own needs.

252 Clause ii sets constraints to the implementation of Clause i. Since we cannot know the needs and preferences
253 of future generations, Clause ii effectively constrains the impacts of current uses on ecosystems to those that
254 cannot be undone by the next generation⁴⁰—extinctions must be avoided⁴¹. It calls for recovery of LPI_{reg} ,
255 which can be achieved, e.g., through net BSC gain policies⁴².

256 There is no easy answer to how to implement Clause [i] under this constraint. Needs and preferences
257 differ between societies⁴⁰. Yet, as a general rule, provision of ecosystem services should be considered, and
258 MSA quantifies these in a first approximation²⁹. Accounts of biodiversity impact should therefore capture
259 both aspects: contributions to species conservation (e.g. *via* BSC) and contributions to conservation of

260 ecosystem functioning and services (e.g. *via* MSA)—and these should not be confounded.

261   We have demonstrated that Biodiversity Stewardship Credits have properties suitable for use in bio-
262 diversity-related disclosures in business and financial contexts. The metric can also support voluntary or
263 legislated no-net-loss policies. In all cases, we recommend its use in conjunction with metrics for ecosystem
264 services or natural capital. Many of the attractive properties of BSC reflect that this metric is strictly
265 science based, mathematically linked to the LPI and the species conservation objective. Data requirements
266 of BSC are similar to those of existing comparable metrics. Pilot studies are now underway to develop
267 suitable protocols for application of BSC in practice.

Box 2 Example of BSC calculation

Schoolchildren at Northgate High School and Sixth Form in Ipswich, UK, participating in the Big Schools' Birdwatch organised by the Royal Society for the Protection of Birds (RSPB), counted the birds listed belowⁱ in the school's Wildlife and Wellbeing Garden on 11 December 2020. Taking these counts at face value (despite some methodological limitations) and combining them with published global abundance estimates³⁰, we calculate (table) that the garden holds around 1.3×10^{-6} species worth of Bird-BSC. More than half of BSC come from only two species (pie chart): black headed gulls that were abundant in the garden; and blue tits, due to their relatively low global abundance. To demonstrate that we are justified in disregarding N^* in these calculations, consider dunnoek, the globally rarest species in the sample. From the worst-case literature estimates^{31,32} $v_e = 0.017 \text{ yr}^{-1}$ and $v_d = 0.5 \text{ yr}^{-1}$, we get $N^* \approx 29$, negligible by a wide margin compared to the estimated global abundance $N \approx 9,000,000$.

RSPB common name	n	Scientific name	Glob. abund. N	$\text{bsc} = n/N (\times 10^6)$	Contribution
Black headed gull	34	<i>Chroicocephalus ridibundus</i>	78878122	0.4310	32.7%
Blue tit	4	<i>Cyanistes caeruleus</i>	17115823	0.2337	17.7%
Long tailed tit	3	<i>Aegithalos caudatus</i>	18889928	0.1588	12.1%
Magpie	5	<i>Pica pica</i>	33377656	0.1498	11.4%
Dunnock	1	<i>Prunella modularis</i>	9347093	0.1070	8.1%
Pied wagtail	19	<i>Motacilla alba</i>	353854467	0.0537	4.1%
Wren	1	<i>Troglodytes troglodytes</i>	18682476	0.0535	4.1%
Robin	1	<i>Erithacus rubecula</i>	21261776	0.0470	3.6%
Greenfinch	1	<i>Chloris chloris</i>	24909193	0.0401	3.0%
Carrion crow	1	<i>Corvus corone</i>	44906864	0.0223	1.7%
Woodpigeon	4	<i>Columba palumbus</i>	334511213	0.0120	0.9%
Blackbird	3	<i>Turdus merula</i>	354433630	0.0085	0.6%
Sum				1.3174	100.0%

Contributions of species to total BSC

ⁱ<https://community.rspb.org.uk/getinvolved/learning/b/learning-blog/posts/big-schools-birdwatch-at-northgate-high-school-and-sixth-form>

Supplementary Information

S1 Numerical test of our model for extinction risk

Here we demonstrate that Eq. (6) provides a good estimate of mean species lifetime despite making use of the diffusion (continuum) approximation and disregarding the decline of the strength of fluctuations near $x = 0$.

For this we simulated an asexual population with non-overlapping generations as

$$\lambda(t) \sim \exp\left(v_e^{1/2} \xi(t)\right), \quad (\text{S1a})$$

$$N(t+1) \sim \text{Poisson}(\lambda(t) N(t)), \quad (\text{S1b})$$

with $\xi(t)$ denoting a standard normal random number and $\text{Poisson}(\mu)$ a Poisson-distributed random number with mean μ . The model implies $v_d = 1$.

We set $v_e = (0.15)^2$ and evaluated 26 values for the initial population size $N(0) = N_0$, spaced equally on the log axis from 1 to 10^5 , except for rounding to the nearest integer. For each N_0 , the probability of species survival ($N(T) > 0$) until time $T = 10^4$ was estimated from 10^4 replicates.

As shown in Fig. S1a, extinction probability increases linearly with $\log(N_0)$ only for large N_0 . By contrast, $x_0 = \log(1 + N_0/N^*)$, with $N^* = v_e/v_e = 44.4$, is linearly related to extinction probability for all N_0 considered (because probabilities are ≤ 1 by definition, the relationship would break down for even larger N_0). The slope of the relation is as predicted by Eq. (6).

Figure S1: **Demonstration that x_0 predicts expected species lifetime better than $\log(N_0)$.** Points are simulation results for the individual-based model Eq. (S1) with $v_e = (0.15)^2$ over $T = 10^4$ time steps. The dashed line is the probability of survival as approximated by Eq. (6). Panels (a) and (b) represent the same data on different horizontal scales.

283 S2 Comparison of BSC with the Species Threat Abatement 284 and Recovery (STAR) metric

285 To understand its relation of BSC to the Species Threat Abatement and Recovery (STAR) metric⁴⁴, let us
286 first note that, if the current area of the habitat suitable to a species, AoH⁴⁵, is known and one assumes, for
287 simplicity, even distributions over the habitat, then the ratio $n_i/(N_i^* + N_i)$ entering the definition of BSC,
288 Eq. (10), can, for $N_i \gg N_i^*$, be approximated by the proportion P_i of species i 's habitat contained in the
289 area to be assessed. Hence

$$\text{BSC} \approx \sum_i^S P_i. \quad (\text{S2})$$

290 The definition of STAR has a similar structure, but with two additional factors in the sum. Firstly, the
291 numeric red-list category W_i of species i , an integer ranging from 0 (Least Concern) to 4 (Critically Endan-
292 gered). Secondly, an index $C_{i,t}$ quantifying either the relative contribution of threat t to the extinction risk
293 of that species, or the contribution threat abatement could make to i 's recovery. Hence

$$\text{STAR}_t = \sum_i^S P_i W_i C_{i,t}. \quad (\text{S3})$$

294 The factor $C_{i,t}$ apportioning impacts over threats adds semantics to STAR that BSC do not carry (though
295 the idea might be transferable). The technical difference between STAR and BSC thus lies mainly in the
296 threat-level factor W_i . We have argued above that extinction risk is already captured by $n_i/(N_i^* + N_i)$,
297 approximated by P_i , suggesting that the W_i factor might duplicate this information. Indeed, the extent of
298 occurrence of a species, closely related to AoH⁴⁵, is a central red-listing criterion. Besides, inclusion of the
299 factor W_i implies choices that may not have an obvious justifications: for example, why not use instead a
300 factor W_i^3 or $W_i^{1/3}$? However, since the range of values that W_i can attain is rather narrow, the overall effect

301 this factor has on incentive structure might be small. We therefore expect BSC and STAR to be largely
302 aligned with each other.

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