

# 1 **A global indicator of utilised wildlife populations: regional trends** 2 **and the impact of management**

3

4 **Louise McRae<sup>1</sup>, Robin Freeman<sup>1</sup>, Jonas Geldmann<sup>2,3</sup>, Grace B. Moss<sup>3</sup>, Louise Kjær-Hansen<sup>2</sup> and Neil**

5 **D. Burgess<sup>2,4</sup>**

6 <sup>1</sup> **Institute of Zoology, Zoological Society of London, London, UK**

7 <sup>2</sup> **Center for Macroecology, Evolution and Climate, Globe institute, University of Copenhagen,**

8 **Copenhagen, Denmark**

9 <sup>3</sup> **Conservation Science Group, Department of Zoology, University of Cambridge, Downing St.,**

10 **Cambridge CB2 3EJ, United Kingdom**

11 <sup>4</sup> **UN Environment Programme World Conservation Monitoring Center (UNEP-WCMC), 219**

12 **Huntington Road, Cambridge, UK**

13

## 14 **Abstract**

15 The sustainable use of wildlife is a core aspiration of biodiversity conservation but is the subject of  
16 intense debate in the scientific literature as to how, and whether, species are best used and  
17 managed. While both positive and negative outcomes of sustainable use are known for specific taxa  
18 or local case studies, a global and regional picture of trends in wildlife populations in use is lacking.  
19 We use a global data set of over 11,000 time-series to derive indices of ‘utilised’ and ‘not utilised’  
20 wildlife populations and assess global and regional changes, principally for mammals, birds and  
21 fishes. We also assess whether ‘management’ makes a measurable difference to wildlife population  
22 trends, especially for the utilised species populations. Our results show that wildlife population  
23 trends globally are negative, but with utilised populations tending to decline more rapidly, especially  
24 in Africa and the Americas. Crucially, where utilised populations are managed, using a variety of

25 mechanisms, there is a positive impact on the trend. It is therefore true that use of species can both  
26 be a driver of negative population trends, or a driver of species recovery, with numerous species and  
27 population specific case examples making up these broader trends. This work is relevant to the  
28 evidence base for the IPBES Sustainable Use Assessment, and to the development of indicators of  
29 sustainable use of species under the post-2020 Global Biodiversity Framework being developed  
30 under the Convention on Biological Diversity.

31

## 32 **Keywords**

33 Sustainable use, biodiversity indicators, Convention on Biological Diversity, vertebrates, population  
34 trends, overexploitation, livelihoods, wildlife management

35

## 36 **Introduction**

37 The direct use of wild species is one of the ways in which biodiversity is fundamental to the  
38 livelihoods of people (Hutton and Leader-Williams 2003; Díaz, Demissew et al. 2015; IPBES 2019).  
39 Consequently, any unsustainable impact of anthropogenic activity on species, particularly those that  
40 are important for people's livelihoods or wellbeing, presents a threat not just to conservation but to  
41 human health and development (Pascual, Balvanera et al. 2017). The importance of the sustainable  
42 use of resources has been recognised as central to biodiversity conservation and is embedded in  
43 international bodies and conventions for nature (United Nations 1992; Hickey 1998; IUCN 2000;  
44 United Nations General Assembly 2015; IPBES 2018). However, progress towards achieving the  
45 sustainable use of resources globally remains a challenge. Progress towards Aichi target 4.2 on use  
46 within safe ecological limits was assessed as 'poor' in the final decadal review of the success of the  
47 strategy plan for biodiversity 2010-2020 (Secretariat of the Convention on Biological Diversity 2020)  
48 and impacts of hunting are thought to be increasing (Gallego-Zamorano, Benítez-López et al. 2020).

49 Whilst land use change is the predominant driver of terrestrial biodiversity decline and is expected  
50 to increase in many areas (Kehoe, Romero-Muñoz et al. 2017; IPBES 2019), overexploitation is also a  
51 highly prevalent threat (Joppa, O'Connor et al. 2016) with evidence showing that harvesting,  
52 logging, fishing and hunting often occur at unsustainable levels (IPBES 2019). Together with land use  
53 change, hunting has had negative impacts on species populations, particularly in the tropics where  
54 anthropogenic pressure is currently highest and intensifying (Venter, Sanderson et al. 2016). These  
55 combined pressures have reduced the distribution of terrestrial tropical mammals, with large-bodied  
56 species the most impacted (Gallego-Zamorano, Benítez-López et al. 2020). The effect of hunting,  
57 especially for commercial use, has been implicated in causing overall decline in population  
58 abundance of 97 tropical bird and 254 tropical mammal species (Benítez-López, Alkemade et al.  
59 2017), and the global assessments of 301 terrestrial mammals threatened with extinction list  
60 hunting as a primary threat (Ripple, Abernethy et al. 2016). In the marine realm, the percentage of  
61 commercial fish stocks that are within biologically sustainable levels decreased from 90% to 65.8%  
62 between 1974 and 2017 (FAO 2020), although recent trends suggest that stocks which are  
63 scientifically assessed are now increasing on average and intensively managed stocks are faring  
64 better (Hilborn, Amoroso et al. 2020).

65 The role of wildlife management is also evident in some notable examples on land. The rise of  
66 Community-Based Natural Resource Management over 30 years ago, which may include managing  
67 the use of species in place of more centralised wildlife management policies, has yielded examples  
68 of both economic and ecological benefits in many countries worldwide (Roe, Nelson et al. 2009;  
69 Anderson and Mehta 2013; Cooney, Roe et al. 2018). In regions where utilised species, particularly  
70 mammals, have been heavily impacted over centuries (Ceballos and Ehrlich 2002; Laliberte and  
71 Ripple 2004), conservation action has been implemented to stem unsustainable use and promote  
72 recovery of populations. Arguably there have been some successes with recoveries in many bird and  
73 mammal species in Europe from legal protection and habitat restoration (Deinet, Ieronymidou et al.  
74 2013). In North America, examples of conservation and wildlife management efforts mean that once

75 depleted populations have recovered to a level where they can be sustainably used e.g. North  
76 American bison (Sanderson, Redford et al. 2008).

77 As the examples above suggest, there is both evidence of successful instances of species use and of  
78 negative impacts. We propose that global and large regional views are now needed to understand  
79 how species in use are faring at scale, to measure progress towards policy targets and for identifying  
80 trends in resources that are important for people. Developing a biodiversity indicator based on  
81 species in use could fulfil these aims and also inform global processes such as the IPBES thematic  
82 assessment of sustainable use of wild species (IPBES 2018) and the development of indicators for the  
83 post-2020 global biodiversity framework. To date, synthesis of trends in species in use (herein  
84 ‘utilised species’ or ‘utilised populations’) has largely been done at the species level e.g. (Butchart  
85 2008; Tierney, Almond et al. 2014), which may have overlooked spatially heterogeneity of impacts of  
86 use, as has been identified for commercial harvesting (Di Minin, Brooks et al. 2019). A population-  
87 based approach with information on utilisation at the site-level could provide insight that is not  
88 available at the level of species assessments and would allow small scale information on use, threats  
89 and management to be incorporated into the analysis.

90 To follow this approach, we develop an indicator of utilised vertebrate populations following the  
91 method used to calculate the Living Planet Index (LPI) (Loh, Green et al. 2005; Collen, Loh et al. 2009;  
92 McRae, Deinet et al. 2017), a multi-species indicator based on population trends of vertebrates used  
93 to monitor progress towards international and national biodiversity targets (Butchart, Walpole et al.  
94 2010; Tittensor, Walpole et al. 2014; Green, McRae et al. 2020). We explore differences in these  
95 trends with respect to taxonomic groups and IPBES regions and test the sensitivity of the indicator to  
96 data quality. The Living Planet Database that sits behind the index can be disaggregated  
97 geographically and thematically at the population-level, which enables within-species comparisons  
98 and the identification of correlates predicting trends e.g. (Collen, McRae et al. 2011; Hardesty-  
99 Moore, Deinet et al. 2018). This is the basis for the second part of our analysis to contrast trends in

100 utilised populations with those that are not used, for the complete set of species in the data set and  
101 for only those species with data for both utilised and non-utilised populations (“matched”). Finally,  
102 we explore the role that targeted management has in predicting populations trends in utilised  
103 populations.

## 104 **Results**

### 105 **Geographic, taxonomic and threat data summary**

106 The final data set comprised 11,123 population time-series from 2,944 species, of which 5,811  
107 populations from 1,348 species were coded as utilised and 5,312 populations from 1,996 species  
108 were coded as not utilised (Table S1). In terms of utilised populations, most data were available for  
109 fish (n = 3,233) followed by mammals (n = 2,098), birds (n = 331), reptiles (n = 142) and amphibians  
110 (n = 7). Fish and mammals had more utilised populations than not, while the reverse was true for  
111 birds, reptiles and amphibians (Table S1). Geographically, our sample contained data from all IPBES  
112 regions and from 146 countries (Figure 1; see also Table S2). Both utilised and not-utilised  
113 populations were found in all regions but noticeable clusters of more utilised populations in parts of  
114 Africa, Central Asia and Canada. The largest regional data set was for the Americas. Results for Africa  
115 are based on the smallest data set of the regions; data availability throughout the time-series  
116 dropped after 2012 so the indices were shorter than the other regions, finishing in 2015 and 2013  
117 for terrestrial/freshwater and marine respectively.

118 Threat information was available for 3,195 populations, 1,694 utilised and 1,501 not utilised (Table  
119 S3). There was a difference in the distribution of threats coded between utilised and not utilised  
120 populations, with a greater proportion of threats listed as Overexploitation for utilised populations  
121 (Figure S1). Nearly three-quarters of the Overexploitation threats coded for utilised populations  
122 were as a result of hunting and collecting (Figure S2). Of the utilised populations, 46% had  
123 information available on targeted management and 23% were unmanaged (remainder had no  
124 information; Table S4).

125 **Global indicators for utilised populations show average declines**

126 The index for utilised populations shows a decrease of 69% for terrestrial and freshwater  
127 populations (Figure 2. Index value in 2016: 0.31; range: 0.21 to 0.44) and a decrease of 34% for  
128 marine populations (Figure 2. Index value in 2016: 0.66; range: 0.52 to 0.85), between 1970 and  
129 2016. While the overall trend for utilised populations showed a steep decline, this was not the case  
130 for all individual populations, with 46.3% showing an overall increase, 48.9% showing an overall  
131 decrease and 4.8% were stable in the terrestrial and freshwater index. In the marine index, 53.2% of  
132 utilised populations showed an overall decline, 42.6% an overall increase and 4.2% were stable.

133 We tested the robustness of the indices to time-series length, an important check when using  
134 population trends which vary in sample duration (Wauchope, Amano et al. 2019). We observed  
135 whether similar rates of decline were seen when restricting the data set to different thresholds for  
136 the minimum time-series length in numbers of years. When a more stringent minimum threshold for  
137 time-series length was applied, similar rates of declines were observed for the indices with a  
138 minimum of 5 years and shallower declines reported for the indices with a minimum of 10 years  
139 (Figure S3).

140 **Regional indices show disparate trends**

141 The indices for utilised populations trends since 1970 by IPBES regions show disparate trends, with  
142 the largely tropical regions faring worse than the more temperate (Figure 3). The Africa indices show  
143 the greatest decline among the regions since 1970 in both the terrestrial/freshwater and marine  
144 subsets (Figure 3. Terrestrial/freshwater index value in 2015: 0.07; range 0.03 to 0.16; Marine index  
145 value in 2013: 0.08; range 0.04 to 0.17). The Asia-Pacific index shows a near-continuous decline in  
146 the marine index from 1970 to 2016 and an 83% overall decline (Figure 3. Index value in 2016: 0.17;  
147 range 0.09 to 0.31); there is a lot of uncertainty surrounding the index values in the terrestrial and  
148 freshwater index which fluctuate and ends at a similar baseline value to 1970 (Figure 3. Index value  
149 in 2016: 1.07; range 0.31 to 3.76). The terrestrial/freshwater index for the Americas showed an

150 overall decrease of 67% between 1970 and 2016 (Figure 3. Index value in 2016: 0.33; range 0.19 to  
151 0.58), whereas the marine index fluctuated throughout the time-series and ended at a similar  
152 baseline value to 1970, with no significant overall change (Figure 3. Index value in 2016: 1.07; range  
153 0.78 to 1.45). The marine index for Europe and Central Asia showed a slow increase for most of the  
154 time-series after an initial decline, ending in an overall increase of 41% between 1970 and 2016  
155 (Figure 3. Index value in 2016: 1.41; range 0.95 to 2.13). The terrestrial/freshwater index had a  
156 fluctuating trend for most of the time period but ended with a recent decline (Figure 3. Index value  
157 in 2016: 0.76; range 0.43 to 1.30).

158 The subregional results for utilised populations showed either less negative or more positive trends  
159 compared to the regional index in the case of Southern Africa and North America, both  
160 terrestrial/freshwater and marine, and Central and Western Europe terrestrial/freshwater (Figure  
161 S4).

#### 162 **Utilised populations show more negative trends than non-utilised on average**

163 To explore the effect of utilisation we removed all reptile and amphibian data as these two taxa  
164 contained low number of species and populations in general but particularly those that are in the  
165 utilised category, and a large proportional difference compared to those that are not. This is likely to  
166 make unbalanced comparisons especially when dividing the data set into systems (Table S1). The  
167 remaining taxa – mammals, birds and fish – show a more stable trend for populations that are not  
168 utilised, with index values above the 1970 baseline throughout the time-series, except for a recent  
169 decline, resulting in an overall decrease of 3% over the time period (Figure 4A. Index value in 2016:  
170 0.97; range 0.80 to 1.18). In comparison, the index for utilised populations for these taxa shows a  
171 similar trend to the index including amphibians and reptiles with an overall decline of 50% (Figure  
172 4A. Index value in 2016: 0.50; range 0.41 to 0.62). After 1985, there is no overlap in the confidence  
173 intervals of each index which means they are significantly different.

#### 174 **Utilisation found to be a useful predictor of population trends**

175 Utilisation was a predictor of overall population trends, with utilised populations more likely to be in  
176 decline than non-utilised. Removing utilisation from our models produced significantly worse  
177 predictions of population trends ( $\Delta AIC = -10$ ,  $\chi^2 = 11.835$ ,  $p < 0.01$ ). We found no significant  
178 interaction between utilisation and taxonomic group ( $\Delta AIC = 2$ ,  $\chi^2 = 2.0449$ ,  $p = 0.3597$ ), suggesting  
179 that all taxonomic groups are impacted by utilisation. Using our most comprehensive dataset  
180 (Mammals, Birds, Fish in Terrestrial, Freshwater and Marine systems) suggests that overall and  
181 regardless of utilisation; bird populations are slightly increasing, fish populations are generally  
182 stable, while mammals are in decline (Figure 4B). The length of a population time-series has no clear  
183 positive or negative effect on overall population trends.

184

185 We explored two taxonomic refinements to this dataset. The first removed marine fish which may  
186 represent groups of species that are under particular utilisation pressure and management.  
187 However, after removing marine fish, our results show the same pattern with utilised populations in  
188 more significant decline, (Figure S5). Our second taxonomic refinement explored these effects on  
189 terrestrial and freshwater birds and mammals (excluding all fish). Again, the results are consistent,  
190 with utilization predicting greater significant declines in population abundance (Figure S6).

191 As our classification of utilisation is at the population level, this may result in our models comparing  
192 groups of different species (e.g. all utilised populations may be different species to those that are  
193 not utilised). We therefore also explored a further refinement of the data only including terrestrial  
194 and freshwater bird, mammal and fish species for which we had *both* utilised and non-utilised  
195 populations (2,622 populations of 184 species. Figure S7). The comparison of trends between  
196 utilised and not utilised indices shown in Figure 4A largely holds when the trends for “matched”  
197 species are compared, although there is considerable overlap in confidence intervals until the final  
198 ten years of the time-series (Figure S8A). The mixed model result shows that utilised population  
199 trends are generally less positive than non-utilised, and there is a significant interaction between



200 utilisation and class (Figure S8B). We also note that in this case the random effects were reduced to  
201 *Family* and *Location* (to avoid singular model fits).

## 202 **Managed populations that are utilised are less likely to be declining**

203 For those species where we also record whether the populations are under some form of  
204 management, we find that utilised populations are less likely to be declining when management  
205 actions are in place (Figure S9). Our models suggest that - within our limited data - managed, utilised  
206 populations may be stable, but unmanaged, utilised populations are more likely to be in decline.  
207 However, we note a large taxonomic variation in these population trends, and that many  
208 populations with unknown management status have been removed.

209

## 210 **Discussion**

### 211 **Global and regional trends in utilised populations**

212 We present the first global indicators of trends in utilised vertebrate populations. The indices show  
213 there has been an average decline globally in monitored utilised populations between 1970 and  
214 2016, with a starker trend amongst terrestrial and freshwater populations compared to marine.  
215 Contrasting utilised populations with those that are not utilised, we see a clear difference in the  
216 trend between the groups and this result largely holds when the same species are compared. Mixed  
217 effects models show that utilisation is a significant predictor of a more negative overall population  
218 trend. This result is robust across taxonomic subsets of our data. Whilst populations that are not  
219 utilised may be affected by threat processes such as habitat loss, it appears that the impact of  
220 utilisation in addition to the presence of other threats here is significant, as suggested in other  
221 studies (Benítez-López, Alkemade et al. 2017; Gallego-Zamorano, Benítez-López et al. 2020).  
222 Crucially, we found a positive effect of management on utilised populations, suggesting that this is a  
223 vital factor in ensuring sustainability for wildlife and people's livelihoods.

224 On average, monitored populations that are utilised are in decline, according to the results  
225 presented here. This suggests that, on average, use among these populations may not be  
226 sustainable given that long-term declines are indicative of unsustainable use (Sutherland 2001).  
227 However, the global average masks some interesting variation as just under half of utilised  
228 populations had a stable or increasing trend over the time period. This implies that for some  
229 populations the use may be sustainable (according to population trend only), and that uncovering  
230 explanatory factors behind the trends is crucial. Even within a very limited suite of species for which  
231 we have both utilised and non-utilised populations and so can compare between the two, the  
232 utilised populations exhibited an overall downward trend compared to the non-utilised (Figure S8).  
233 We note, however, that this limited suite does not include species that are *only* utilised or for which  
234 we have no data on their utilisation in our dataset.

235 We found regional differences in trends in utilised populations. As reported in similar regional  
236 analysis of vertebrate populations (McRae, Deinet et al. 2017; WWF 2020), we found more positive  
237 trends in Europe and Central Asia, and even more so for Central and Western Europe (terrestrial and  
238 freshwater) than in other regions. However, comparisons between regions should be interpreted  
239 with care because the baseline of 1970 set for this analysis sets relatively different starting points for  
240 the state of species abundance for each region. The baseline year chosen can be important for  
241 assessing long-term trends (Collins, Böhm et al. 2020), particularly in regions where high human  
242 impact has been prevalent over centuries. In the case of North America and Western Europe, the  
243 baseline of 1970 hides historical declines in species abundance which occurred as land use was  
244 transformed after the Industrial Revolution (Ellis, Klein Goldewijk et al. 2010); post-1970 trends may  
245 therefore show fewer declines as populations stabilise but at lower numbers. This is illustrated by  
246 changes in the intactness of mammal communities drawn from estimated historical and present  
247 distributions, which suggests that mammal intactness is still high in many tropical regions but low in  
248 Europe and some areas of North America (Belote, Faurby et al. 2020), although intactness was not  
249 always directly related to the level of human modification. Another causal factor for positive trends

250 in Central and Western Europe is the increased legal protection for hunted species, habitat  
251 conservation and agricultural land abandonment which has provided an opportunity for wildlife to  
252 rebound in many areas of Europe (Deinet, Ieronymidou et al. 2013).

253 Whilst the trend for North America was less negative than the wider regional trend, there was a  
254 smaller contrast in the terrestrial and freshwater index than expected given the similar context of  
255 baselines and legal protection as in Europe. Over half of the bird and mammal (55%) and freshwater  
256 fish (54%) populations in the North America index showed a declining trend. Interestingly, other  
257 analyses of trends in the United States over a similar time period showed largely positive national  
258 trends for some big game species, although declines were seen in smaller game birds (Flather,  
259 Knowles et al. 2013). One reason for the difference is likely to be the inclusion of more species in this  
260 analysis but also that freshwater fish comprised a large proportion of the utilised data set and this  
261 may be driving the average trend to an overall decline. A decrease in abundance of fish that are part  
262 of the recreational industry is thought to be occurring in Canada (Post, Sullivan et al. 2002); more  
263 broadly it has been suggested that this sector has received less cohesive management strategies  
264 than commercial fisheries (Arlinghaus, Abbott et al. 2019) and that freshwater habitats in particular  
265 can be logistically hard to manage (Post, Sullivan et al. 2002). The decline may also be attributed to  
266 changes in habitat, especially fragmentation of river systems, which is a particular threat to  
267 migratory freshwater fish (Deinet, Scott-Gatty et al. 2020).

268 Data availability was a limitation to assessing trends for Asia Pacific and Africa; for the latter it was  
269 mainly an issue in the later years of the time-series. However, significant declines compared to the  
270 1970 baseline were seen in both Africa indices and in the marine Asia-Pacific index. The extensive  
271 variability in the terrestrial and freshwater data from Asia-Pacific resulted in an inconclusive trend.  
272 The subregional analysis for Southern Africa did produce significantly less negative trends for both  
273 terrestrial and freshwater as well as marine indices, indicating that approaches to sustainable  
274 management of wildlife, including incentivised use of species may have mitigated steeper declines

275 and promoted some populations to stables or recover, although not enough to prevent an overall  
276 decline on average. With the analysis conducted at a regional and subregional scale, the results may  
277 mask the relative differences between countries. For example, positive trends in wildlife have been  
278 shown in Namibia as a result of Community-Based Natural Resource Management (Naidoo, Weaver  
279 et al. 2011), which runs counter to the overall result for Africa and Southern Africa and illustrate that  
280 management strategies that focus on sustainable use can be successful. These regional indices  
281 therefore have the advantage of providing a large-scale indicator as an overview, but the results  
282 don't necessarily represent trends at smaller scales and can hide many local examples of 'best-  
283 practice'. However, the data and method described here is applicable at national and regional level  
284 (McRae, Böhm et al. 2012; WWF-Canada 2020) and could be tailored to assess trends in utilised  
285 species at difference scales provided sufficient data is available.

#### 286 **Results in the context of sustainable use**

287 The Convention on Biological Diversity definition of sustainable use as: "*the use of components of*  
288 *biological diversity in a way and at a rate that does not lead to the long-term decline of biological*  
289 *diversity, thereby maintaining its potential to meet the needs and aspirations of present and future*  
290 *generations*" (United Nations 1992).

291 Our results show a long-term decline, on average, amongst utilised populations globally suggesting  
292 use, overall, is not sustainable. This aligns with broad-scale findings of the impacts of utilisation on  
293 mammals and birds (Ripple, Abernethy et al. 2016; Benítez-López, Alkemade et al. 2017) and of  
294 trends in utilised fishes (FAO 2020; Palomares, Froese et al. 2020). However, the breakdown of  
295 utilised populations showing increasing and declining trends reveals a roughly equal split, inferring  
296 that the use of half of the populations in the data set appears to not jeopardize the long-term  
297 persistence of these population. Sustainable use as a tool is harder to analyse explicitly with this  
298 data set as the implementation of this as a tool was not recorded. However, utilised populations  
299 where the use was incentivised for conservation are likely to also be categorised as 'managed' due

300 to regulations or guidance to manage the use. On this assumption, the impact of management seen  
301 on utilised populations in this analysis indicates that populations intentionally utilised in this way are  
302 likely to have more positive trends.

303 The incorporation of management into this analysis introduces important nuance, suggesting that  
304 less negative or more positive trends are likely if sustainable management of utilised species is  
305 pursued. Management can take many forms and utilisation itself can be a tool for both conservation  
306 and human development, providing incentives for habitat and species conservation to support the  
307 provision of resources for people into the future e.g. (Lichtenstein 2009; Fukuda, Webb et al. 2011).  
308 Harvesting of Saltwater crocodile (*Crocodylus porosus*) eggs for commercial farms in the Northern  
309 territory of Australia has been an incentive for its conservation and an increase in density indices  
310 suggested a full recovery from uncontrolled hunting since 1975 (Fukuda, Webb et al. 2011).

311 Likewise, the establishment of communal conservancies in Namibia were found to provide dual  
312 benefits to the local community from tourism and hunting, especially when these activities occurred  
313 in parallel (Naidoo, Weaver et al. 2016); recoveries in the abundance of species have also been  
314 recorded (Naidoo, Weaver et al. 2011). Given the importance of community involvement in  
315 sustainable management (Persha, Agrawal et al. 2011; Coad, Fa et al. 2019), context is key and there  
316 may not be a single approach to take for sustainably managing wildlife for conservation and  
317 livelihood outcomes (Persha, Agrawal et al. 2011; Anderson and Mehta 2013); however some over-  
318 riding considerations, such as avoiding protectionist approaches and engaging community in  
319 decision-making, have been noted (Cooney, Roe et al. 2018).

320 Sustainable management has arguably had more focus in the marine realm which could offer an  
321 explanation of the more positive trends seen in the marine indices for Central and Western Europe,  
322 and North America. In response to concerns about overfishing and in light of well-documented cases  
323 of fish stock collapse, such as Newfoundland cod (Hutchings and Myers 1994) and northeast Atlantic  
324 herring (Dickey-Collas, Nash et al. 2010), efforts to manage fisheries at national and international

325 levels began to develop in the 1970s and 80s (Sissenwine, Mace et al. 2014). Although commercial  
326 stocks are often reported as in decline globally (Palomares, Froese et al. 2020), there are studies  
327 which highlight positive trends in stocks, particularly those which have been intensively managed to  
328 avoid over-fishing (Hilborn, Amoroso et al. 2020). The nature of the global fishing industry means  
329 that global management is required for many fish stock – in particular those outside national waters.  
330 However for fisheries nearer to coastal communities, management at smaller scales, specifically  
331 community co-management, is advocated as a viable and realistic long-term solution for sustainable  
332 fishing (Gutiérrez, Hilborn et al. 2011). Importantly, this form of management is also likely to be  
333 more equitable. Successful case studies of community co-management have been found from an  
334 assessment across all regions of the world, with the best outcomes determined by attributes of the  
335 community - the presence of community leaders, strong social cohesion – and of the management  
336 approach – community-based protected areas and individual or community quotas (Gutiérrez,  
337 Hilborn et al. 2011).

### 338 **Suitability as an indicator of utilised populations**

339 The use of the LPI data and method as the basis for an indicator for utilised populations has the  
340 advantage of capitalising on available data and a method already established in research and policy  
341 (Collen, Loh et al. 2009; Tittensor, Walpole et al. 2014; McRae, Deinet et al. 2017; Secretariat of the  
342 Convention on Biological Diversity 2020). Population trend data procures advantage over species  
343 level assessments as it incorporates site specific information on utilisation and management which  
344 can vary across a species range and over time. Abundance trends also incorporate a sensitivity  
345 meaning the index can respond quickly to changes in populations (Santini, Belmaker et al. 2017). As  
346 an indicator of populations that are important for human use, this can be a useful early-warning  
347 signal that management intervention needs to be initiated or made more effective to sustain vital  
348 resources.

349 A primary shortcoming of this approach is with respect to the shortage of comprehensive  
350 information for all vertebrate groups and the lack of plant or invertebrate data. The data set behind  
351 the index suffers much of the same biases as found in other data sets and indicators (McRae, Deinet  
352 et al. 2017; Proença, Martin et al. 2017), with data available for well-studied taxa such as birds and  
353 mammals, or those of commercial importance such as fish. Geographic gaps in the data also remain,  
354 particularly in South America and South-east Asia, regions that are hotspots of both wildlife trade  
355 (Scheffers, Oliveira et al. 2019) and of mammals threatened by hunting (Ripple, Abernethy et al.  
356 2016). However, it can be prudent to develop indicators in lieu of comprehensive data, providing  
357 that the gaps in data are clear and biases addressed when feasible (Jones, Collen et al. 2011; McRae,  
358 Deinet et al. 2017).

359 Whilst population trend is one measure of sustainability, there are other factors which are not  
360 considered here and might not be appropriate to aggregate into a global indicator. Changes in  
361 population structure as a result of selective hunting pressure can occur e.g. (Garel, Cugnasse et al.  
362 2007), which may start to occur prior to a population decline being detected. A utilised population  
363 may show altered behaviours e.g. (Ciuti, Muhly et al. 2012) which may not necessarily correlate with  
364 population trends. Finally, this index is not able to demonstrate what is the level of sustainable use  
365 and how far beyond this limit are current levels of pressure – i.e. how much would the current use  
366 need to be reduced to reverse the declines observed. The human dimension of sustainable use,  
367 relating to the needs and benefits of people’s use of wildlife is not factored into this analysis but is a  
368 fundamental aspect of how sustainably species are used (Hutton and Leader-Williams 2003).  
369 Dividing the utilised populations into types of use could help in this regard and incorporating socio-  
370 economic data would be challenging but an interesting consideration to develop this indicator  
371 further. This work also does not address the non-consumptive component of utilisation.  
372 Incorporating trend data for species under this type of use might introduce more positive or stable  
373 trends, on the assumption that non-consumptive use is less likely to directly cause population

374 decline, even though the effects of uses such as tourism could be detrimental to some species e.g.  
375 (Kelly, Pickering et al. 2003; Burgin and Hardiman 2015).

## 376 **Conclusion**

377 The alignment of conservation and human development goals is challenge, particularly when it  
378 comes to the sustainable use of resources (Hutton and Leader-Williams 2003). The results presented  
379 here suggest that whilst the global trend is negative on average for utilised populations, evidence  
380 from a substantial data set of utilised populations suggest that managing the use of wildlife has had  
381 a positive impact on species trends. This is an important finding for the conservation of species  
382 directly of benefit to people. With sustainable use a core component of both the post-2020 Global  
383 Biodiversity Framework and the Sustainable Development Goals, indicators are required to monitor  
384 progress towards the associated targets; the index presented here can address this need.

## 385 **Experimental procedures**

### 386 **Resource availability**

#### 387 *Lead contact*

388 Further information and requests for resources should be directed to and will be fulfilled by the lead  
389 contact, Louise McRae ([louise.mcrae@ioz.ac.uk](mailto:louise.mcrae@ioz.ac.uk)).

#### 390 *Materials availability*

391 This study did not generate new unique materials

#### 392 *Data and code availability*

393 The data used in this paper is stored in the online database at [www.livingplanetindex.org](http://www.livingplanetindex.org). The R  
394 package used for analysis is available here: <https://github.com/Zoological-Society-of-London/r/ipli>

395

### 396 **Collection and coding of data set**



397 Vertebrate population time series data were extracted from the Living Planet Database (WWF/ZSL  
398 2020), a global repository of annual abundance estimates collated primarily from the scientific  
399 literature and online databases (Collen, Loh et al. 2009; McRae, Deinet et al. 2017). The annual  
400 abundance measures were collected using a consistent monitoring method in a given and consistent  
401 location. The time-series vary from 2 to 46 years in terms of length of timeframe and in number of  
402 raw annual data points. Units of abundance were population size estimates, densities or proxies of  
403 abundance, such as nests or breeding pairs (see (McRae, Deinet et al. 2017) for more details).  
404 Alongside the abundance data for each population, several ancillary data fields were extracted to  
405 use for summaries, disaggregation and modelling of the data (Table S6).

406 The use of species can be consumptive - hunting, fishing, harvesting - or non-consumptive – tourism,  
407 cultural experiences, catch and release fishing – and for commercial, subsistence or recreational  
408 purposes (Sustainable Use and Livelihoods Specialist Group 2020). The definition of utilised in the  
409 Living Planet Database, refers only to consumptive use but does not include non-consumptive uses.  
410 If a population is in use as a form of management, it will be tagged as both ‘utilised’ and ‘managed’.  
411 The two categories allow us to differentiate between populations that are utilised and under  
412 management with those that are utilised and unmanaged. Additionally, we consider populations  
413 that are not utilised but are managed for some other purpose e.g. provision of nest boxes for a  
414 species whose nesting habitat has been degraded.

#### 415 **Index calculation**

416 Using the R package *rlpi* (<https://github.com/Zoological-Society-of-London/rlpi>) and following the  
417 Generalised Additive Modelling framework in (Collen, Loh et al. 2009), we calculated global and  
418 regional indices of abundance for populations that were utilised and populations that were not.  
419 IPBES regions were chosen to divide the data sets, as these are commonly used for reporting on  
420 broad scale biodiversity trends. The indices were calculated for different subsets of the data (Table

421 S7). The subset of species in the data set with data for both utilised and non-utilised populations are  
422 referred to as “matched” species.

423 The finer scale subregional analysis was conducted for three subregions – Southern Africa, Central  
424 and Western Europe and North America. Wildlife management in these subregions has arguably  
425 been more widespread so a comparison with the wider regional trends is of interest.

426 The baseline year set for the index was 1970 and it was run until 2016, as data availability decreases  
427 beyond this year due to the publication time lag. Each population trend carried equal weight within  
428 each species and each species trend carried equal weight within each index; we did not incorporate  
429 any additional weighting as has been done for the global LPI (McRae, Deinet et al. 2017). The  
430 confidence intervals were calculated using bootstrap resampling of 10,000 iterations to indicate  
431 variability in the underlying species trends (Collen, Loh et al. 2009).

#### 432 **Mixed models**

433 We considered how total population abundance change ( $T\_lambda$ , *sum of annual rates of change*)  
434 had changed in response to utilisation (*Utilised*) for different taxonomic groups (*Class*: Mammalia,  
435 Aves, Fishes). Time-series length was included to understand if longer population trends tended to  
436 reflect more positive or negative overall change. Taxonomic and site effects were accounted for by  
437 including a random intercept for Family, Binomial (Genus + species) and population location.  
438  $T\_lambda$  values were taken from the *rlpi* package, which generates a matrix of annual rates of  
439 change for each population. The annual rates were summed to give a logged value of total change in  
440 abundance for each population ( $T\_lambda \sim \theta + TS\_length + Utilised + Class +$   
441  $(1|Family/Binomial) + (1|Location)$ ).

442 We also explored how the removal of marine populations and fish population affected this model.  
443 For a subset of these populations we also have information on whether they are subject to some  
444 form of management. We therefore assess a second model structure including Management as an

445 additional explanatory factor ( $T_{\lambda} \sim \theta + TS_{\text{length}} + \text{Management} + \text{Utilised} + \text{Class}$   
446  $+ (1|\text{Family}/\text{Binomial}) + (1|\text{Location})$ ).

## 447 **Supplemental Information**

448 Document S1. Figures S1-S9 and Tables S1-S7

449

## 450 **Acknowledgements**

451 This output has been funded in whole or part by the UK Research and Innovation's Global Challenges  
452 Research Fund, under the Trade, Development and the Environment Hub project (project number  
453 ES/S008160/1). LM and NDB were funded by UNEP-WCMC through the above grant. The authors  
454 would like to kindly thank Dilys Roe for her insightful review of the draft manuscript.

## 455 **Author contributions**

456 Conceptualisation, All authors; Data curation, L.M. and R.F.; Formal Analysis, L.M. and R.F.; Funding  
457 acquisition, N.D.B; Writing – original draft, L.M.; Writing – review & editing, All authors

458

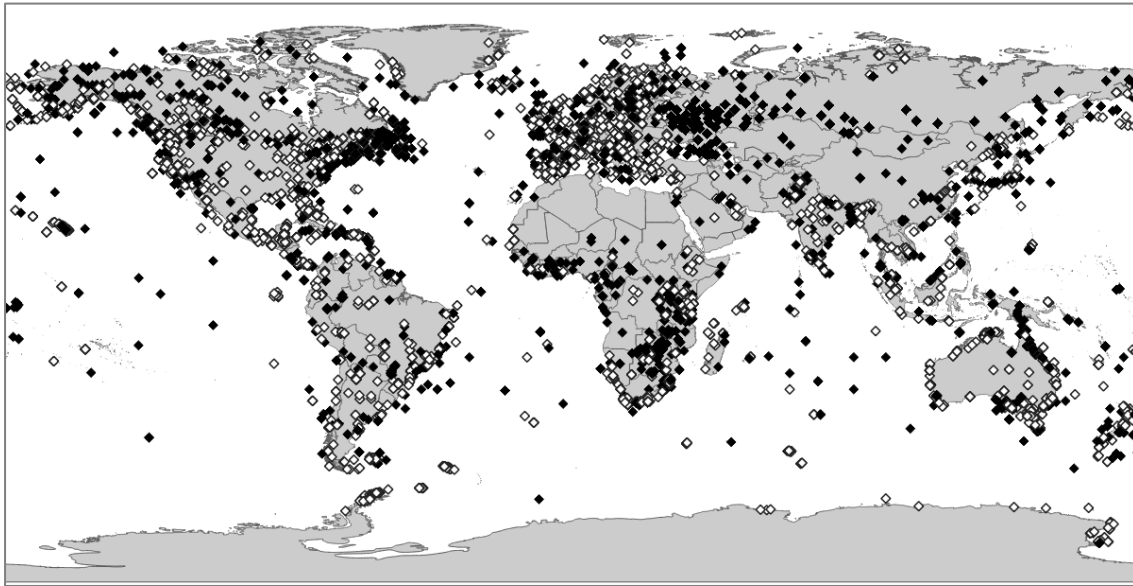
## 459 **Declaration of Interests**

460 The authors declare no competing interests

461

462

463 **Figures**



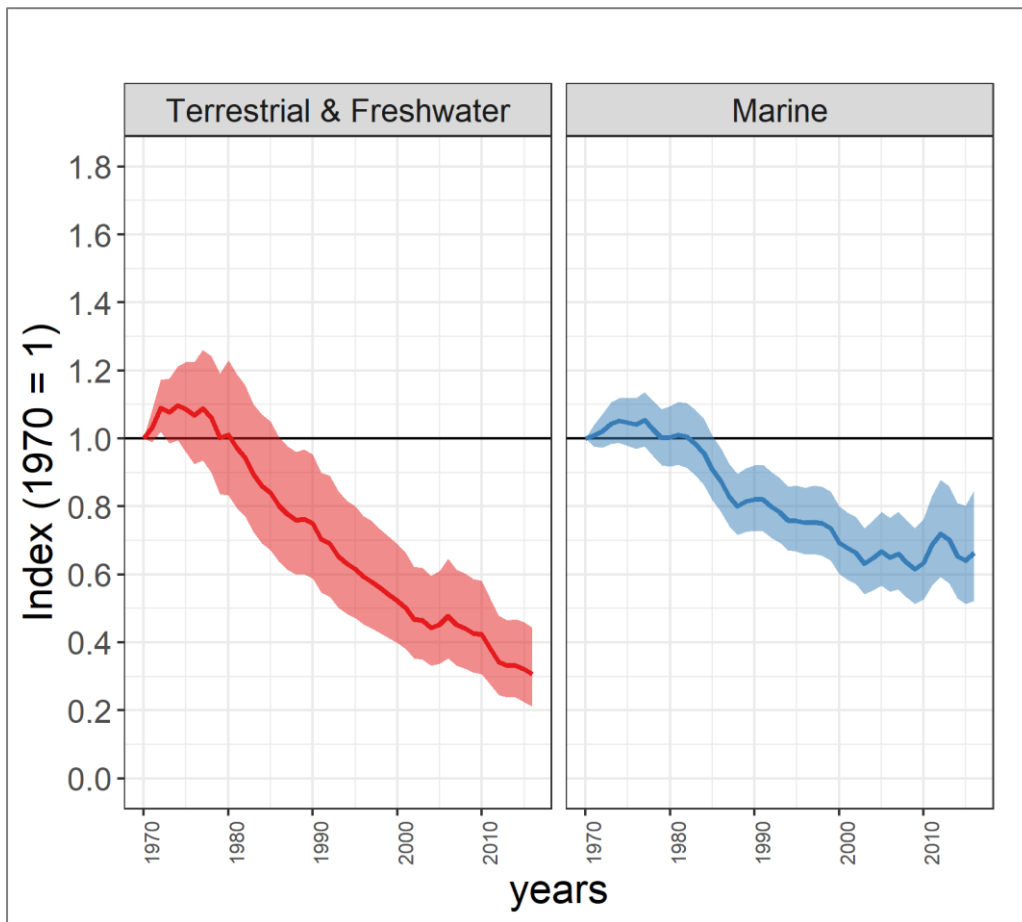
464

465 **Figure 1. Locations of populations used in the analysis.**

466 The point location is shown for the utilised (black diamonds) and non-utilised (white diamonds)  
467 populations used in the analysis. See also Table S2

468

469



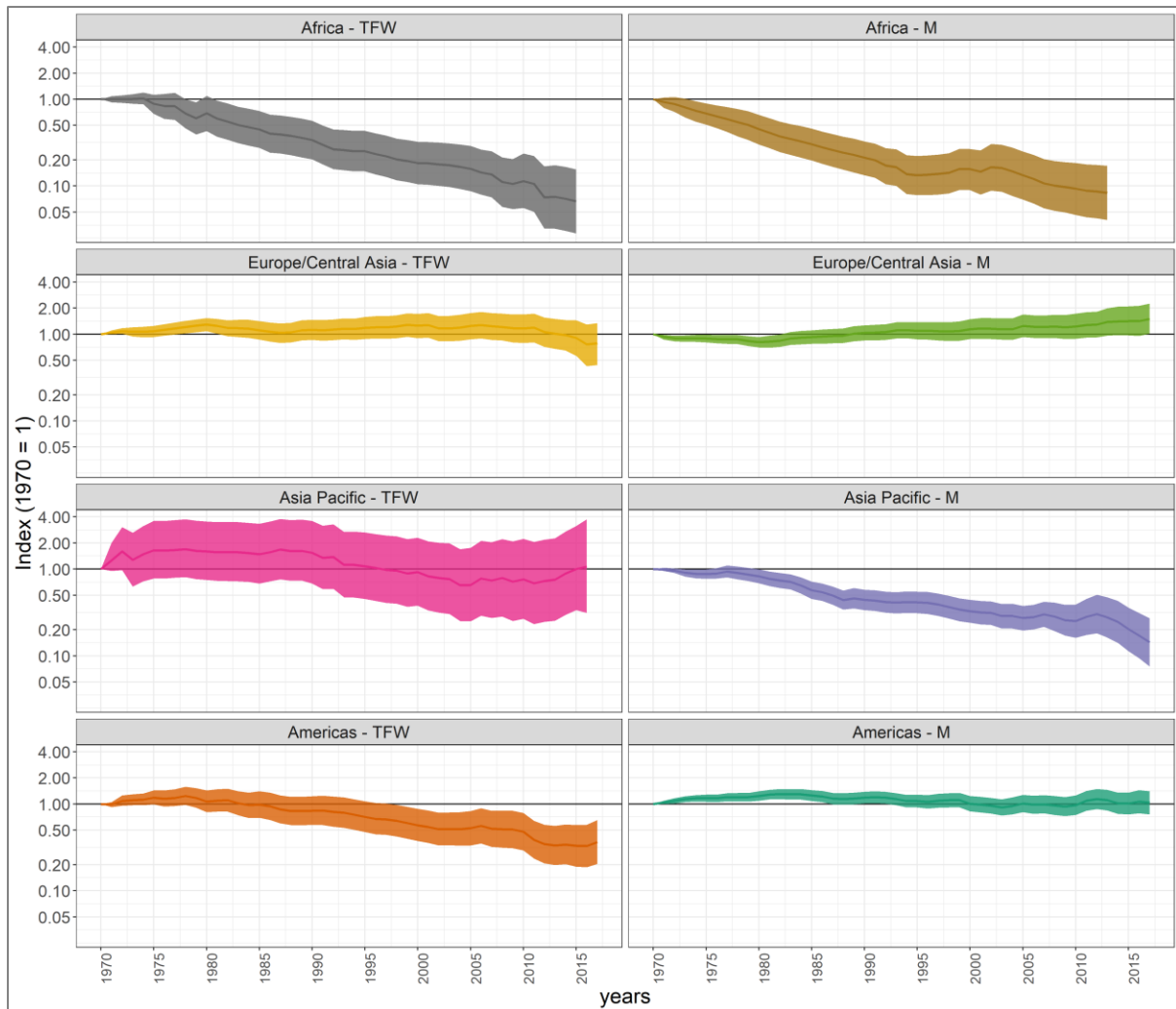
470

471 **Figure 2. Index of utilised populations globally from 1970 to 2016.**

472 Terrestrial and freshwater index: -69%; nspp = 607, npop = 3123. Marine index: -34%; nspp = 761,

473 npop = 2688. See Table S5 for confidence intervals.

474

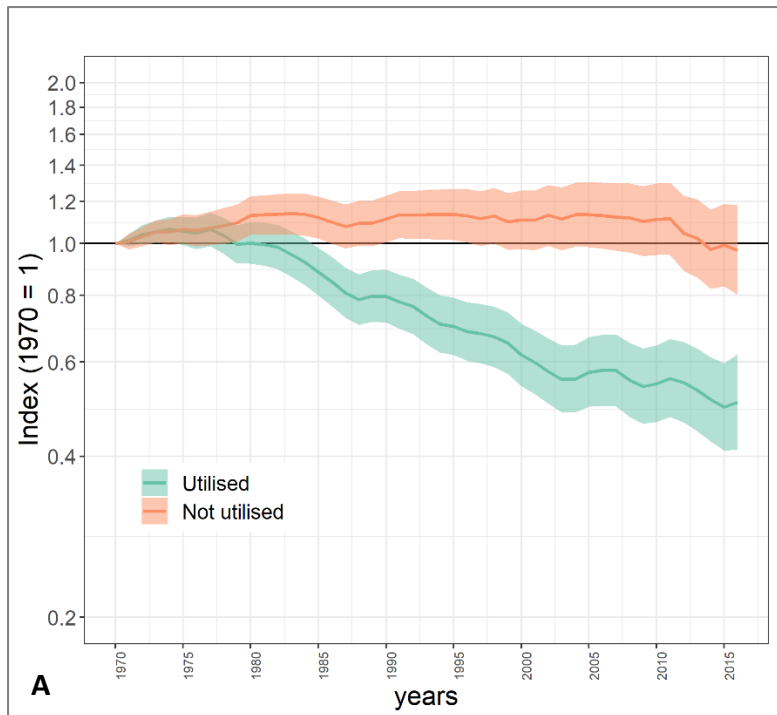


475

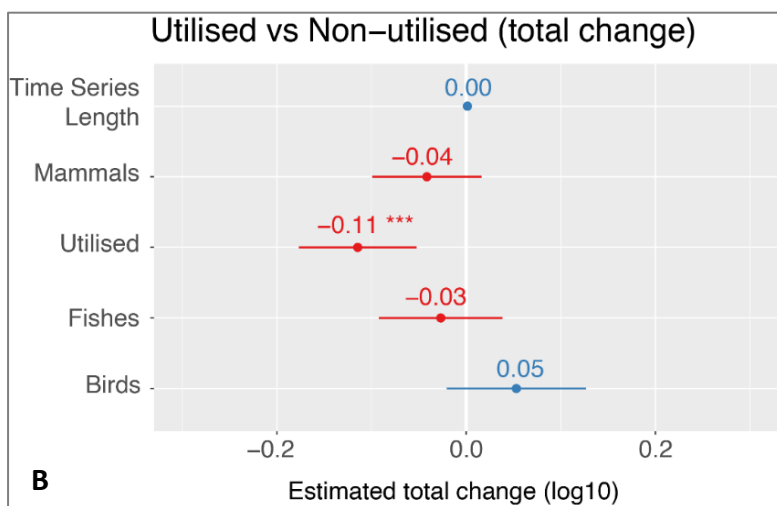
476 **Figure 3. Index of utilised populations for IPBES regions from 1970 to 2016.**

477 Terrestrial and freshwater (TFW) indices: Africa (-93%; nspp = 110, npop = 314), Europe and Central  
478 Asia (-24%; nspp = 124, npop = 1886), Asia Pacific (+7%; nspp = 166, npop = 286), Americas (-67%;  
479 nspp = 239, npop = 637). Marine (M) indices: Africa (-92%; nspp = 77, npop = 132), Europe and  
480 Central Asia (+21%; nspp = 100, npop = 252), Asia Pacific (-83%; nspp = 204, npop = 349), Americas  
481 (+7%; nspp = 465, npop = 1852). See Table S5 for confidence intervals.

482



483



484

485 **Figure 4. Comparison of trends in utilised and non-utilised populations from 1970 to 2016**

486 (A) Index of utilised and non-utilised populations for species of bird, mammal and fish. Between  
487 1970 and 2016, on average, utilised populations had declined by 50% (0.41 - 0.62) and non-utilised  
488 populations had declined by 3% (0.80 - 1.18).

489 (B) Estimated overall total change from the best linear mixed-effect model including Family,  
490 Binomial and location as random effects. Coefficients show the estimated overall change (log10) in  
491 each group. We found no significant interaction between taxonomic group and utilisation, with  
492 utilised populations of any taxa (*Utilised*) significantly more likely to be in decline.

493

494

## 495 **References**

- 496 Anderson, J. and S. Mehta (2013). Global Assessment of Community Based Natural Resource  
497 Management: Addressing the Critical Challenges of the Rural Sector. USAID, Washington, DC,  
498 USA.
- 499 Arlinghaus, R., J. K. Abbott, et al. (2019). "Opinion: Governing the recreational dimension of global  
500 fisheries." Proceedings of the National Academy of Sciences **116**(12): 5209-5213.
- 501 Belote, R. T., S. Faurby, et al. (2020). "Mammal species composition reveals new insights into Earth's  
502 remaining wilderness." Frontiers in Ecology and the Environment **18**(7): 376-383.
- 503 Benítez-López, A., R. Alkemade, et al. (2017). "The impact of hunting on tropical mammal and bird  
504 populations." Science **356**(6334): 180-183.
- 505 Burgin, S. and N. Hardiman (2015). "Effects of non-consumptive wildlife-oriented tourism on marine  
506 species and prospects for their sustainable management." Journal of Environmental  
507 Management **151**: 210-220.
- 508 Butchart, S. H., M. Walpole, et al. (2010). "Global biodiversity: indicators of recent declines." Science  
509 **328**(5982): 1164-1168.
- 510 Butchart, S. H. M. (2008). "Red List Indices to measure the sustainability of species use and impacts  
511 of invasive alien species." Bird Conservation International **18**(S1): S245-S262.
- 512 Ceballos, G. and P. R. Ehrlich (2002). "Mammal Population Losses and the Extinction Crisis." Science  
513 **296**(5569): 904-907.
- 514 Ciuti, S., T. B. Muhly, et al. (2012). "Human selection of elk behavioural traits in a landscape of fear."  
515 Proceedings of the Royal Society B: Biological Sciences **279**(1746): 4407-4416.
- 516 Coad, L., J. E. Fa, et al. (2019). Towards a sustainable, participatory and inclusive wild meat sector.  
517 Bogor, Indonesia: CIFOR.
- 518 Collen, B., J. Loh, et al. (2009). "Monitoring Change in Vertebrate Abundance: the Living Planet  
519 Index." Conservation Biology **23**(2): 317-327.



- 520 Collen, B., L. McRae, et al. (2011). "Predicting how populations decline to extinction." Philosophical  
521 Transactions of the Royal Society B: Biological Sciences **366**(1577): 2577-2586.
- 522 Collins, A. C., M. Böhm, et al. (2020). "Choice of baseline affects historical population trends in  
523 hunted mammals of North America." Biological Conservation **242**: 108421.
- 524 Cooney, R., D. Roe, et al. (2018). Wild life, Wild Livelihoods: Involving Communities in Sustainable  
525 Wildlife Management and Combatting the Illegal Wildlife Trade. United Nations  
526 Environment Programme, Nairobi, Kenya.
- 527 Deinet, S., C. Ieronymidou, et al. (2013). Wildlife comeback in Europe: the recovery of selected  
528 mammal and bird species. Zoological Society of London, UK.
- 529 Deinet, S., K. Scott-Gatty, et al. (2020). The Living Planet Index (LPI) for migratory freshwater fish -  
530 Technical Report.
- 531 Di Minin, E., T. M. Brooks, et al. (2019). "Identifying global centers of unsustainable commercial  
532 harvesting of species." Science Advances **5**(4): eaau2879.
- 533 Díaz, S., S. Demissew, et al. (2015). "The IPBES Conceptual Framework — connecting nature and  
534 people." Current Opinion in Environmental Sustainability **14**: 1-16.
- 535 Dickey-Collas, M., R. D. M. Nash, et al. (2010). "Lessons learned from stock collapse and recovery of  
536 North Sea herring: a review." ICES Journal of Marine Science **67**(9): 1875-1886.
- 537 Ellis, E. C., K. Klein Goldewijk, et al. (2010). "Anthropogenic transformation of the biomes, 1700 to  
538 2000." Global Ecology and Biogeography **19**(5): 589-606.
- 539 FAO (2020). The State of World Fisheries and Aquaculture 2020. Sustainability in action. Rome.
- 540 Flather, C. H., M. S. Knowles, et al. (2013). Wildlife population and harvest trends in the United  
541 States: A technical document supporting the Forest Service 2010 RPA Assessment, Fort  
542 Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research  
543 Station. **Gen. Tech. Rep. RMRS-GTR-296**: 94 p.

- 544 Fukuda, Y., G. Webb, et al. (2011). "Recovery of saltwater crocodiles following unregulated hunting  
545 in tidal rivers of the Northern Territory, Australia." The Journal of Wildlife Management  
546 **75**(6): 1253-1266.
- 547 Gallego-Zamorano, J., A. Benítez-López, et al. (2020). "Combined effects of land use and hunting on  
548 distributions of tropical mammals." Conservation Biology **34**(5): 1271-1280.
- 549 Garel, M., J.-M. Cugnasse, et al. (2007). "Selective harvesting and habitat loss produce long-term life  
550 history changes in a mouflon population." Ecological Applications **17**(6): 1607-1618.
- 551 Green, E. J., L. McRae, et al. (2020). "Below the canopy: global trends in forest vertebrate  
552 populations and their drivers." Proceedings of the Royal Society B: Biological Sciences  
553 **287**(1928): 20200533.
- 554 Gutiérrez, N. L., R. Hilborn, et al. (2011). "Leadership, social capital and incentives promote  
555 successful fisheries." Nature **470**(7334): 386-389.
- 556 Hardesty-Moore, M., S. Deinet, et al. (2018). "Migration in the Anthropocene: how collective  
557 navigation, environmental system and taxonomy shape the vulnerability of migratory  
558 species." Philosophical Transactions of the Royal Society B: Biological Sciences **373**(1746):  
559 20170017.
- 560 Hickey, M. J. (1998). "Acceptance of Sustainable Use Within the CITES Community." Vt. L. Rev. **23**:  
561 861.
- 562 Hilborn, R., R. O. Amoroso, et al. (2020). "Effective fisheries management instrumental in improving  
563 fish stock status." Proceedings of the National Academy of Sciences **117**(4): 2218-2224.
- 564 Hutchings, J. A. and R. A. Myers (1994). "What Can Be Learned from the Collapse of a Renewable  
565 Resource? Atlantic Cod, *Gadus morhua*, of Newfoundland and Labrador." Canadian Journal  
566 of Fisheries and Aquatic Sciences **51**(9): 2126-2146.
- 567 Hutton, J. M. and N. Leader-Williams (2003). "Sustainable use and incentive-driven conservation:  
568 realigning human and conservation interests." Oryx **37**(2): 215-226.

- 569 IPBES (2015). Report of the Plenary of the Intergovernmental Science-Policy Platform on Biodiversity  
570 and Ecosystem Services on the work of its third session. Plenary of the Intergovernmental  
571 Science-Policy Platform on Biodiversity and Ecosystem Services. Third session, Bonn,  
572 Germany.
- 573 IPBES (2018). Report of the Plenary of the Intergovernmental Science-Policy Platform on Biodiversity  
574 and Ecosystem Services on the work of its sixth session. Plenary of the Intergovernmental  
575 Science-Policy Platform on Biodiversity and Ecosystem Services. Sixth session, Medellin,  
576 Colombia. **IPBES/6/INF/8**.
- 577 IPBES (2019). Summary for policymakers of the global assessment report on biodiversity and  
578 ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and  
579 Ecosystem Services. J. S. S. Díaz, E. S. Brondízio E.S., H. T. Ngo, M. Guèze, J. Agard, A. Arneth,  
580 P. Balvanera, K. A. Brauman, S. H. M. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S.  
581 M. Subramanian, G. F. Midgley, P. Miloslavich, Z. Molnár, D. Obura, A. Pfaff, S. Polasky, A.  
582 Purvis, J. Razzaque, B. Reyers, R. Roy Chowdhury, Y. J. Shin, I. J. Visseren-Hamakers, K. J.  
583 Willis, and C. N. Zayas (eds.). IPBES secretariat, Bonn, Germany: 56 pages.
- 584 IUCN (2000). The IUCN Policy Statement on Sustainable Use of Wild Living Resources. Adopted at the  
585 IUCN World Conservation Congress, Amman, Jordan, October.
- 586 Jones, J. P., B. Collen, et al. (2011). "The why, what, and how of global biodiversity indicators beyond  
587 the 2010 target." Conservation Biology **25**(3): 450-457.
- 588 Joppa, L. N., B. O'Connor, et al. (2016). "Filling in biodiversity threat gaps." Science **352**(6284): 416-  
589 418.
- 590 Kehoe, L., A. Romero-Muñoz, et al. (2017). "Biodiversity at risk under future cropland expansion and  
591 intensification." Nature Ecology & Evolution **1**(8): 1129-1135.
- 592 Kelly, C. L., C. M. Pickering, et al. (2003). "Impacts of tourism on threatened plant taxa and  
593 communities in Australia." Ecological Management & Restoration **4**(1): 37-44.

- 594 Laliberte, A. S. and W. J. Ripple (2004). "Range Contractions of North American Carnivores and  
595 Ungulates." BioScience **54**(2): 123-138.
- 596 Lichtenstein, G. (2009). "Vicuña conservation and poverty alleviation? Andean communities and  
597 international fibre markets." International journal of the commons **4**(1).
- 598 Loh, J., R. E. Green, et al. (2005). "The Living Planet Index: using species population time series to  
599 track trends in biodiversity." Philosophical Transactions of the Royal Society B: Biological  
600 Sciences **360**(1454): 289-295.
- 601 McRae, L., M. Böhm, et al. (2012). "The Arctic Species Trend Index: using vertebrate population  
602 trends to monitor the health of a rapidly changing ecosystem." Biodiversity **13**(3-4): 144-156.
- 603 McRae, L., S. Deinet, et al. (2017). "The Diversity-Weighted Living Planet Index: Controlling for  
604 Taxonomic Bias in a Global Biodiversity Indicator." PLOS ONE **12**(1): e0169156.
- 605 Naidoo, R., L. C. Weaver, et al. (2011). "Namibia's community-based natural resource management  
606 programme: an unrecognized payments for ecosystem services scheme." Environmental  
607 Conservation **38**(4): 445-453.
- 608 Naidoo, R., L. C. Weaver, et al. (2016). "Complementary benefits of tourism and hunting to  
609 communal conservancies in Namibia." Conservation Biology **30**(3): 628-638.
- 610 Palomares, M. L. D., R. Froese, et al. (2020). "Fishery biomass trends of exploited fish populations in  
611 marine ecoregions, climatic zones and ocean basins." Estuarine, Coastal and Shelf Science  
612 **243**: 106896.
- 613 Pascual, U., P. Balvanera, et al. (2017). "Valuing nature's contributions to people: the IPBES  
614 approach." Current Opinion in Environmental Sustainability **26-27**: 7-16.
- 615 Persha, L., A. Agrawal, et al. (2011). "Social and Ecological Synergy: Local Rulemaking, Forest  
616 Livelihoods, and Biodiversity Conservation." Science **331**(6024): 1606-1608.
- 617 Post, J. R., M. Sullivan, et al. (2002). "Canada's Recreational Fisheries: The Invisible Collapse?"  
618 Fisheries **27**(1): 6-17.

- 619 Proença, V., L. J. Martin, et al. (2017). "Global biodiversity monitoring: From data sources to Essential  
620 Biodiversity Variables." Biological Conservation **213**: 256-263.
- 621 Ripple, W. J., K. Abernethy, et al. (2016). "Bushmeat hunting and extinction risk to the world's  
622 mammals." Royal Society Open Science **3**(10): 160498.
- 623 Roe, D., F. Nelson, et al. (2009). Community management of natural resources in Africa: Impacts,  
624 experiences and future directions. Natural Resource Issues No. 18.
- 625 Sanderson, E. W., K. H. Redford, et al. (2008). "The Ecological Future of the North American Bison:  
626 Conceiving Long-Term, Large-Scale Conservation of Wildlife." Conservation Biology **22**(2):  
627 252-266.
- 628 Santini, L., J. Belmaker, et al. (2017). "Assessing the suitability of diversity metrics to detect  
629 biodiversity change." Biological Conservation **213**: 341-350.
- 630 Scheffers, B. R., B. F. Oliveira, et al. (2019). "Global wildlife trade across the tree of life." Science  
631 **366**(6461): 71-76.
- 632 Secretariat of the Convention on Biological Diversity (2020). Global Biodiversity Outlook 5. Montreal.
- 633 Sissenwine, M. M., P. M. Mace, et al. (2014). "Preventing overfishing: evolving approaches and  
634 emerging challenges." ICES Journal of Marine Science **71**(2): 153-156.
- 635 Sustainable Use and Livelihoods Specialist Group (2020). Sustainable Use of Wild Species. A critical  
636 element of conservation and rural livelihoods. [https://iucnsuli.org/wp-](https://iucnsuli.org/wp-content/uploads/2020/03/webSULIbrochure.pdf)  
637 [content/uploads/2020/03/webSULIbrochure.pdf](https://iucnsuli.org/wp-content/uploads/2020/03/webSULIbrochure.pdf).
- 638 Sutherland, W. J. (2001). "Sustainable exploitation: a review of principles and methods." Wildlife  
639 Biology **7**(3): 131-140, 110.
- 640 Tierney, M., R. Almond, et al. (2014). "Use it or lose it: measuring trends in wild species subject to  
641 substantial use." Oryx **48**(3): 420-429.
- 642 Tittensor, D. P., M. Harfoot, et al. (2020). "Evaluating the relationships between the legal and illegal  
643 international wildlife trades." Conservation Letters **13**(5): e12724.

644 Tittensor, D. P., M. Walpole, et al. (2014). "A mid-term analysis of progress toward international  
645 biodiversity targets." Science **346**(6206): 241-244.

646 United Nations (1992). Convention on Biological Diversity. [https://www.cbd.int/doc/legal/cbd-  
en.pdf](https://www.cbd.int/doc/legal/cbd-<br/>647 en.pdf).

648 United Nations General Assembly (2015). Transforming our world: the 2030 Agenda for Sustainable  
649 Development. <https://sustainabledevelopment.un.org/post2015/transformingourworld>.

650 Venter, O., E. W. Sanderson, et al. (2016). "Global terrestrial Human Footprint maps for 1993 and  
651 2009." Scientific Data **3**(1): 160067.

652 Wauchope, H. S., T. Amano, et al. (2019). "When can we trust population trends? A method for  
653 quantifying the effects of sampling interval and duration." Methods in Ecology and Evolution  
654 **10**(12): 2067-2078.

655 WWF-Canada (2020). Living Planet Report Canada: Wildlife At Risk. J. Currie, J. Snider and E. Giles.  
656 Toronto, Canada, World Wildlife Fund Canada.

657 WWF (2020). Living Planet Report - 2020: Bending the curve of biodiversity loss. R. E. A. Almond,  
658 Grooten M. and Petersen, T. (Eds). WWF, Gland, Switzerland.

659 WWF/ZSL (2020). Living Planet Database. [www.livingplanetindex.org](http://www.livingplanetindex.org).

660

661