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Review



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Accounting for functionality in the identification of global conservation priorities: promises and pitfalls

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Whereas preventing species extinctions remains a central objective of conservation efforts, it must be complemented by the long-term preservation of functional ecosystems and of the benefits humans derive from them. Here, I review recent approaches that explicitly account for functionality in setting large-scale conservation priorities, discussing their promise while highlighting challenges and pitfalls. Crossing data on species' distributions and ecological traits has enabled the mapping of global patterns of functional diversity and functional rarity and the identification of species that stand out for their functional distinctiveness. However, the priorities identified through these general indices do not directly address ecosystem functionality, instead, they are methods for ensuring the representation of individual functional traits as intrinsically valuable biodiversity elements. Three other approaches integrate functionality into large-scale priorities by taking into account the specific context of each ecosystem, site or species: the International Union for Conservation of Nature's Red List of Ecosystems, Key Biodiversity Areas and the Green Status of Species. Currently at various stages of development, testing and implementation, these approaches are playing an increasingly important role in the definition, implementation and monitoring of global- and national-scale conservation strategies to ensure the long-term persistence of ecosystem functions and associated ecosystem services.

This article is part of the discussion meeting issue 'Bending the curve towards nature recovery: building on Georgina Mace's legacy for a biodiverse future'.

1. Introduction

The last word in ignorance is the man who says of an animal or plant: 'what good is it?' If the land mechanism as a whole is good, then every part is good, whether we understand it or not. If the biota, in the course of aeons, has built something we like but do not understand, then who but a fool would discard seemingly useless parts? To keep every cog and wheel is the first precaution of intelligent tinkering.

Aldo Leopold, 1949, *Round River* [1, pg. 145]

The famous quote by Aldo Leopold [1] both inspires [2,3] and neatly encapsulates [4,5] the conservation paradigm of preventing species' extinctions. While evoking intrinsic ('good') and aesthetical values of nature ('something we like'), it sets forth as the main argument for species conservation the possibility that each species may have a key and irreplaceable role in ecosystem function ('the land mechanism'), even if we do not know it yet. In doing so, Leopold was arguing for what were subsequently termed 'option value' (the idea that the diversity of life is valuable because we cannot predict

which elements will provide uses or benefits in the future) and 'insurance value' (diversity is valuable because we cannot predict which elements will help maintain system stability or integrity in the face of future changes) [6].

At the time Leopold wrote these words, evidence was accumulating that non-trivial numbers of species had already disappeared or were at risk of doing so, as reflected in the first published lists of 'extinct and vanishing animals' [7,8], precursors to The International Union for Conservation of Nature (IUCN) Red List of Threatened Species [9]. Today, over 150 000 species have been assessed through the Red List, including more than 42 000 classified as being at risk of extinction, and nearly 1000 classified as 'Extinct' or as 'Extinct in the Wild' [10].

Leopold's intuition that species are important to the function of 'the land mechanism' has been vindicated by a substantial body of work showing a generally positive relationship between species diversity and both ecosystem functions (such as biomass production and nutrient cycling) and ecosystem resilience [11–13]. Still, this same field of research also revealed that ecosystem functioning is better explained by measures of community diversity that reflect the diversity of species' functional traits than by measures that treat all species as functionally equivalent [11–16]. For example, Gagic and colleagues [14] analysed data from eight field studies to quantify the effects of the diversity of bees on pollination, of carabid beetles on the biocontrol of pests and of weeds, of earthworms on bioturbation, of nematodes on nutrient cycling and of dung beetles on dung removal and seed burial. They found that indices based on the diversity of species traits consistently provided greater explanatory power of ecosystem processes than either species' richness or species' abundance.

Findings such as these sparked concerns that species-focused conservation efforts are not an efficient or sufficient approach for ensuring the long-term persistence of ecosystem functions and of the benefits humans derive from them, leading to calls for conservation strategies that target ecosystem functionality directly [12,13,15,17]. These concerns are now at the forefront of the international biodiversity policy agenda, with the Kunming–Montreal Global Biodiversity Framework (GBF) introducing goals and targets that explicitly address the conservation of ecosystem functions and services [18,19]. Adopted in 2022 by the Parties to the Convention on Biological Diversity, the GBF is accompanied by a monitoring framework, comprising a set of indicators for tracking progress towards these goals and targets, including headline indicators (recommended for national, regional and global monitoring) and more detailed component and complementary indicators [20].

Here, I review four recent approaches that place functionality at the heart of large-scale biodiversity conservation strategies. They are: conservation priorities based on metrics of functional diversity or rarity; the incorporation of functionality in assessments of ecosystems' risk of collapse through the IUCN Red List of Ecosystems; the identification of Key Biodiversity Areas based on functionality criteria; and assessments of species' recovery status under the IUCN Green Status of Species. I argue that despite its growing popularity in the scientific literature, the first of these approaches does not directly address ecosystem functionality, focusing instead on ensuring the conservation of individual functional traits as intrinsically valuable biodiversity elements. The three other approaches, currently at various stages of development, testing and implementation, hold a strong promise to support the definition, implementation and monitoring of global- and national-scale conservation strategies for ensuring the long-term persistence of ecosystem functions and associated ecosystem services.

2. Metrics of functional diversity and functional rarity

An alternative to targeting species as the key biodiversity units in conservation is to focus on traits. A functional trait is a well-defined, measurable property of an organism that potentially affects its performance [21]. Traits can be physical (e.g. body mass, bill length, corolla depth), biochemical (e.g. plant photosynthetic pathway, secondary metabolites), behavioural (e.g. nocturnal versus diurnal foraging, diving depth) or temporal (e.g. pelagic duration of the larval stage, flower phenology) [15]. These traits characterize not only the individuals or the species themselves but also how they engage in interactions with other species (e.g. through facilitation or mutualisms [22]) and with the physical elements of their habitat (e.g. as soil engineers [23]). A species can be seen as a collection of individuals whose traits determine when and where the species can exist, and how it interacts with other species and with the abiotic environment. In other words, traits define the species' ecological niche and its role in ecosystem functioning [15,21]. Functional diversity measures the diversity of traits within a community [15]. Given the mounting evidence that functional diversity is a good predictor of ecosystem function and stability [11–16], it is increasingly considered a proxy for ecosystem functionality [15].

Two main mechanisms (non-mutually exclusive) have been put forward to explain the relationship between community diversity and ecosystem functionality [12,13,24,25]. The first is a (stochastic) selection effect: the higher the diversity of species in a community, the higher the probability that it includes species with particularly important traits that dominate ecosystem functioning (e.g. driving high productivity). The second mechanism (deterministic) is based on niche complementarity: niche differentiation between co-existing species (e.g. in terms of strategies for resource exploitation) should translate both into differentiation of species' ecological traits and of species' effects on ecosystem function. Accordingly, the stronger the complementarity of species' niches, the stronger the expected relationship between community diversity and ecosystem functioning, with the disappearance of individual species more likely to result in a reduction of both trait diversity and ecosystem function [13]. The relationship between community diversity and ecosystem stability has in turn been explained by an insurance effect: high functional diversity means having a range of species with a diversity of responses to environmental perturbation, which stabilizes ecosystem process rates in the face of disturbance and variation in abiotic conditions [25].

Functional diversity can be measured through a variety of metrics, including a simple count of the overall number of traits [26], the number of functional groups [27], the branch length of a functional dendrogram built from species' distinctiveness in traits [28] and the size of a hypervolume in an n -dimensional trait space where each species is positioned based on its traits [29].

Phylogenetic diversity is a measure of the overall evolutionary history embodied in a set of species, typically quantified as the total branch length of the corresponding phylogenetic tree [30,31]. It too has been found to be a good predictor of ecosystem function [32] and stability [33], and it is often treated as a proxy for functional diversity [34–36] (but see [37]).

The increasing availability of datasets on the spatial distribution of species (e.g. [38–40]), species' traits (e.g. [41,42]) and phylogenetic relationships (e.g. [43,44]) ushered in the new field of functional biogeography, defined as the analysis of the patterns, causes and consequences of the geographic distribution of the diversity of form and function, i.e. of trait diversity [17]. The IUCN Red List of Threatened Species [10] played an important role in this development, as the datasets compiled through species' assessments (maps, habitat preferences and species' categories of extinction risk) underpin a substantial fraction of the studies in this new field (e.g. [26,45–49]). Datasets covering entire taxonomic groups made it possible to map global patterns of functional and phylogenetic diversity, and these patterns have in turn been used in a growing number of studies as the basis of recommendations for large-scale conservation priorities (e.g. [26,45,48–51]).

One type of analysis contrasts patterns of multiple facets of biodiversity—species' richness, functional diversity and phylogenetic diversity—invariably finding mismatches between them and recommending the establishment of protected areas in hotspots of functional and phylogenetic diversity [26,45,49,50,52]. There is, however, a logical inconsistency in this approach, because the evidence of a mechanistic relationship between diversity and ecosystem function or stability comes from studies at the scale of communities, i.e. assemblages of co-occurring species whose niche characteristics and ecological interactions translate into ecosystem functions. Even though such studies routinely compare multiple sites or plots, they treat them as replicates of one same community, i.e. presumed to be similar in all environmental parameters except in species'/trait composition, with species'/trait variation obtained through manipulation in experimental plots [53], or by sampling across gradients of human impact [14,54]). Interpreted as space-for-time substitutions [55], these studies support the assumption that temporal changes in community functional diversity (e.g. declines associated with local extinctions [56], or increases in response to ecosystem restoration [57,58]) translate into changes in functionality. What these studies do not support is the assumption that macroecological variation in functional diversity, observed across very different distant communities, reflects variation in ecosystem functionality. Even when an empirical positive relationship is found between diversity and functionality (e.g. between species' diversity and ecosystem productivity [59]), there are other confounding variables (e.g. available energy and evolutionary history) that preclude the assumption of a causal relationship between the two. In summary, whereas there is good evidence that temporal variation in the functional diversity of a given community holds conservation-relevant information, there is no empirical or theoretical basis for assuming that macroecological patterns of functional diversity are informative of large-scale priorities for conserving ecosystem function.

Other studies investigated the extent to which the current global network of protected areas covers the overall diversity of species' functional traits (and/or of branches of the global phylogenetic tree) and then identified priorities for expanding the protected area network in order to optimize such coverage [26,46,47]. This, however, is not about the conservation of ecosystem functionality but about ensuring the efficient representation of traits (or tree branches). It only makes sense as a conservation strategy if the traits (branches) are valued as biodiversity elements in their own right (e.g. based on their intrinsic value or for aesthetic reasons), in which case they deserve protection irrespectively of where in the world they occur, and of the ecological context in which they are found. From a functional perspective, however, a trait is only valuable where it exists, and the extent to which it contributes to ecosystem function depends on its complementarity in relation to other traits in those same communities (and not on its complementarity in relation to a global pool of traits).

A similar logical inconsistency besets analyses that focus on functional distinctiveness as a basis for identifying priority species for conservation actions such as reintroductions [60] or the designation of protected areas [48,51]. Functional distinctiveness [61] (or functional uniqueness [62]) reflects the extent to which a species' traits make it dissimilar to others in the same community. Given that species with high functional distinctiveness make unique contributions to the functional diversity—and so, presumably, to the function—of the communities they belong to, it makes sense to consider them as conservation priorities within the context of those communities. However, this logic does not apply when species' functional distinctiveness is assessed in relation to large pools of not necessarily co-occurring species (e.g. all birds or mammals of Europe [60] or of the world [51]; globally distributed coral reef fishes [48]). Species that stand out as functionally distinct in these large-scale contexts are valid conservation priorities under existence and aesthetic considerations, as unusual and irreplaceable outputs of the planet's evolutionary history (likewise for Evolutionarily Distinct and Globally Endangered species [63]). There is no evidence that they contribute disproportionately to large-scale ecosystem function.

Even less defensible is to use functionality arguments to call for the conservation of species that are 'ecologically rare'—defined as being simultaneously functionally distinct at the global scale and geographically restricted [48,51]—given that these species can only contribute to ecosystem function in the (few) places where they occur. Arguably, it makes instead sense to call for the conservation of species that are locally functionally distinctive (i.e. within the specific context of the communities where they occur) and very widespread (i.e. with functional roles across large geographic areas), such as some apex predators [64] and ecosystem engineers [65].

In summary, whereas there is a solid body of evidence showing a relationship between trait diversity and ecosystem functionality, it is not straightforward to translate it into recommendations of large-scale conservation priorities for conserving functionality itself. A main limitation of macroecological indices like functional diversity or species' functional distinctiveness is that they are general, whereas functionality is necessarily context-specific. The patterns obtained through these indices hold clues to understanding the ecological and evolutionary mechanisms underpinning the distribution of biodiversity form and function [17], but are not easily actionable as the basis of conservation priorities. Below, I discuss three approaches for integrating functionality into large-scale priorities while taking into account the specific context of each ecosystem, site or species: the IUCN Red List of Ecosystems, Key Biodiversity Areas and the IUCN Green Status of Species.

3. The IUCN Red List of Ecosystems

The IUCN Red List of Ecosystems (RLE) is a global framework for assessing and monitoring the conservation status of ecosystems [66]. Developed to complement the IUCN Red List of Threatened Species as a tool for evaluating the risk of global biodiversity loss [67], it addresses a higher level of biodiversity organization than that of species, by focusing on ecosystems and their functioning [68,69]. The RLE standard is well established and accepted, with important roles in 16 of the 23 targets of the Global Biodiversity Framework [19], including as a headline indicator for goal A (maintaining, enhancing and restoring the integrity, connectivity and resilience of all ecosystems by 2050) and target 1 (to bring the loss of ecosystems of high ecological integrity close to zero by 2030) [18,20]. Since 2014, the RLE has been applied to over 4000 ecosystem units across over 100 countries, including 63 countries with assessments completed for all terrestrial ecosystems (32 for all marine and 42 for all freshwater ecosystems) [19]. A recently developed global typology of ecosystems, based on ecosystem functions, ecological processes and characteristic biota, is fundamental to ensure the global standardization and comparability of RLE assessments [70].

A RLE assessment involves classifying an ecosystem into one of seven categories of risk ('Collapsed'; 'Critically Endangered'; 'Endangered'; 'Vulnerable'; 'Near Threatened'; 'Least Concern'; 'Data Deficient'), through the application of five criteria (A–E) (table 1) [66]. Two of these criteria explicitly assess the functional symptoms of ecosystem collapse: criterion C focuses on environmental (abiotic) degradation, while criterion D addresses the disruption of biotic processes and interactions [66]. In each case, sub-criteria 1 to 3 refer to the timeline of impacts: ranging from the historical to the recent past (past 50 years), to ongoing impacts and to those predicted for the near future (next 50 years). Each assessment is supported by a conceptual model that renders explicit the assessors' assumptions regarding the key processes underpinning the functioning of the ecosystem, as well as the main threats it faces [66].

RLE assessments through criterion C evaluate the severity and the spatial extent of environmental degradation, i.e. the deterioration of the physical, non-living attributes that have a defining role in ecological processes and/or the spatial distribution of the ecosystem [66]. These assessments entail the identification of variables for which there is plausible evidence (either direct evidence or inference from comparable ecosystem types) that they measure a causal relationship between the process of environmental degradation and the loss of characteristic native biota. Variable choice must, therefore, be tailored to the characteristics of the ecosystem and the threats affecting it, such as quantifying cloud cover and cloud altitude to measure the dehumidification of cloud forests, or remote sensing of ground surface albedo to measure the salinization of soils or wetlands [66]. For example, the Mediterranean White Dunes of France have been evaluated as 'Endangered' under criterion C1 as a result of a significant reduction in sediment supply combined with an increased restriction to natural dune movement in the past 50 years [71].

RLE assessments based on criterion D follow a conceptually similar approach, but are instead focused on the disruption of biotic processes and interactions, such as competition, predation, facilitation, mutualism, trophic and pathogenic processes, migration and species' invasions [66]. Again, assessments involve the identification of meaningful biotic variables for which there is plausible evidence that they measure a causal relationship between biotic disruption and the degradation of ecosystem function. Potential variables include: changes in species' richness, composition and dominance; relative abundance of species' functional types, guilds or alien species; measures of interaction diversity; changes in identity and frequency of species movements; measures of niche diversity and structural complexity [71]. For example, the Oyster Reef Ecosystem of Southern and Eastern Australia has been evaluated as 'Critically Endangered' under criterion D3 (i.e. very close to collapse in relation to a historical baseline) as a result of a massive decline in the biomass of key species—oysters—across the entire extent of the ecosystem, inferred from historical records of oyster exploitation [72].

In addition, RLE assessments based on criterion E can use ecosystem models (of various levels of complexity) that integrate multiple functional processes and their interactions and dependencies to quantify the probability of ecosystem collapse. For example, in the Mountain Ash Forest Ecosystem of the Central Highlands of Victoria (southeastern Australia), the density of hollow-bearing trees was identified as a biotic variable integrating the interaction between forest age structure, natural disturbance (fires) and human disturbance (logging) [73]. This ecosystem was evaluated under criterion E by applying a probabilistic model of tree growth stages (built from empirical data from long-term field sites) to simulate the density of hollow-bearing trees under a wide diversity of scenarios varying in harvesting intensity and fire regimes. These simulations consistently predicted a $\geq 50\%$ probability of collapse within 50 years, corresponding to a 'Critically Endangered' ecosystem [73]. Even if criterion E has had so far limited application in RLE assessments, it holds great potential because similar types of models can be applied to assessing groups of many related ecosystem types, owing to similar mechanisms of functionality [70].

The RLE is related to another global standard for measuring and assessing ecosystem change: the United Nations System of Environmental Economic Accounting Ecosystem Accounting (SEEA-EA; [74]). Both the RLE and the SEEA-EA evaluate ecosystem conditions, based on condition variables selected for their relevance to measuring ecosystem features and processes, and for their sensitivity at detecting ecosystem change. They differ in their ultimate aim: whereas the RLE focuses on assessing risks of ecosystem collapse and biodiversity loss, the SEEA-EA focuses on evaluating the contributions of ecosystems to people and the economy. They thus have complementary roles in informing policy for meeting the dual imperatives of conserving biodiversity and sustaining human well-being [75].

Table 1. High-level criteria and sub-criteria for assessing the risk of ecosystem collapse based on the IUCN Red List of Ecosystems. For details, see [66].

high-level criteria	sub-criteria
A. reduction in geographic distribution	A1. in the past 50 years
	A2a. predicted in the next 50 years
	A2b. over a 50-year window, including the past, present and future
	A3. historical (since approximately 1750)
B. restricted geographic distribution	B1. highly restricted extent of occurrence and continuous decline or found in very few locations
	B2. highly restricted area of occupancy
	B3. very few locations and potentially at risk from human activities or stochastic events
C. environmental degradation (based on a change in an abiotic variable)	C1. in the past 50 years
	C2a. predicted in the next 50 years
	C2b. over a 50-year window, including the past, present and future
	C3. historical (since approximately 1750)
D. disruption of biotic processes or interactions (based on a change in a biotic variable)	D1. in the past 50 years
	D2a. predicted in the next 50 years
	D2b. over a 50-year window, including the past, present and future
	D3. historical (since approximately 1750)
E. quantitative analysis estimating the probability of ecosystem collapse	

4. Key Biodiversity Areas

Key Biodiversity Areas (KBAs) are sites that contribute significantly to the global persistence of biodiversity, identified using a set of globally standardized criteria and quantitative thresholds [76], through the combined efforts of 13 organizations that together form the KBA partnership. Current KBA criteria have been built from a methodology first developed for the identification of Important Bird and Biodiversity Areas (IBAs) [77] and subsequently expanded to cover a wider diversity of taxa [78,79] as well as ecosystems. The World Database of KBAs currently lists over 16 300 sites [80]. Whereas most of these still correspond to IBAs, the database is expanding quickly to cover other taxa, as well as sites identified for their global importance for the conservation of biodiversity at the ecosystem level. KBAs also contribute to the GBF monitoring framework, including as a component indicator for target 3 (effective conservation of at least 30% of the land and seas by 2030, especially areas of particular importance for biodiversity and ecosystem functions and services) [20].

A site is defined as a KBA if it meets at least one of eleven criteria, grouped into five higher-level criteria (A to E; table 2) [76,81]. Sites identified through criteria A2 or B4 contribute significantly to the persistence of, respectively, threatened or geographically restricted ecosystem types, with the application of these criteria directly linked to the IUCN RLE. In addition, functionality is explicitly taken into account in KBA assessments through criteria C (which addresses ecosystem-level functionality) and D (focused on species-level biological processes).

Sites qualifying as KBAs under criterion C represent globally outstanding examples of high ecological integrity, with only one or two such sites to be identified per ecoregion. For the purpose of KBA assessments, ecological integrity is defined as the degree to which an ecosystem's observed structure, function and composition resemble those characteristics of regionally appropriate historical benchmarks or other high-integrity reference states, within the range of natural variability characteristic of the geographic location, and with minimal impacts from threatening processes (including industrial human activities) [81]. Assessment of integrity is thus context-dependent, and should rely on indicators of ecosystem structure, function and composition that enable a meaningful comparison between the candidate site and an appropriate reference state. Examples of such indicators include the presence and density of: species that underpin key ecosystem functions (e.g. top predators, ecosystem engineers, seed-dispersers); species sensitive to broad-scale ecological processes (e.g. fire, flooding); species highly sensitive to human impacts (e.g. hunting, habitat degradation); area-demanding species. Indicators of ecological integrity may also quantify ecosystem properties such as species' composition (e.g. trophic structure), ecosystem structure (e.g. the vertical and horizontal spatial arrangement of the vegetation) or relevant ecological processes (e.g. migration, productivity) [81]. The methods for identifying KBAs under criterion C are still being refined and tested. A first global scoping of ecological integrity found that sites with low levels of human impact, complete mammal faunal assemblages (in relation to the year 1500 AD), relatively high densities of wide-ranging large mammals, and that are larger than 10 000 km² occupy only 2.8% of the global land surface, of which only 11% are covered by protected areas [82].

Whereas criterion C focuses on functionality at the ecosystem level, criterion D approaches it from the perspective of species, by identifying sites that contribute significantly to the global persistence of demographic aggregations (D1), ecological refugia (D2) or recruitment sources (D3). A demographic aggregation corresponds to a geographically restricted clustering of individuals, indicated by a highly localized relative abundance [76,81], typically during a specific lifecycle process such

Table 2. High-level criteria and sub-criteria for classifying sites as KBAs. For details, see [76,81].

high-level criteria	sub-criteria
A. threatened biodiversity	A1. threatened species A2. threatened ecosystem types
B. geographically restricted biodiversity	B1. individual geographically restricted species B2. co-occurring geographically restricted species B3. geographically restricted assemblages B4. geographically restricted ecosystem types
C. ecological integrity	
D. biological processes	D1. demographic aggregations D2. ecological refugia D3. recruitment sources
E. irreplaceability through quantitative analysis	

as breeding, feeding or migration. A site qualifies as a KBA under criterion D1 if: (i) it predictably holds an aggregation representing $\geq 1\%$ of the global population size of any given species or (ii) if the site ranks among the largest 10 aggregations known for the species [76]. Criterion D2 is also triggered by the aggregation of individuals of one or more species, but rather than seasonal aggregations (as in D1), it refers to concentrations that take place during periods of environmental stress [81], for example, permanent wetlands that are disproportionately important during droughts. A site qualifies as KBA under criterion D2 if historical evidence shows that it has previously served as an ecological refuge for a species (by supporting $\geq 10\%$ of its global population size) during periods of environmental stress, and for which there is evidence to suggest it would continue to do so in the foreseeable future [76,81]. Finally, criterion D3 focuses on sites that are disproportionately important to the persistence of one or more species as sources of propagules, larvae or juveniles that make a large contribution to the recruitment of mature individuals elsewhere (by maintaining $\geq 10\%$ of the global population size of a species) [76]. Examples include: sites where plants or fungi produce a large number of seeds or spores that have a high probability of dispersing, germinating and surviving to maturity elsewhere; and nursery sites where large numbers of larvae settle and have a high probability of growing into juveniles that survive to maturity and disperse elsewhere [81].

5. IUCN Green Status of Species

The IUCN Green Status of Species is a new global standard for assessing species' recovery and conservation impact [83]. Like the IUCN Red List of Threatened species, it evaluates individual species, but rather than analysing risk of extinction, it assesses how close each species is to being in a favourable conservation status termed 'Fully Recovered'. A species is considered as 'Fully Recovered' if it is present, viable and ecologically functional across its entire range [83,84].

A Green Status assessment involves the calculation of 'Green Scores', ranging from 0% for species that are either 'Extinct' or 'Extinct in the Wild', to 100% for a 'Fully Recovered' species [83]. Green Scores are calculated by combining information on the state of each species (absent, present, viable or functional) in each spatial unit within its range (corresponding to range subdivisions, delineated by subpopulation, ecological and geographical features, or location; e.g. watersheds, ecoregions and flyways). Multiple Green Scores are estimated per species, corresponding to different scenarios: at different points in time (past, current, short-term future and long-term future); and either assuming that conservation was in place (or is, or will be), or not. Five different metrics are then derived from these scores. The 'Species Recovery Score' corresponds to the current Green Score (i.e. at the time of assessment), and its value is used to allocate each species into one of the eight recovery categories ('Extinct in the Wild'; 'Critically Depleted'; 'Largely Depleted'; 'Moderately Depleted'; 'Slightly Depleted'; 'Fully Recovered'; 'Non-Depleted'; 'Indeterminate'). The other four metrics estimate the effects of conservation action on the species' status, and are obtained by contrasting Green Scores in scenarios with and without conservation: 'Conservation Legacy', which measures the impact of past conservation actions on the current score; 'Conservation Dependency', measuring the extent to which species' status would deteriorate in the short term (10 years) if all ongoing and planned conservation actions were to cease; 'Conservation Gain', measuring how much the species' status stands to improve in the short term from ongoing or planned conservation actions; and 'Recovery Potential', measuring how much the species' status could improve in the long term (100 years) from plausible conservation effort and innovation [83].

The Green Status framework thus promotes a vision of conservation that is far more ambitious than just preventing extinctions. By treating species as embedded in ecosystems, fostering the conservation of their ecological functions across their native distributions and the recovery of such functions where they have been lost [83–85], it connects and integrates conservation at the species' and ecosystem levels [85]. Defining and quantifying species' ecological function in the context of species' recovery is thus key to Green Status assessments. This should not be misinterpreted as prioritizing species based on their functional importance, but as an evaluation of the degree to which each species performs its role as an integral part of the ecosystem in which it is embedded. Facets of species' functionality include their interactions with other species (e.g. pollination,

seed dispersal, predation, facilitation), their structural or landscape effects (e.g. creation of habitat for other species, ecosystem engineering, substrate stabilization), their contribution to ecosystem-level processes (e.g. primary production, decomposition, effect on fire regimes) and within-species processes (e.g. migration, aggregations, adaptation) [85]. Calculating Green Scores involves evaluating whether or not each species is functional within each spatial unit within its range. Functionality can be assessed directly (by quantifying the extent of the species' contribution to interactions and ecosystem processes), indirectly (based on symptoms of reduced functionality, such as population declines or a contraction of occupied habitats) or using proxies (e.g. by comparing population density or age structure in the assessed unit with observed values in areas of low human impact or at a historical baseline) [85].

The Green Status framework is still under development, including through a recent test on a diverse set of 181 species that has enabled further refinement of the standard [86]. In the medium- to long-term, the ambition is that Red List and Green Status assessments will be undertaken simultaneously, producing for each species an evaluation of its extinction risk as well as of its recovery status and the impact of conservation actions. Thirty-eight species have already been formally assessed under the Green Status standard, with the results accessible via the IUCN Red List website (<https://www.iucnredlist.org>). For example, the American horseshoe crab (*Limulus polyphemus*) is classified as 'Moderately Depleted' (Species' Recovery Score = 69%) because even though it is viable throughout much of its range, it is not functional in most units. Harvest management and habitat protection are key to enabling its recovery to the high densities underpinning its functional roles, including as a major food source for migratory shorebirds, and in structuring the physical and chemical environment of benthic environments and estuarine beaches through bioturbation [87].

Even though it is still being developed, a future Green Status of Species Index is already foreseen as a component indicator for target 4 of the GBF (ensure the recovery and conservation of species, in particular, threatened species) [20]. In parallel, the IUCN's Commission on Ecosystem Management is in the early stages of conceptual development towards a Green Status of Ecosystems, a new standard for assessing past, current and future recovery of ecosystems [88].

6. Beyond keeping the 'cogs and wheels'

Whereas preventing species' extinctions remains an essential objective of conservation efforts, it is undeniable that it must be complemented by the long-term preservation of ecosystem functions and the benefits humans derive from them. This ambition is clearly stated in the Kunming–Montreal Global Biodiversity Framework [18], which sets as goals for the year 2050 not only that species' extinction should be halted and extinction risk reduced but also that the abundance of species should be increased to healthy and resilient levels, and that the integrity, connectivity and resilience of all ecosystems should be maintained, enhanced or restored. To meet these goals, the GBF includes a set of action-oriented targets for urgent action over the decade to 2030, which include: urgent management actions to halt the extinction of known threatened species, and for the recovery and conservation of species; bringing the loss of areas of high biodiversity importance, including ecosystems of high ecological integrity, close to zero; ensuring that at least 30% of areas of degraded ecosystems are under effective restoration; and ensuring the protection of at least 30% of global land and sea area, especially areas of particular importance for biodiversity and ecosystem functions and services.

Defining strategies for meeting these goals and targets—as well as for monitoring progress towards their attainment—requires global standards that operationalize key concepts such as 'threatened species', 'extinction risk', 'species' recovery', 'ecological integrity', 'effective restoration' and 'areas of particular importance for biodiversity and ecosystem functions and services'. The IUCN Red List of Threatened Species [10] plays a crucial role as the global standard for identifying threatened species and assessing extinction risk (with the derived Red List Index as a headline indicator for monitoring goal A of the GBF of reducing the extinction rate and risk of all species tenfold by 2050, and of target 4 on urgent management actions for threatened species) [89]. The immense value of the Red List of Threatened Species to conservation [90] has been accrued through the joint work of thousands of experts, who have already undertaken over 150 thousand species' assessments. Expanding its taxonomic coverage, and ensuring repeated assessments to monitor changes in extinction risk over time, are key to tackling global conservation challenges [89].

Similarly, the three other standards reviewed here are vital to meeting the GBF's ambition of ensuring the conservation of functional ecosystems and associated services. The IUCN RLE [66] is key to standardized evaluations of the ecological integrity of ecosystems, and to set priorities for the conservation and restoration of ecosystem functions [19,91]. The identification of KBAs [76,81] enables the identification of specific sites that are globally important for preventing ecosystem collapse, and for preserving functionality at the levels of species and ecosystems. Finally, the IUCN Green Status of Species will support strategic conservation efforts for promoting the conservation and recovery of species that are functionally embedded within their ecosystems.

These three standards differ substantially in their levels of development and implementation, at either global or national scales. Global-scale analyses are important for global synthesis, review of progress and for research, yet most conservation action takes place within countries. Accordingly, the practical usefulness of these standards will rely critically on the extent to which they can be translated into national-level recommendations [67]. The RLE framework is well established and accepted and has been widely applied at the national scale. Whereas it still lacks spatial coverage across many countries and regions, this coverage is likely to increase via the GBF monitoring framework [19]. KBAs have already been mapped out in most countries of the world [80]. They are, however, mostly based on an earlier version of the standard, and mainly focused on IBAs, with KBA criteria related to ecosystem persistence, integrity and functionality still at various stages of development, testing and implementation. The Green Status of Species has great potential but is still in a relatively early phase of development and

testing, having so far been applied to just a few dozen species [10]. Developing and applying each of these standards to its full potential will require the diligent collaboration of an even larger number of experts than those already involved in species Red List assessments. Such detailed work cannot easily be bypassed because evaluating the functionality of ecosystems, or the role of each species in ecosystem function, is necessarily context-specific. In the meantime, we must resist the temptation of applying broad macroecological indicators of functional diversity as shortcuts, lest they give the false reassurance of being sufficient to ensure the conservation of ecosystem functionality.

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