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Why it matters how biodiversity is measured in environmental valuation studies compared to conservation science

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A R T I C L E I N F O

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A B S T R A C T

The undervaluation of biodiversity in decision-making is a critical issue that contributes to continued biodiversity declines and loss of environment. This issue is exacerbated in environmental economics by the need to keep measures of biodiversity simple for communication to the public due to limited background knowledge and cognitive limitations. Therefore, there is a clear need to improve the biodiversity metrics used in biodiversity valuation and environmental economics, without using overly complex measures. However, it is unclear how much overlap exists in the metrics currently used in these fields as compared to those being used in more biodiversity focused disciplines such as conservation and ecology. Here, we use a rapid evidence assessment approach to categorise the measures and attributes used in environmental valuation studies into broad groups of biodiversity metrics. We compare this to previous research categorising biodiversity metrics used in conservation and ecology to determine how well environmental valuation studies are capturing the values important for measuring biodiversity in practice. We find a high degree of overlap in the broad biodiversity metrics used in environmental valuation compared to conservation and ecology. However, the overlap mostly consistent of simplistic easy to measure habitat attributes and species occurrence measures. The measures generally fail to capture the ecosystem processes driving biodiversity persistence and therefore may not capture the ecosystem services or welfare attributes important to people. We discuss the implications of these areas of mismatch, and point towards future directions in stated preference research and technological advances, which might allow for the valuation of more complete and complex dimensions of biodiversity.

1. Introduction

1.1. *Maximizing the economic net-gains from conservation investments*

The ongoing decline in global biodiversity and its capacity to safeguard human welfare (Díaz et al., 2019) continues partly because biodiversity is undervalued in decision-making at both micro and policy scales (Dasgupta, 2021). Undervaluing biodiversity in decision-making (by allocating it a zero-price value) implicitly enables its destruction. Policies that negatively impact on nature are often approved in part because the costs of these projects on nature and biodiversity loss are inadequately accounted for in decision-making processes. Additionally these impacts are often perceived as less than the monetary benefits of the projects such policies support (Vilela et al., 2020).

This underweighting of biodiversity also leads to under allocation of resources to its protection and restoration. Despite evidence that nature conservation can come with substantial economic benefits (Sims et al., 2019; Waldron et al., 2020), governments consistently underfund nature conservation and protection. For example, the European Commission in 2022 released a study which estimated that the annual funding needs between 2021 and 2030 to implement the EU Biodiversity Strategy for 2030 were EUR 48 billion with a remaining financing need of EUR 19 billion per year (Nesbit et al., 2022). At the global scale, Deutz et al. (2020) estimate that $124–143 billion per year is spent on actions

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relating to biodiversity conservation, contrasting with the estimated spending need of $722–967 billion per year. Despite these shortfalls, in some countries such as in the UK, funding trends continue to move in the wrong direction (Seidl et al., 2021; zu Ermgassen et al., 2021).

For societal decision making, where biodiversity conservation is one among many goals, it is important that benefits outweigh the costs. Applying economic principles and tools may be useful to assess the costs and benefits of biodiversity conservation (Naidoo et al., 2005; Petersen et al., 2016). This approach may also help to improve the effectiveness and efficiency of biodiversity conservation (Iftekhar et al., 2017). Principles of ecology allow us to properly estimate the biodiversity outcomes and benefits of conservation, which can be valued economically. Bringing economics and ecology together make it possible to apply cost-benefit analyses to identify which interventions should be prioritized (Dasgupta, 2021; Naidoo and Adamowicz, 2005) and to apply decision theory to identify the most cost-effective allocations of a budget (Wilson et al., 2006). Stated preference research in environmental economics has been applied extensively to evaluate the economic benefits of biodiversity (Bartkowski et al., 2015; Christie et al., 2006; Hanley and Perrings, 2019). Even if not included directly in valuation, preference research may be useful for evaluating whether policy goals correspond to the general public's perception (Jacobsen et al., 2008), or for advocacy, performance tracking, and accounting in public and private settings (Tinch et al., 2019).

1.2. Coupling conservation and economic models

An aspiration for applied biodiversity valuation research has been to create coupled ecological-economic models that can accurately predict the effectiveness of conservation interventions, their associated benefits and costs, and their resulting enhancement of economic welfare (Strange et al., 2007). A number of studies include considerations of conservation costs and outcomes (Evans et al., 2016; Hammill et al., 2016; Petersen et al., 2016). Comparatively, few studies compare the economic costs of conservation actions with the associated economic benefits of outcomes of such conservation actions. One example of a coupled ecological-economic conservation model is Naidoo et al. (2005), who applied a willingness-to-pay estimate for tourists seeing avian biodiversity alongside a species-area-relationship model to predict the optimal reserve area compared to the opportunity cost of lost land rent. Similarly, Strange et al. (2007) applied welfare economic benefit and cost estimates for preserving endangered species in heathland to identify the optimal investment. Both studies demonstrate that coupling of ecological-economic conservation models makes it possible to compare the costs and benefits of conservation interventions and identify economic optimal conservation strategies.

Economic models used in biodiversity valuation must rely on biologically sound and evidence-based conservation models. For example, García-Díaz et al. (2019) suggest that the use quantitative models in conservation management could significantly improve our capacity to solve "wicked" conservation problems, as well as improving the ability of management to account for both individual species needs and ecological system dynamics. However, to achieve this long-term research ambition of coupling ecological and economic conservation models, it is important that economic studies valuing biodiversity 'measure' the same aspects of biodiversity as conservation. In some instances, there need not be a conflict. For example, a charismatic animal being highly valued by people may also have an important ecological function; or is only present if the biodiversity characterising the wider ecosystem is doing well. However, in situations where biodiversity conservation is considered a dynamic approach e.g. as in rewilding projects which have uncertain outcomes (Perino et al., 2019), it could lead to discrepancies between what is valued ecologically versus what people value most. Therefore, it is important that the estimates of welfare enhancement derived from economic studies are related to the biodiversity benefits of the conservation interventions they are attempting to accompany. There are fundamental constraints to ensuring that economic valuation and conservation science use the same biodiversity indicators, which is that the purposes and needs from the two disciplines are different.

Within the valuation literature non-use values are particularly important, yet challenging to assess (Hanley and Perrings, 2019). There are a limited number of methodological tools available when attempting to value the non-use values of biodiversity: most commonly these include, stated preference methods such as choice experiments (Christie et al., 2006), contingent valuation (Jacobsen et al., 2008) or ranking or deliberative monetary valuation (Kenter et al., 2016). They rely on surveys where proposed changes in conservation are carefully described and explained, and for choice experiments divided into separable and mutually independent attributes (e.g. effect) or characteristics. For example, a choice experiment may use the metric species richness as the attribute characterising the effect of a policy on biodiversity. The choice set in a choice experiment should foremost be relevant, concise, comprehensive, and complete (Bakhtiari et al., 2014; Christie et al., 2006; Johnston et al., 2017). This also means, that it should be explicit about explaining biodiversity in a way that captures the values people have for biodiversity. Furthermore, the survey should include a practical description of the attributes of the intervention, i.e., the extent and purpose of the intervention, as well as the effect on biodiversity. It may be practical only to use a limited number of attributes due to cognitive limitations. Maintaining simplicity may minimize the risk of over-burdening respondents and the risk of attributes capturing overlapping components. However, as a result the original metric of biodiversity, species richness, may not be entirely captured within the designed choice experiment.

Quantifying the value of a biodiversity preservation intervention requires that the characteristics of the intervention and the predicted impacts on biodiversity, alongside associated uses and non-use values are specific and measurable. However, the overly simplistic scenarios frequently used in valuations do not always capture people's perspectives of biodiversity or the complicated nature of measuring biodiversity in practice (Bakhtiari et al., 2014). Incomplete description of what is valued increases the risk of undervaluing biodiversity because it causes respondents to only consider specific aspects of biodiversity (Ojea and Loureiro, 2011), potentially ignoring some of the factors that would be considered critical in conservation and ecological research. However, using realistic scenarios with informative metrics for describing biodiversity may risk presenting respondents with scenarios which encompass associated benefits, thus biasing their responses. The balance between addressing the appropriate dimensions of biodiversity and avoiding jargon and overly complex choice decisions to laypeople is evident. Stated preference survey techniques are challenging because respondents often have little understanding of biodiversity or poorly formed preferences (Bateman et al., 2014). The main functions of metrics (attributes) in environmental valuation studies are usually to identify the implicit trade-offs between the levels of the attributes in the different alternatives included in a choice set; to try to estimate the marginal utility of the attributes in the different alternatives; and to apply the results within a cost-benefit analysis framework (Hoyos, 2010).

In contrast, metrics describing biodiversity are used in conservation for a wide variety of purposes generally with two major applications. Firstly, biodiversity metrics are critical for monitoring and evaluation of ecological trends over time, which can be used to estimate species decline, predict tipping points or extinction risks, and to raise awareness of conservation needs (Pereira et al., 2013; Turak et al., 2017; Schmeller et al., 2018; Home et al., 2009). Secondly, metrics are used to compare the spatial and temporal distributions of species and ecological communities which is often critical for conservation planning and management. Biodiversity metrics are also necessary for measuring the impact of conservation projects as well as the adverse effects caused by infrastructure projects, urban development or changed land uses, etc.

(Butchart et al., 2010; Halpern et al., 2008; Purvis and Hector, 2000). Previous research by Hanley and Perrings (2019) found that within stated preference studies there is generally a focus on a narrow set of species and habitats. Additionally, Bartkowski et al. (2015) found that that even though the complexity and multidimensionality of biodiversity as a concept are well recognised, most studies only applied single attribute metrics. They recommended i) a multi-attribute approach to valuing biodiversity, keeping in mind cognitive limitations of humans; and ii) to focus the valuation research away from attributes and towards the role of biodiversity for humans. Farnsworth et al. (2015) re-evaluated the studies in Bartkowski et al., and argue that the studies were not applying appropriate scientific metrics for biodiversity but rather valuing 'naturalness' or some specific biological component of diversity. Both reviews lack a systematic comparison of metrics applied in economic valuation and ecology and conservation science.

Here, we expand both of their work by reviewing and comparing biodiversity metrics applied in ecology and conservation science with those applied in valuation studies which use stated preference methods. We compare biodiversity metrics applied in environmental economics, more specifically stated preference studies, with metrics found in Marshall et al. (2020). Marshall et al. (2020) conducted a quantitative review, based on 255 peer-reviewed publications from three fields of research; offsetting ($n = 43$), conservation planning ($n = 54$) and ecology ($n = 158$). They explored which biodiversity metrics are commonly used in offsetting, conservation, and ecology, and assigned these metrics into 24 categories which captured the breadth of approaches used to measure biodiversity in all three fields of research. While our aims and topics are quite different to Marshall et al. (2020) the biodiversity categories collated from the conservation and ecological literature in their review is a useful framework with which to assess how it is treated in environmental economics. We do not seek to capture the full suite of biodiversity metrics used in economic valuation studies, but rather to gather a representative sample of the literature with which we can compare the broad trends of biodiversity metric use between environmental economic studies, conservation and ecology. We seek to understand the degree of overlap between the metrics used to quantify biodiversity in environmental economics compared to conservation or ecological research. We aim to identify commonalities and mismatches in how biodiversity is measured within these two fields of research to identify how environmental economics may align more closely with conservation objectives. We discuss the implications and point towards future directions in stated preference research and technological advances, which might allow for the valuation of more complete and complex dimensions of biodiversity.

2. Materials and methods

We broadly followed the protocol developed by Marshall et al. (2020) to conduct a rapid evidence assessment (Pickering and Byrne, 2014; Varker et al., 2015) of the stated preference literature (Fig. 1). We used the search string: 'Biodiversity' AND {"contingent valuation" OR "choice experiment"'} in Scopus (accessed 5th December 2019), capturing all papers from 1999. By that, we ensure capturing studies with a specific emphasis on biodiversity and also, expectedly, the most nuanced in terms of metrics used to measure it. But we acknowledge that we do not get a comprehensive review of all studies conducted related to biodiversity (e.g. conservation of specific species, genetic studies of populations, or nature conservation in a broader sense). Instead, we aimed to gather a representative sample of literature which we could use to gather broader trends of biodiversity metric use in stated preference methods such as choice experiments. We conducted two stages of screening to identify which papers were suitable for inclusion in the review. First, we filtered papers by the relevance of titles and by reading the abstracts to ensure that all publications were within scope (related to environmental economic studies) and all were primary studies (Fig. 1). Secondly, we excluded papers, which did not contain biodiversity or conservation-outcome attributes/metrics. We also excluded papers that were not available online, or were based on datasets from earlier papers already included in the dataset, or included insufficient descriptions of methods. Additionally, due to linguistic constraints in the project team, we excluded non-English literature which can be defended in this case as the purpose of the review was to get a general overview of the literature and the distribution of metrics rather than a detailed analysis requiring the input of all available information (e.g., a meta-analysis). Information on exclusion reason is available in the database in Supplementary Information. Once a final database was achieved, we assessed the first 25 % of the literature to determine whether any changes in the biodiversity categories were needed. This 25 % of the literature was assessed by two individuals and notes on their categorisation of biodiversity metrics were compared. This allowed us to achieve consensus on the metric categories and how biodiversity metrics were assigned into these categories. The final stage of this analysis was to assess the bulk of the literature. We split the total databased between two assessors and compared categorisations across approximately 10 % of the studies to ensure consistency in the categorisations. We then analyse the resulting proportions of metric use.

From each study, we extracted the:

- country and location of the choice scenario;
- participant type (e.g. local people, national population, farmers);
- spatial scale of the decision-making context being informed by the choice scenario (e.g. choice experiment is informing local, regional, national or international decision-making);
- decision-making context informed by the scenario;
- proposed type of conservation intervention delivering the changes in biodiversity captured in the valuation exercise;
- biodiversity feature being valued;
- ecosystem type being valued, or whether the scenario was focused on a specific species or aspect of wildlife;
- specific biodiversity attribute and attribute levels used in the study.

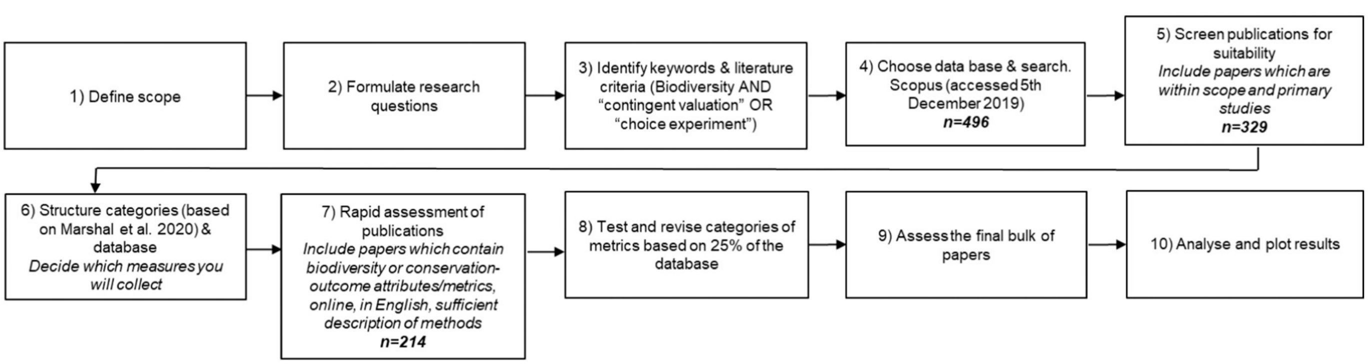

Fig. 1. Schematic illustration of each step in the review approach. The squares illustrate the broad assessment procedures.

As such, we have developed a searchable database of biodiversity metrics used in a large (but not exhaustive) sample of contingent valuation and choice experiments, which can be filtered by any of these categories and may be a useful resource for informing future contingent valuation studies (Supplementary Information). Importantly, we note that biodiversity metric use was recorded as a presence absence measure. For example, when a single study surveyed multiple sites but used the same metric to determine conservation effectiveness (e.g. species richness) this metric was recorded only once as present. However, if multiple biodiversity metrics were used, we recorded each of these attributes separately as part of the one study. For example, if a study used both species richness and a population specific assessment of abundance, we recorded presence for both these biodiversity metrics.

For all of the choice experiment studies in our final database we categorised each attribute measured into a set of qualitative metric categories adapted from Marshall et al. (2020; Table 1). These metrics capture the types of measurements which are used in conservation planning and ecology to quantify biodiversity. We also excluded contingent valuation studies from this component of the analysis because after initial assessments it became apparent that the biodiversity attributes in contingent valuation studies are normally presented within a bundle of other ecosystem goods and services, so they defy neat classification into different attribute categories or outcome metrics. The exact biodiversity indicator used in any given scenario is context-specific, and the decision-making contexts differ between the ecology and the environmental economics literatures. In stated preference studies (such as choice experiments) it is necessary for participants to be able to conceptualise the good being valued, and as such, the exact biodiversity metric being used must in most contexts be something innately identifiable. In conservation planning, the intended output is to develop spatial plans of nature conservation priorities for focusing conservation attention. We use the typology outlined in Marshall et al. (2020) to classify the indicators into broader dimensions of biodiversity, which allows us to investigate whether there are any systematic differences between the dimensions of biodiversity being used between the two literatures (Table 1). As with the original analysis conducted in Marshall et al. (2020), indicators could belong to multiple categories. For example, an attribute such as 'Percent (%) change in populations and range of threatened charismatic species' would be classified as both a 'taxa abundance' and 'distributions' metric, as both the abundance and the range size are being captured in the same metric. When comparing the distribution of metrics between the environmental economics, conservation science, and ecology literatures, we applied a distance measure (see Eq. (1) later in the text) to assess how dissimilar the use of metrics are between the three literature categories.

3. Results and discussion

From our initial Scopus search we collected a total of 496 papers. We reduced this to 329 publications by assessing the titles and abstracts for suitability (within scope and based on primary studies) and removing duplicates. We comprehensively assessed the remaining papers, and excluded studies that did not fulfil our inclusion criteria (see description above, i.e. they did not measure biodiversity per se, they used data that was included in an alternative paper in our database, could not be accessed online, or were not written in English). Therefore, we extracted information about the study design and choice context from 214 papers. Of those 116 papers were choice experiments from which we identified 250 specific biodiversity metrics in our final dataset. We identified more biodiversity metrics than the number of papers since most papers used more than one metric to capture biodiversity. The metrics were categorised into our 24 categories based on Marshall et al. (2020).

3.1. General results

Our dataset shows a number of expected patterns: the number of

Table 1
The categories of biodiversity metrics used in Marshall et al. (2020), and the description of metrics falling into these categories used in this review. Categories marked with * denote metrics that were only found in the environmental economics literature and not in the ecological literature (i.e. in Marshall et al., 2020).

Category	Sub-Category	Definition within this review
Abundance	Species abundance	The number of individuals per species
	Taxonomic abundance	Taxonomic abundance for the purposes of this review was defined as the relative abundance of each taxonomic group within the area or ecosystem of interest
	Taxa abundance	The number of individuals of a given taxa
Area	Area	Any measurement of area which was used to define the size of site or habitat considered of interest. For example, in offsetting, area is used to describe the extent of habitat removed by a development.
Connectivity	Connectivity Indices	Connectivity indices could be considered both structural and functional. Functional connectivity indices included dispersal, emigration and immigration rates. Structural connectivity indices were those, which examined fragmentation and the impacts of structural features within the landscape on the connectivity of populations.
	Landscape metrics	Landscape metrics consisted of any measurement used to describe the landscape configuration of a site or habitat. These included patch size, number of patches within an area, number and size of core areas, edge density, distance of nearest neighbour, distance for core etc.
Density	Population density	Population density is a measurement of the population size per unit area i.e. total population size divided by total land area
	Biomass	Total quantity or weight of organisms in each area or volume
Distinctiveness	Phylogenetic distinctiveness	Measures of how distinct a feature of the population, community or species is relative to other features as determined by its phylogenetic separation.
	Functional distinctiveness	Functional distinctiveness was usually used to determine how unique a biodiversity value or feature is in terms of the function it provides to the community or ecosystem.
	Species type*	Eliciting preferences for a specific species
Diversity	Diversity indices	Indices which describe species diversity at a site or within a specific habitat. Diversity indices included alpha, beta, and gamma diversity as well as Shannon, Simpsons, Berger-Parker indices and their adapted versions.
	Functional diversity	Functional diversity measures the number of functions performed in the ecosystem or community. Functional diversity provides an indication of the richness of functional groups within the site or habitat of interest.
	Genetic diversity	Genetic diversity is the total number of genetic characteristics within a population or species of interest.
	Phylogenetic diversity	Measures the phylogenetic differences between the species within a site or habitat. May be useful for describing

(continued on next page)

Table 1 (*continued*)

Category	Sub-Category	Definition within this review
		the overall number of phylogenetic groups representative in a community
Habitat	Habitat attributes	For the purposes of this review, habitat, included metrics of habitat or vegetation features, and/or condition or quality. These included vegetation cover, habitat type, habitat condition or quality, and individual habitat features such as number of large trees or hollows.
	Distributions	Species distributions were often used to describe the patterns of species occurrence or likelihood of occurrence in an area based on habitat and environmental variables (e.g. species presence/absence, species ranges)
Richness	Richness	Number of species or a given taxa present at a site
	Taxonomic richness	Number of taxonomic groups present
Abundance and richness	Abundance and richness*	A measure simultaneously incorporating both changes in abundance and richness of taxa
Other	Complementarity	A measure of the degree to which a site complements other sites in terms of the defined conservation target or objective. In conservation planning this was often used to find a set of sites which protects as many species as possible while maximizing the most efficient cost allocation or protecting the most endangered species.
	Disturbance	Measurement made to describe a disturbance to the area or site of interest. These may have included measures such as time since last fire, and natural disasters or anthropogenic impacts such as development, regulated harvesting or climate change.
	Rarity/ threat/ persistence	Whether species is endangered, threatened, rare or at risk from catastrophic disturbance
	Uncertainty	Uncertainty was a very broad measure used across the literature. Generally, it was included to measure uncertainty of success of conservation actions or offsets, and uncertainty within data samples or models.
	None/Other	Any other measures or metrics not covered in the above list were noted. Most were not directly relevant to biodiversity, for example, cost.

valuation projects focusing on biodiversity has risen over time (Fig. 2) and is dominated by willingness-to-pay studies (93 %; 4 % estimate willingness-to-accept; 1 % estimate both; the remainder are non-monetary). Countries with the most studies recorded (12–17) included England and Spain. Countries with the second most studies recorded (7–11) included United States, China, France, Germany, Finland, Italy, Japan, Norway, Poland, and Sweden (Fig. 3). Broadly the studies collected covered a wide distribution around the world. However, it is noted that the overlap between countries publishing valuation studies and countries which are in the 36 hotspot regions of the world (Hoffman et al., 2016; Macdonald et al., 2020) is very small or not existing for some hotspots of biodiversity, in particular for East Melanesian Islands, Madagascar, New Caledonia, and Wallecea (Fig. 3 and Table 2). On the other hand a number of valuation studies are from countries in the Mediterranean Basin.

The share of studies applying choice experiments is 53 % and contingent valuation studies is 46 %, with the remainder conducting both. Stated preference studies have predominantly focused on temperate forests, coastal ecosystems, and agricultural landscapes. When studies reported which conservation intervention the values were elicited to inform, the most common conservation interventions were extending or increasing the level of protection of protected areas, or changing land use policies, which would have implications for how ecosystems were managed on the ground (see Supplementary Information).

3.2. Comparing conservation and valuation metrics

Marshall et al. (2020) produced 24 classes of biodiversity metrics based on the conservation literature. Here, we applied this structure for categorising which metrics were applied in the environmental economic literature on valuation of biodiversity. We compared the frequency of metrics in the conservation science and ecology literature (based on data developed in Marshall et al. (2020)) with the occurrence in environmental economic literature (Fig. 4). To do so we first examined the data from Marshall et al. (2020) where each assessed paper was categorised as an ecology or conservation science paper. We determined which of the 24 classes of biodiversity metrics were applied in these papers. We then assessed which of the 24 classes of biodiversity metrics were included in the environmental economic papers and estimated their percentage share within this category of literature. Similarly, we estimated the percentage of biodiversity metrics for the two other categories of literature. Let $j = 1,..,J$ denote the three categories of ecology, conservation science and environmental economics, and $i = 1, \ldots, I$ the 24 classes of biodiversity metric. The percentage share of each biodiversity metric, i, within a literature category, j, is denoted $s_{j,i}$.

We then estimated the normalised Euclidean distance between the ecology, conservation, and environmental economic literatures as the square root of the sum of squared differences between the standardised scores across all 24 classes, I, of biodiversity metrics for pairs of the three categories of literatures (Eq. (1)). We estimate the distance between literature category $k \in J$ and another of the remaining two categories, $j \neq k \in J$:

$$d_{k,j\neq k} = \sqrt{\sum_{i\in I}\left(s_{k,i} - s_{j\neq k,i}\right)^2} \tag{1}$$

The smaller distance the more similar are the number of studies across categories. If the percentage share within a biodiversity metric is similar across all literature categories the distance is zero. The maximum distance is achieved when all percentage shares, except for three metrics, are similar, and the three metrics are different and each represent 100 % of biodiversity metrics within a literature category. In this case the distance between two literatures would be $\sqrt{(100-0)^2 + (100-0)^2 + (0-0)^2 \ldots + (0-0)^2} = 141.42$. We found the smallest distance (19.15) between ecology and conservation science literatures, and the largest distance between ecology and environmental economic literature (26.73). The distance between environmental economic and conservation science was marginally smaller (24.43) than the distance between ecology and environmental economic literature.

A key finding is that there is strong overlap between the most common types of metrics used in conservation and ecological science, and environmental economics (Fig. 4). The most common metrics across both samples were the easily observable characteristics of biodiversity, including species distributions (commonly captured in the environmental economics literature by presence/absence), habitat area, habitat attributes such as habitat type, richness, and abundance (see also Supplementary Information).

However, there were sets of metrics encountered in the ecological and conservation literature which never appeared in the economics literature. These metrics included phylogenetic distinctiveness, functional distinctiveness, diversity indices, functional diversity, phylogenetic diversity, complementarity, disturbance, and uncertainty (Fig. 4). The general pattern was that such metrics, which are capturing complex

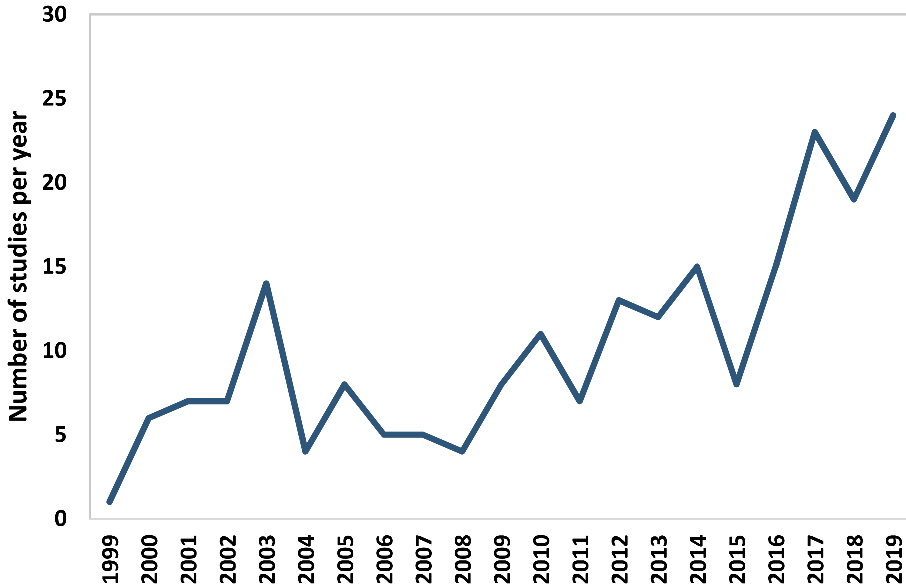

Fig. 2. Number of valuation studier per year focusing on biodiversity.

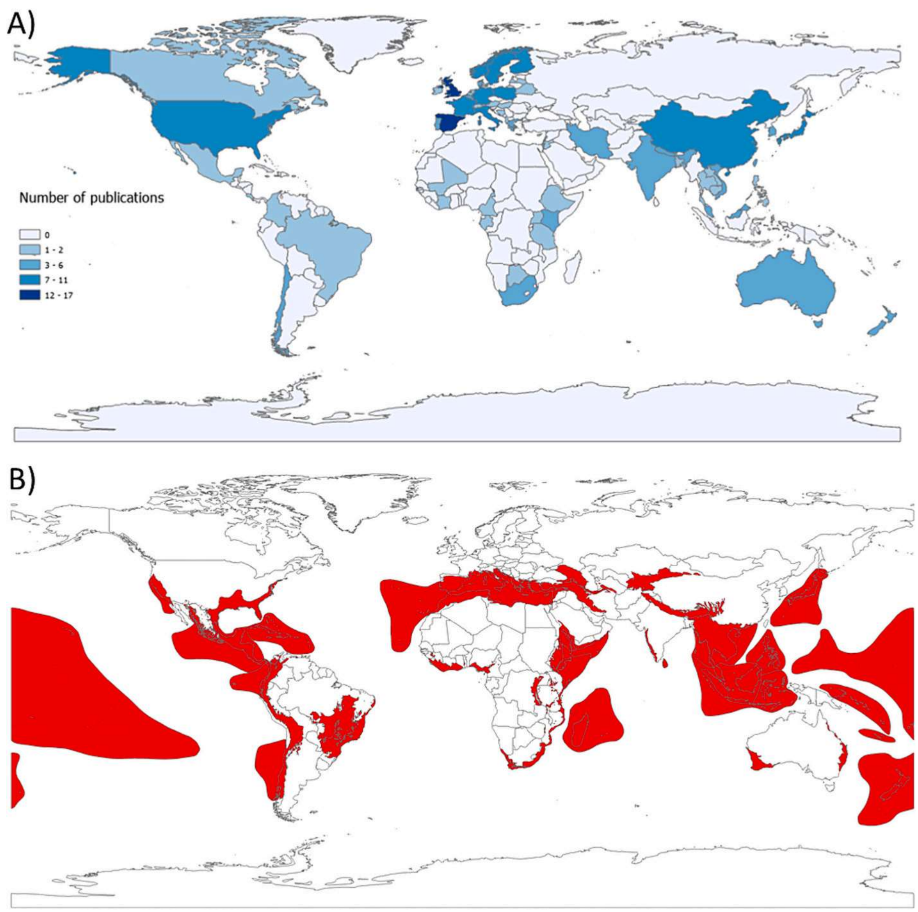

Fig. 3. A) World map of the distribution of studies evaluated across countries. B) Map of 36 recognised biodiversity hotspots, which have at least 1500 endemic species of vascular plants, and have lost at least 70 % of their primary native vegetation. The biodiversity hotspot polygons have been downloaded from the Critical Ecosystem Partnership Fund (CEPF)'s website (source: Hoffman et al. (2016)).

dimensions or ecological properties of biodiversity were absent from the stated preference literature. This implies that there are multiple dimensions of biodiversity that are not being captured by stated preference studies, and therefore components of the non-use value of biodiversity which are not being appropriately accounted for in valuation studies or associated cost-benefit analysis.

The applied search strategy may not cover all studies related to biodiversity, as the search was restricted to studies found in Scopus,

Table 2
Overlaps between 36 recognised biodiversity hotspots and countries which include valuation studies.

Hotspot	Countries (no. of publications)
Atlantic Forest	Brazil (2)
California Floristic	Mexico (1) United States (11)
Cape Floristic Region	South Africa (5)
Caribbean Islands	Barbados (2) Jamaica (1) Netherlands Antilles (2) United States (11)
Caucasus	Iran (4)
Cerrado	Brazil (2)
Chilean Winter Rainfall and Valdivian Forests	Chile (3)
Coastal Forests of Eastern Africa	Kenya (3) Tanzania (1)
East Melanesian Islands	
Eastern Afromontane	Ethiopia (2) Kenya (3) Tanzania (1) Uganda (2)
Forests of East Australia	Australia (6)
Guinean Forests of West Africa	Cameroon (1)
Himalaya	Bangladesh (1) China (10) India (5) Nepal (3)
Horn of Africa	Ethiopia (2) Kenya (3)
Indo-Burma	Bangladesh (1) Cambodia (1) China (10) India (5) Laos (1) Malaysia (3) Thailand (1) Vietnam (4)
Irano-Anatolian	Iran (4)
Japan	Japan (7)
Madagascar and the Indian Ocean Islands	Mauritius (1) Seychelles (1)
Madrean Pine-Oak Woodlands	Mexico (1) United States (11)
Maputaland-Pondoland-Albany	South Africa (5)
Mediterranean Basin	Albania (1) Bosnia and Herzegovina (1) Cyprus (1) France (10) Greece (3) Israel (4) Italy (8) Jordan (1) Lebanon (1) Portugal (3) Spain (17)
Mesoamerica	Colombia (1) Costa Rica (1) Mexico (1)
Mountains of Central Asia	China (10)
Mountains of Southwest China	China (10)
New Caledonia	
New Zealand	Australia (6) New Zealand (3)
North American Coastal Plain	Mexico (1) United States (11)
Philippines	Philippines (1)
Polynesia-Micronesia	Chile (6) United States (22)
Southwest Australia	Australia (6)
Succulent Karoo	South Africa (5)
Sundaland	India (5) Malaysia (3) Thailand (1)
Tropical Andes	Chile (3) Colombia (1)
Tumbes-Choco-Magdalena	Colombia (1)
Wallacea	
Western Ghats and Sri Lanka	India (5)

which mention biodiversity and choice experiments or contingent valuation in the title, abstract or keywords. Thus, it would be more correct to state that the search included stated preference based valuation papers, which have an emphasis on biodiversity. Given the high number of papers that were reviewed (a total of 214 papers, which is a relatively high sample size for reviews of this type) it is assumed the search strategy captures a large proportion of trends in biodiversity metrics applied within environmental economic valuation literature. It may be expected that studies focusing on single species or studies focusing on assessing nature conservation in a broad sense would be underrepresented. However, with the selection of studies including the term biodiversity, it is assumed that the most detailed metrics used on exactly this concept are included.

3.3. Implications for stated preference studies

The focus on the easily measurable and accessible components of biodiversity in stated preference studies is understandable given the need to have an identifiable biodiversity attribute to which participants can relate. However, the focus in environmental economics on species richness and easily observable aspects of biodiversity is potentially problematic (Fleishman et al., 2006). Conservation is increasingly moving away from species richness as an indicator in recognition that it is limited in its ability to account for turnover in species identity, their relative rarity, ecological characteristics or persistence (Chiarucci et al., 2011; Fleishman et al., 2006).

In the ecological literature there are increasing efforts to move towards holistic monitoring of biodiversity by tracking six classes of Essential Biodiversity Variables, enabled by technological advances in remote sensing, genetic analysis and monitoring (Turak et al., 2017; https://geobon.org/ebvs/what-are-ebvs/). These EBVs include 1) genetic composition (not covered in the review of environmental economic literature), 2) species populations (covered), 3) species traits (not covered), 4) community composition (slight overlap), 5) ecosystem functioning (slight overlap), and 6) ecosystem structure (slight overlap). In general, many of the essential biodiversity variables are not captured within our dataset of environmental economic studies. This is especially problematic if we are interested in analysing value development over time. One suggestion would be that future biodiversity valuation studies contain a suite of attributes recommended by conservation science such as the Essential Biodiversity Variables.

3.4. Complexity and public understanding biodiversity metrics

We would argue that including multiple, complex dimensions of biodiversity in environmental economic valuation studies would capture a larger set of the total value of biodiversity. Despite these benefits, more complex metrics may be constrained by what laypeople are familiar with and the fact that what is explained should also capture what they perceive as valuable. Characterising and describing the dimensionality of biodiversity may be rather complex. Although a comprehensive description of the biodiversity metrics/attributes may make scientific sense, it is important that the metrics are compatible with people's mental constructs of biodiversity (Bakhtiari et al., 2014). Yet, at the same time, we know that this is highly dependent on knowledge (Hanley and Perrings, 2019; Lundhede et al., 2014). Some metrics may appeal more to lay people, e.g., species richness and populations, however as Hanley and Perrings (2019) argue, biodiversity metrics should not only focus on species and habitat characteristics but also on functions of diversity. Hence limiting the number of metrics used in choice experiments (or contingent valuation) studies to only species numbers for example would not holistically capture the value the public has for biodiversity. It remains for future studies to understand if including more complex biodiversity metrics are realistically and contributes significantly to capture the total economic value of biodiversity. Thus, there is a trade-off between including a science-based and comprehensive description of biodiversity in valuation surveys and a description, which are understandable by lay people and which captures people's values.

One solution for optimising trade-offs between ecological complexity and comprehensibility is to implement more deliberative processes that explore which aspects of biodiversity are most valued by participants and align these within stated preference studies. Various methods have been proposed, such as citizen juries (Geleta et al., 2018), which attempt to allow for inclusion of a more complex set of attributes in choice experiments, alongside citizen preferences which are more relevant in social decision making than individual preferences (Alvarez-Farizo and Hanley, 2006). To ensure that such a description also captures what people consider as biodiversity value, i.e. that the excessively informed respondents in a citizen jury represents the larger population, effort should be made to more thoroughly understand people's perceptions.

Detailed qualitative and semi-quantitative work could be conducted to explore these perceptions. For example, the Q methodology (Stephenson, 1982) is increasingly applied to study understanding and value formation of complex terms where there are multiple viewpoints on a topic. The Q methodology consists of first identifying the possible viewpoints or perceptions of a given topic, ranking the statements, and finally conducting an analysis of patterns of opinions. Austen et al. (2021) used the Q methodology to explore the specific components of

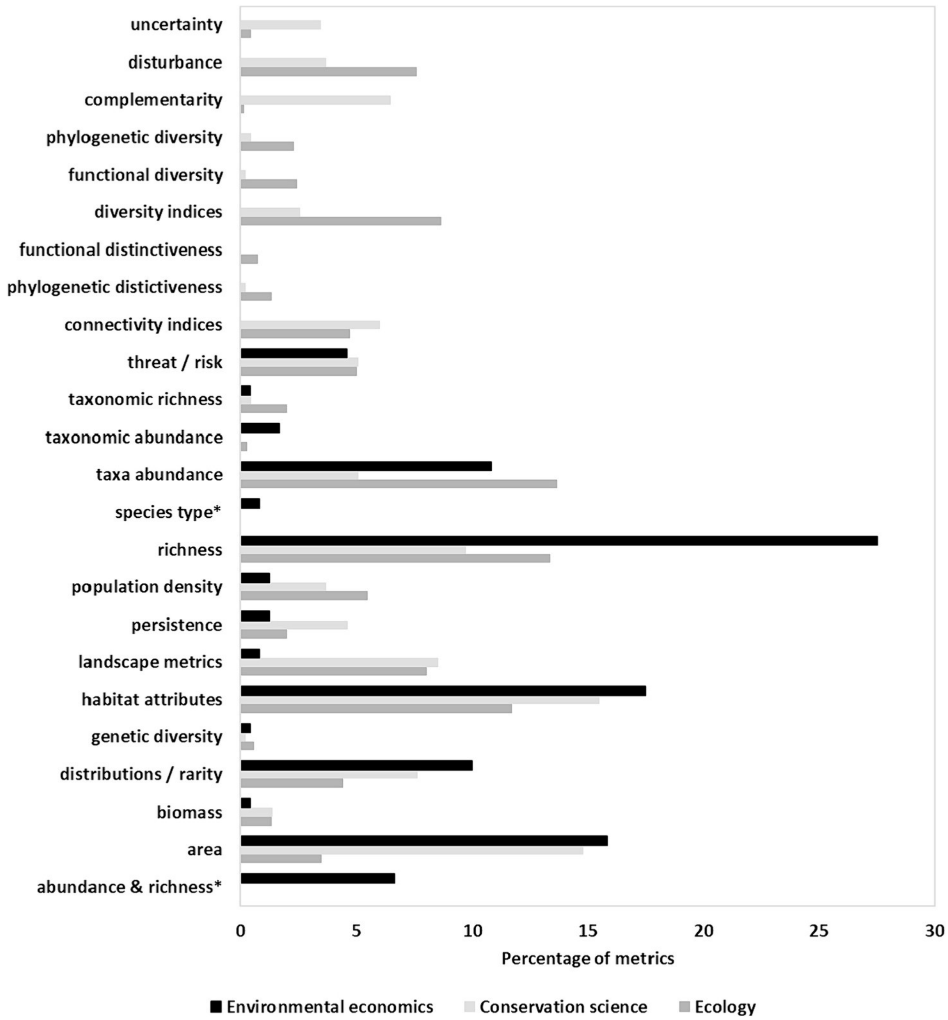

Fig. 4. Applied biodiversity metrics in ecology, conservation science, and environmental economics. Data from studies in ecology and conservation science were categorised in Marshall et al. (2020), and the data from environmental economics were categorised here. Categories marked with * denote metrics that were only found in the environmental economics literature and not in the ecological literature (i.e. in Marshall et al., 2020).

nature that laypeople derived wellbeing from, including various sensory experiences and types of wildlife. Qualitative understanding of laypeople's perceptions could be used to derive biodiversity measures that closely resemble the emotional relationships people have with nature, thereby providing a more complete representation of their non-use values. Apart from constituting a value itself, it could also result in better scenario descriptions in valuation studies where quantitative economic measures are sought. Likewise, deliberative valuation may be a way to ensure that shared and social values are better incorporated in the estimates of non-use values of ecosystems and biodiversity (Kenter et al., 2016).

3.5. Using technological advances to capture a wider range of biodiversity metrics

Advances in remote sensing, genetic and monitoring technologies (e.g. eDNA) are allowing for tracking different aspects of biodiversity in higher resolution and with greater frequency than ever before. These allow for quantification of dimensions of biodiversity that have previously been impractical to measure. These also create a justification for broadening out the dimensions of biodiversity that are captured in stated preference studies. Many of these complex components of biodiversity may well have human welfare implications, but there have simply never been the capabilities to integrate these metrics into choice experiments. For example, phylogenetic diversity was not investigated in our sample even though people may care about phylogenetic diversity as it captures evolutionary history and indicates a variety of forms and functions are present within a species set (Mazel et al., 2018). eDNA monitoring reports now have the capacity to make genetic information about the biodiversity present in the natural environment available to the public. Such methods could well be integrated into choice experiments to capture previously unmeasurable preferences for new dimensions of biodiversity.

Additionally, technological innovation within choice experiment settings may also enable us to measure preferences for a wider variety of components of biodiversity than ever before. Most choice experiments work with drawings, pictures or videos as means for communicating how landscapes and land use change would look under different scenarios (Bateman et al., 2009; Nielsen et al., 2007). Visualization techniques may improve respondents' interpretation of complex information and decrease choice uncertainty. Uggeldahl et al. (2016) applied eye-tracking techniques to show that use of pictures in choice experiments may reduce complexity in the choice sets and increase certainty compared to use of text descriptions. Recently, Virtual Reality (VR) has been applied in scientific experiments within e.g. psychology (Wilson and Soranzo, 2015), architecture (Portman et al., 2015) and environmental economics (Mokas et al., 2021) to convey complex information and associated responses. The findings in Mokas et al. (2021) document that VR further reduce uncertainty compared to videos and pictures, as well may even reverse the assessments of attributes in some cases. This may be a result of VR providing a more realistic presentation of the decision context than a description in a survey.

It is also well-known that, environmental valuation studies should aim to reduce information bias, meaning that preferences are sensitive to information resulting in warm-glow, anchoring effects, or cognitive fatigue etc. (Andreoni, 1995; Needham et al., 2018). Furthermore, it has been found that information and knowledge is of large importance for value assessment (Lundhede et al., 2014; Needham et al., 2018). Therefore, contextual information should be at a minimum yet sufficient level, which allows the individual to express preferences without being too influenced by context description. Determining the appropriate level of detail is a challenge. As part of the contextual setting it is important that the specific mechanism for achieving biodiversity change is justified. In our sample of stated preference studies, more than 65 % of our included environmental economic studies specify a mechanism. The remaining 35 % are suggesting general and unspecific mechanisms, e.g. such as changes in management regimes, sustainable tourism, land use policy. If valuation studies are to be used to evaluate socio-economic gains from various conservation interventions, then it is important that the scenarios/policies formulated in the valuation studies are i) based on a mechanism that is consequential, ii) but with separate potential co-benefit and iii) based on a sufficient natural scientific basis. Ideally, the valuation studies would be formulated on the basis of state-of-the art quantitative conservation science approaches to modelling biological effects (consequences for biological metrics) of different scenarios. Additionally, we found that the majority of valuation studies did not shed light on the biological context, and if they described the biological context, then most studies did not document on what scientific basis they found the described effects (outcomes). While it could, in principle, still be part of the background work of the survey design, it is a pity when not reported in the studies as it limits its applicability in integrated ecological-economic modelling approaches.

3.6. Ethics and biodiversity's contribution to people

An advantage of simplifying biodiversity into quantifiable metrics of biodiversity is that it may more easily be translated into policy guidance and evaluation. This requires that the metric captures all dimensions of human values of biodiversity. Bakhtiari et al. (2014) estimated the willingness to pay for biodiversity protection in Sweden and Denmark. However, the study demonstrated that public preferences for biodiversity protection was much more than just a concern for protecting species. The respondents expressed concerns for maintaining forest ecosystem resilience and less intangible measures such as ecosystem intactness. The more reductive we are, the less likely we are to capture more worldviews and morally difficult issues, such as bequest values and altruism (Hanley and Milne, 1996). The diversity of values among different stakeholders, importance of indigenous and local knowledge, have been recognised in the work by the IPBES and others through their development of the framework 'nature's contributions to people (Ellis et al., 2019; IPBES, 2022; Pascual et al., 2017). They suggest a six steps approach to valuation including i) identify the purpose, ii) scope the process, iii) valuation methods, iv) integration, bridging and up-scaling, v) communicate, and vi) review the process. Furthermore, the choice of valuation method should be guided by the purpose of the valuation, i.e. if it should be used in decision making, raising awareness or informing, accounting or for litigation for environmental liabilities and conflict resolution (Pascual et al., 2017). It has also been argued that this new framework should help overcome existing power asymmetries between western science and indigenous and local knowledge (Díaz et al., 2018). Our review found that a majority of the environmental economic studies were focused in wealthy nations. It is found within other fields that the scientific production of knowledge may show a geographical bias against the developing and more vulnerable regions of the world (Pasgaard et al., 2015; Pasgaard and Strange, 2013). We would argue that this may hide knowledge gaps and contribute with biases in understanding the preferences and values of biodiversity. Research initiatives targeted to enhance research in biodiversity values, including the less tangible ones, in emerging economies and the global south may help diminishing such global knowledge divides. We also found that the overlap between countries publishing valuation studies and countries which are in the 36 hotspot regions of the world, except for the Mediterranean Bassin, is generally very small or even not existing for some of the hotspots. This is particular the case for the hotspots East Melanesian Islands, Madagascar, New Caledonia, and Wallecea (see Fig. 3 and Table 2). This lack of geographical overlap represents a severe knowledge gap, which may reduce the possibilities for making quantitative economic estimates of the value of biodiversity protection in vulnerable parts of the world. Future environmental valuation research is encourage to focus more on these hotspot regions.

4. Conclusion

Based on Bartkowski et al. (2015), Farnsworth et al. (2015) concluded stated preference techniques have failed to measure the value of biodiversity. They argued that the value of biodiversity should be derived indirectly by the functional relationship between biodiversity and ecosystem services, whenever it can be quantified. However, many valuation studies of biodiversity include both use and non-use values (Hanley et al., 2003; Jacobsen et al., 2011, 2008; Jacobsen and Thorsen, 2010) and often find the non-use values are larger than use values. Thus, functional relationships may not fully capture the value of biodiversity to people. We found in this review some overlap between the biodiversity metrics applied in conservation science and environmental valuation studies, but also there is little overlap between more complex dimensions of biodiversity. In a more recent paper Johnston et al. (2017) propose recommendations for best practice stated preference studies that aim at providing information for decision-making. In addition to such guidelines we suggest surveys may be improved by formation of collaboration and interdisciplinary research teams including biodiversity valuation experts and conservation scientists providing a more comprehensive understanding of the ecological dimensions of biodiversity and its valuation. Practically, this could facilitate inputs to the design of valuation surveys.

We realise that biodiversity is an inherently complex concept and that survey respondents in biodiversity valuation studies have limited cognitive capacity and attention span. However, stated preference studies assessing values of biodiversity need to simplify the way impacts on biodiversity are communicated to respondents while still capturing as many dimensions of biodiversity as possible. This is a trade-off where systematic studies are still lacking. To seek alignment with the conservation science, valuation studies should aim should be to provide realistic context, where the alternative scenarios and specific mechanisms for achieving biodiversity change are science-based or based on quantitative predictions of conservation effects. Technological advances may both contribute to improving realism and allowing for tracking a wider variety of biodiversity components over time. Overall, this may contribute to quantification of dimensions of biodiversity that have previously been impractical to quantify and improving survey respondents' interpretation of complex information within the field of biodiversity valuation.

Another approach is to incorporate long-term perspectives in both biodiversity valuation and conservation science, recognising the dynamic nature of ecosystems and the services they provide. Long-term monitoring and modelling can help capture the potential impacts of conservation actions and enable more accurate projections of costs and benefits over time. While the conservation science literature has had a focus on observing changes over time, this is less so in the valuation literature. If we want to follow value development over time, as we do with the increasing attention of natural capital accounting, the most feasible approach would be to continue with relatively simple context independent measures, like number of species or habitat coverage. However, this risk not capturing the full aspect of biodiversity. This speaks in favour of linking the valuation of biodiversity more closely to more complex conservation measures as the EBV. At the same time, it is important to link EBVs to studies of what constitutes value to people. Values are likely to change over time in aspects not related directly to the biodiversity, but to broader societal changes such as substitution possibilities, income/wealth, education, and cultural change. All of these changes are likely to become important in future natural capital accounting.

Author statement

The authors have nothing to report. We have not applied generative AI in scientific writing.

CRediT authorship contribution statement

Niels Strange: Writing – review & editing, Writing – original draft, Visualization, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization. **Sophus zu Ermgassen:** Writing – review & editing, Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Erica Marshall:** Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Joseph W. Bull:** Writing – review & editing, Writing – original draft, Methodology, Funding acquisition, Conceptualization. **Jette Bredahl Jacobsen:** Writing – review & editing, Writing – original draft, Validation, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

I have shared data in attached supplementary files (1, 2, and 3)

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