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Research paper on habitat and biodiversity related ecosystem services

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The first version of this paper was circulated to the participants of the Expert Meeting on Advancing the Measurement of Ecosystem Services for Ecosystem Accounting, New York, 22-24 January 2019. Comments received during and immediately after the Expert Meeting have been addressed by the authors and the paper was revised.

This research paper evaluates the extent to which biodiversity, and the provision of habitat supporting biodiversity, constitutes an ecosystem service. The paper considers the connection between measures of ecosystem services flows in relation to key service-providing species, or to whole sets of species, and measures of biodiversity within a given ecosystem or a complex of ecosystem types. In this context, this paper reflects on whether biodiversity is, itself, an ecosystem service and how this should be best considered in an ecosystem accounting perspective.

Options for the bio-physical measurement of current ecosystem service flows and the capacity for future flows are suggested. Where appropriate, these are supported with options for monetary valuation of ecosystem service flows and the capacity to maintain delivery of these services over time.

Where possible this paper is supported with case studies from the literature and a list of key literature is also provided. Key unknowns and proposed areas for future research for this particular ecosystem service are also identified.

This paper comprises one of a set of ten papers on specific ecosystem services or themes. Coordinating authors for these ten papers convened at a meeting hosted by the UN Statistics Division in New York, 22nd – 24th January 2019 to discuss the full set of papers. During this meeting feedback on this paper was received. This has been addressed to the degree possible in this final version. Outstanding feedback that could not be researched further is summarised below:

- On the potential value of bio-prospecting. It was noted that past studies had identified this to be quite low. However, we would note that these values have fluctuated considerably over the years and that the substantially reduced costs and time needed for gene sequencing means this is now a considerably more financially viable activity. As such further investigation of these values is needed to establish their current significance.
- It was highlighted that biodiversity is a characteristic of ecosystems or representative of a portfolio of assets. As such it should not be viewed as an asset in its own right. We have added further text to elucidate on our treatment of biodiversity as an asset. This sets out our view that it represents an emergent property of a set of ecosystem

assets (and the community assemblages within them) that interact and support multiple ecosystem processes. These, in turn, underpin the resilience of the delivery of ecosystem services at larger scales. We also highlighted that the thematic biodiversity account in the SEEA EEA is derived from the SEEA CF asset account structure. This may also lead to confusions when discussing biodiversity in this context.

- Additional measurement approaches suggested in the meeting for habitat and biodiversity related ecosystem services that could be evaluated further included:
    - Use of nature conservation volunteering time as a measure of the bequest value;
    - Using reserve / protected area planning models (e.g., Marxan) to identify the minimum costs of protecting a set of species and habitats;
    - Use of marginal values for flagship species derived from collective conservation actions (a related approach is presented in this paper with respect to Orangutan reintroduction);
    - Use of compliance costs (or opportunity costs of land) associated with implementing a countries biodiversity policies and any associated subsidies.
- It was highlighted that undertaking stated preference surveys were expensive and that there remain significant challenges in integrating these welfare based values in to the SEEA. We also acknowledge this as a further area for research.
- An important point was made with respect to identifying where the demand for many of the habitat and biodiversity related ecosystem services is. Particuarlaly, do people near the resource (who typically bear opportunity costs) receive benefits form these services?

Conceptual framing for habitat and biodiversity related ecosystem services

As in previous SEEA-EEA documentation, we here adopt the Convention on Biological Diversity's (CBD's) definition of biodiversity as: *"the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems"* (https://www.cbd.int/doc/legal/cbd-en.pdf). Note the strong emphasis on biological "variability" or "diversity" in this definition, an emphasis previously highlighted by the SEEA Experimental Ecosystem Accounting: Technical Recommendations report (2018).

This emphasis has important implications for any consideration of the relationship between biodiversity and ecosystem services in ecosystem accounting. Such consideration needs to focus specifically on the potential values of, or roles played by, diversity itself at multiple levels of biological organisation, or at least of the components comprising this diversity (e.g. individual species as components of species diversity). We therefore regard as 'out of scope' any assessment of services as a function of the overall amount of biological material within an ecosystem (e.g. biomass), or the overall functioning of that system (e.g. primary productivity), without explicitly considering the diversity of organisms underpinning these system-level attributes. Such services are addressed by other papers in this series.

A similar caveat applies here to our use of the term "habitat" which is defined by the CBD as *"the place or type of site where an organism or population naturally occurs"* (https://www.cbd.int/doc/legal/cbd-en.pdf). A forest, for example, qualifies as habitat only in the sense that it provides suitable conditions for particular organisms to live - i.e. it supports the existence and persistence of biodiversity. We therefore regard the role that this same forest might play in, for example, sequestering carbon as a function of the overall biomass and/or functioning of this ecosystem as, again, falling outside the scope of “habitat and biodiversity related ecosystem services”.

People value biodiversity in many different ways, and a plethora of conceptual frameworks, classifications and typologies have been proposed over the past two decades, in an attempt to make better sense of this complexity. Here we adopt the relatively simple typology proposed by Bolt et al. (2016) (Figure 1) to frame our discussion of biodiversity values in an ecosystem accounting context. However, in doing so, we cross-reference this typology to two other prominent typologies: 1) the well-known classification of ecosystem services employed originally in the Millennium Ecosystem Assessment (MA, 2005), and later adapted and extended in the Common International Classification of Ecosystem Services (CICES v 5.1); and 2) the relatively recent typology of “Nature’s contributions to people” proposed by the Intergovernmental Platform for Biodiversity and Ecosystem Services, IPBES (Pascual et al 2017). (See Figure 1)

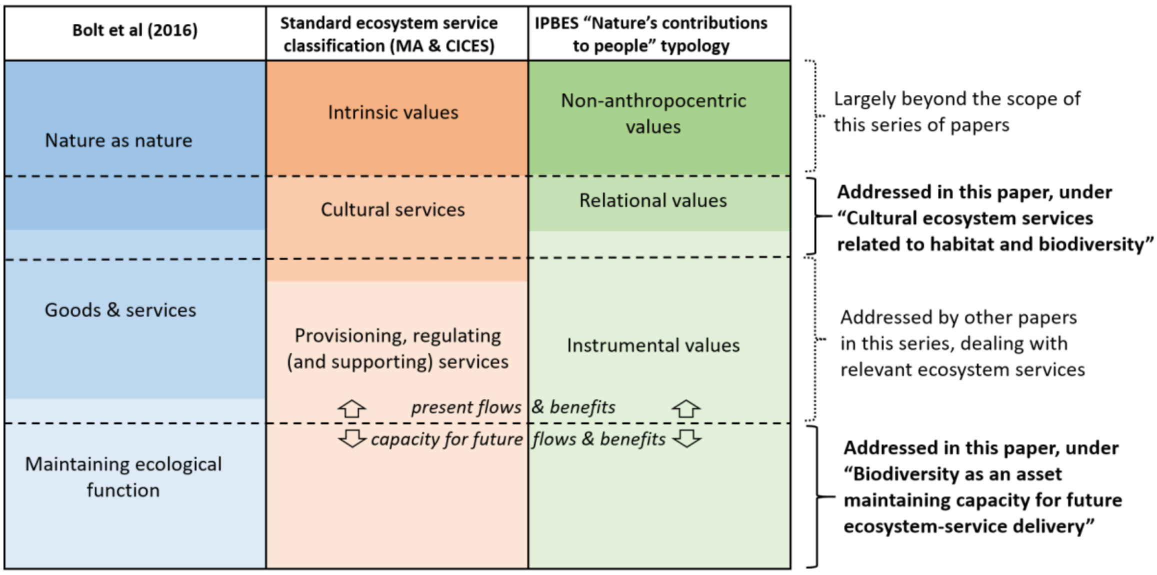

Figure 1: Cross-linkages between major typologies, or classifications, of biodiversity values and associated services, and the topics addressed in this paper

Goods and services: These are the established focus of ecosystem accounting. They are manifold and include goods from the direct use of biodiversity, supported by the presence of suitable habitat. For example, habitats that support species that provide wild foods, materials or cosmetics (i.e., provisioning services). They also include the inputs that biodiversity provides in the production of these types of goods, as well as its role in mediating environmental conditions and ecological processes (i.e., regulating services). For example, seagrass habitat providing nurseries for commercial fisheries and forest providing habitat for pollinators of crops. Examples of these types of consumable ecosystem goods and services are given detailed consideration in other papers in this series. As such they are not considered further in this paper.

These ecosystem goods and services also include inputs that are not physically consumed but do support some form of in-situ interaction. These form part of the bundle of **Cultural Ecosystem Services** and include recreation, education and aesthetic appreciation of habitat and biodiversity and the benefits this brings. For example, national parks that provide habitat for iconic species that attract tourists and habitats that support species that provide opportunities for education. Our treatment of cultural services in this paper focuses on those services not already addressed in the papers on "Nature based tourism related services" and "Urban recreation related services".

Nature as nature: This relates to the way that stocks of habitats and biodiversity are perceived by people (either in-situ or remotely). They include public preferences for maintaining the extent and condition of key habitats and addressing species population loss because they value its existence for themselves and/or wish the benefits biodiversity provides to be available to others (bequest value) now and in the future. They also include the symbolic and spiritual recognition of habitats and biodiversity people may have, such as national emblems or totemic species. In addition, they also provide material for entertainment, such as nature films. Many of these values can also be viewed as forming part of the **Cultural Ecosystem Services** bundle provided by habitats and biodiversity, and are therefore addressed as such in this paper. However, stricter non-anthropocentric perspectives on the intrinsic value of biodiversity are considered as falling outside the scope of this, or any other paper, in the series (Figure 1).

Maintaining ecological function: Another crucial value of biodiversity highlighted in Figure 1 is its importance in the maintenance of ecological systems and functions that underpin the ongoing delivery of ecosystem goods and services into the future (Bolt et al., 2016). For example, the role of biodiversity in cycling energy, nutrients and other materials through the environment. Maintaining a diversity of habitat types and communities of species is key to sustaining healthy ecological functioning. In this context the *resilience of ecosystems* to tolerate shocks and disturbance yet maintain the same level of ecological functioning is prime concern (e.g., in the context of climate change). Different habitats and species can contribute to ecological functioning in similar ways, yet respond differently to environmental change, thereby providing "insurance value" (Elmqvist et al., 2003). As such, maintaining a diversity of habitat types and species is also crucial to maintaining the resilience of ecosystems over the longer term. We here combine consideration of this perspective with that of another forward-looking perspective on the contribution of biodiversity to maintaining capacity for future service delivery, i.e. "option value". These forward-looking perspectives on the value of biodiversity as an asset are generally missing from the broad assessment of ecosystem services (although it should be noted that some typologies treat option value as a cultural ecosystem service – see, for example, the CICES classification of cultural services in Table 1). Part of this stems from the need to focus on final ecosystem services and avoid issues of double counting that may emerge when including the value of this aspect of biodiversity ecosystem service accounts (Fisher, Turner and Morling 2009). Nonetheless, biodiversity is clearly a critical asset for maintaining the capacity of ecosystems and ecosystem complexes to deliver goods and services into the future and this should be a fundamental ecosystem accounting concern.

Cultural ecosystem services related to habitat and biodiversity

This paper aims to provide tangible guidance on defining and measuring the **cultural ecosystem services related to habitat and biodiversity** (in-situ and remote), as described above. Where possible the description of ecosystem services is provided from the perspective of the potential users of the services (e.g. households) and the suppliers (e.g., National Park Authorities and other land managers) to inform a more concrete view of ecosystem service transactions. This is supported with options for monetary valuation where appropriate.

The Common International Classification of Ecosystem Services (CICES v 5.1) provides a useful structured approach for identifying and describing these cultural ecosystem services in tangible terms (see Haines-Young & Potschin, 2018). Drawing on CICES, Table 1 provides a summary of the Cultural (Biotic) ecosystem services and derived benefits that are provided by stocks of biodiversity and habitat.

The final column in Table 1 sets out examples of the tangible benefits the use of these ecosystem services realises. These will be enjoyed by a set of final beneficiaries, typically characterised as individuals, families (or households) that have an interaction or appreciation of the stocks of biodiversity and habitats that they value. The enjoyment of these stocks indicates that a production boundary has been crossed, from the environmental system in to the socio-economic system (i.e., an actual improvement in well-being is realised).

Table 1: CICESv5.1 Extract for biodiversity and habitat related cultural ecosystem services

Type of interaction	Ecosystem service - stocks of biodiversity and habitat that….	Simple descriptor	Example Service	Example Benefits
Direct, in-situ and outdoor interaction with biodiversity and habitat	….enable activities promoting health, recuperation or enjoyment through active or immersive interactions	*Using the environment for sport and recreation; using nature to help stay fit*	*Ecological qualities of woodland that make it attractive to hiker*	*Recreation, fitness; de-stressing or mental health; nature-based recreation*
	….enable activities promoting health, recuperation or enjoyment through passive or observational interactions	*Watching plants and animals where they live; using nature to destress*	*Whales, birds, seals and reptiles can be enjoyed by wildlife watchers*	*Recreation, fitness; de-stressing or mental health; eco-tourism*
	….enable aesthetic experiences	*The beauty of nature*	*Area of Outstanding Natural Beauty; panorama site*	*Artistic inspiration*
	….are resonant in terms of culture or heritage	*The things in nature that help people identify with their history or culture*	*Sherwood Forest*	*Tourism, local identify*
	….enable scientific investigation or the creation of traditional ecological knowledge	*Researching nature*	*Site of special scientific interest, Natura 2000 site*	*Knowledge about the environment and nature*
	….enable education and training	*Studying nature*	*Site used for voluntary conservation activities*	*Skills or knowledge about environmental management*
Indirect, remote, often indoor interaction with	…. used for entertainment or representation	*The things in nature used to make films or to write books*	*Archive records or collections*	*Nature films*
	…. have symbolic meaning	*Using nature as a national or local emblem*	*Bald Eagle*	*Social cohesion, cultural icon*

biodiversity and habitat	…. have sacred or religious meaning	*The things in nature that have spiritual importance for people*	*Totemic species, such as the turtle*	*Mental well-being*
	…. that have an existence value	*The things in nature that we think should be conserved*	*Areas designated as wilderness*	*Mental/Moral well-being*
	…. that have an option or bequest value	*The things in nature that we want future generations to enjoy or use*	*Endangered species or habitat*	*Moral well-being*

The SEEA EEA (UN et al., 2014) presents a model for the measurement of ecosystem services (Section 3.2.4). This sets the linkages between ecosystem assets, ecosystem services and human wellbeing into a specific context for physical measurement. Annex 3 provides a very relevant model for tourism and recreation ecosystem services. This has been adapted to inform a more generic Logic Chain of these linkages for the ecosystem services considered in Table 1 (see Figure 2).

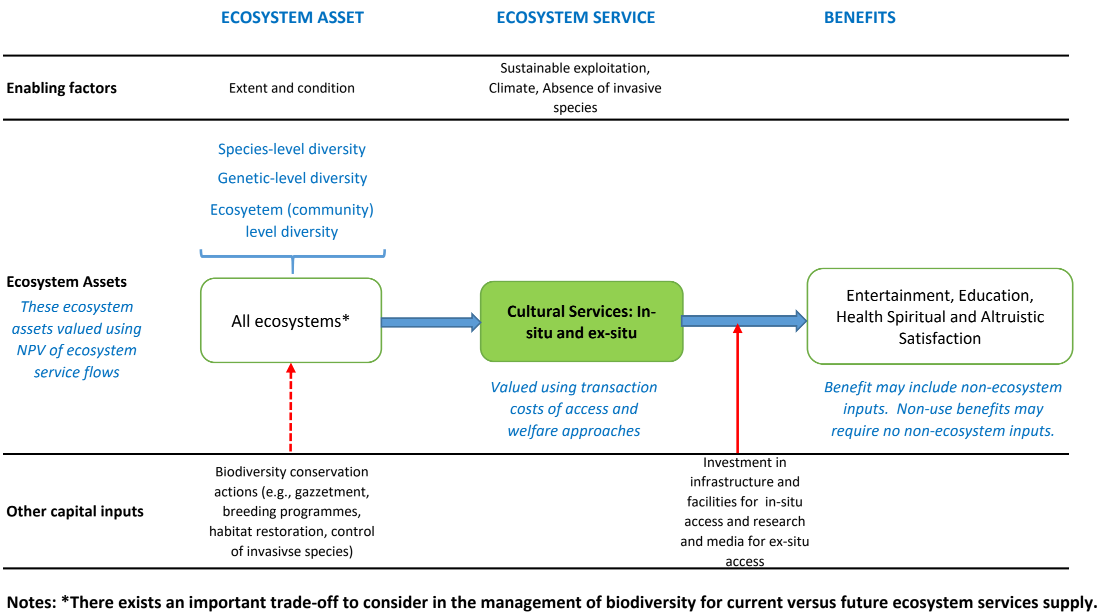

Figure 2: Logic chain for biodiversity and habitat related cultural ecosystem services

With reference to the logic chain, investments are required to conserve and enhance the stocks of biodiversity and habitats that supply the ecosystem services in Table 1. As such the supply of the ecosystem services in Table 1 can be increased by biodiversity conservation activities, such as gazettement, targeted breeding programmes, habitat restoration and eradication of invasive alien species (i.e., other capital inputs). These actions will typically be undertaken by land managers, for example by the department of environment, conservation organisations or other private land owners wishing to realise the benefits listed in Table 1 or responding to subsidies or other incentives (e.g., pursuit of profit) to provide these benefits. In broad terms these types of institutions represent the suppliers of the ecosystem services listed in Table 1.

However, further inputs may be required in order to realise these well-being benefits. These will, largely, be driven by the nature of the interaction. For those related to direct interactions in the top part of Table 1, investment in facilities may be substantial (e.g., hotels, hiking paths, education centres, recreational equipment) and services (e.g., guides, transportation services, catering). For those benefits related to indirect or remote interactions, these inputs may be less substantial (e.g., media production costs for nature documentaries). It may often be the case that the operators providing these inputs are not the same institutions managing ecosystem and biodiversity assets and supplying the ecosystem services themselves. As such, these operators are also beneficiaries of the ecosystem service production process, realising economic benefits associated with selling access to a number of the benefits listed in Table 1.

The management of biodiversity and the habitat that supports it cuts across all ecosystem types. Whilst often a primary concern for natural ecosystems (e.g., ancient forest, savannah and wetlands), it is also highly relevant to managed ecosystems (particularly where there are heavily modified ecosystems, such as in Europe) and urban ecosystems (as there are a large number of beneficiaries from interactions with biodiversity). However, across all ecosystems there exists a variety of pressures on the stocks of biodiversity and habitat. Key enabling factors required to realise the maximum level of benefits listed in Table 1 vary accordingly but include: avoiding overexploitation of these stocks, absence of pollution, suitable climatic conditions and absence of invasive species (see logic chain in Figure 2).

Physical measurement of biodiversity and habitat-related cultural ecosystem services

There are a number of direct and habitat based approaches for measuring the stocks of biodiversity. For instance using surveying methods, species distribution / abundance modelling or modelling approaches based on ecological community metrics. These are discussed in the following section and in further detail in UNEP-WCMC (2016). These can inform on key biodiversity indicators relevant to the all of the ecosystem services summarised in Table 1. This reveals a direct link between thematic biodiversity accounts and ecosystem service supply and use accounts. This reflects that the cultural services in Table 1 are non-material and, generally, non-rival (i.e., one person using the service does not reduce the level of supply to another). Although, it is acknowledged that interactions with biodiversity and habitats can be compromised where there is high demand (e.g., large crowds at a viewpoint) and there may also be wider environmental qualities that enhance experiences, for example topography, climate and absence of litter.

In order to characterise the ecosystem service flow in physical terms a measure is required that tangibly represents the magnitude of the interaction between beneficiaries and biodiversity and habitat stocks (i.e., the level of demand). For direct interactions, this is conceptually reasonably straightforward and can be measured in terms of visit frequency and time spent enjoying the interaction. These types of measure are routinely used in estimating nature based recreation services (e.g., for the recreation / wildlife watching services captured in the top two rows of Table 1, discussed in detail in paper No. 10 of this series). They also lend themselves to characterising the cultural ecosystem service flows derived via aesthetic experiences of nature, experiences in cultural landscapes associated with semi-natural ecosystems, ecological knowledge or education services related to biodiversity and habitats captured in Table 1.

- Number of visits to experience the interaction (e.g., national park visitor counts, number of visits to nature sites of cultural significance, number of students on nature based field trips)

- Time engaged experiencing the interaction (e.g., volunteered hours for conservation activities, hours engaged in on-site ecological research, hours engaged in landscape painting)

For specific sites data could be directly available on the above, or a well correlated observation (e.g., car park vehicle counts). These will support local scale ecosystem accounting applications but several assumptions are likely to be required to upscale data to national levels for different ecosystem types. Typically these will be grounded in the transfer of estimates between similar sites or using a transfer function to adjust measures on the basis of site characteristics. Alternatively, there may be national statistics that can be disaggregated to support physical estimation of these service flows. For example the monitoring engagement with nature reports produced for England and Wales provide official aggregate statistics of people's interaction with biodiversity and habitat (Natural England, 2017). Paper number 10 also highlights the potential use of random utility modelling as a means of estimating the likelihood of visit rates for a given site. These can be based on the sites biodiversity stock, accessibility, facilities and other characteristics.

Social media and mobile technology are also increasingly being used to measure people's direct interactions with biodiversity and habitat. For instance, Hamstead et al., (2018) use Flickr and Twitter data to generate geolocated social media (GSM) indicators that have been used to examine urban park visitation. Cord et al., (2015) use Geocaching (a GPS based game of hide and seek) to explore nature-related recreational ecosystem services. However, further research is required on how these approaches could be applied in an ecosystem accounting context with a reasonable degree of confidence.

Mourato et al., (2010) evaluate the outdoor learning and education as an ecosystem service as part of the UK National Ecosystem Assessment. This seems to be one of the few studies to attempt to qualify and quantify this particular cultural ecosystem service. In terms of physical measurement they employ educational visits, specifically to the UK's Royal Society for the Protection of Birds (RSPB) sites. There may be similar records for other conservation organisations that could also be used to communicate the importance of biodiversity and habitats to educational attainment.

The above should not be considered a comprehensive review of all possibilities for measuring these 'direct' interaction type of ecosystem services. It should be noted that in many cases it will be quite challenging to identify which particular ecosystem service a person is deriving benefits from, for example is it in terms of aesthetics, cultural heritage, wildlife watching and physical recreational experiences? In most cases a combination is quite likely (Henandez-Morcillio et al. 2013), indicating that all these aspects should be considered when managing ecosystems for the supply of these services.

Physical measurement of ecosystem services grounded in remote interactions with biodiversity and habitat is clearly challenging. For the entertainment or representation ecosystem service in Table 1, options could include audience numbers for nature films, book sales or visits to nature based collections. However, there needs to be some care in how these types of estimates are generated as nature based entertainment may not be derived from ecosystems within the same national jurisdiction for which the accounts are being compiled.

For those ecosystem services with symbolic, sacred or religious meanings or with bequest or existence value (bottom of Table 1), the flow of ecosystem services is directly grounded in the conservation and enhancement of biodiversity and habitat stocks. The measurement of how marginal changes in these stocks affect well-being is, typically, captured by willingness-to-pay for conservation or increases in biodiversity stocks. For these types of services, marginal changes in the stocks can be considered as an indicator of the changes on the ecosystem service flow, although this is likely to be characterised by non-linearities. These types of flows do not lend themselves to direct physical measurement, although cardinal measures associated with the number of individuals expressing positive preferences for increases in the stocks could also provide some indication of ecosystem service flow based on characterisation of its demand.

Valuation of biodiversity and habitat-related cultural ecosystem services

The Nature based recreation paper summarises a number approaches to value trips to engage with biodiversity and visit habitats for recreational benefits. These are generally grounded in estimating expenditure incurred to participate in the trip or cost of time associated with participating in the trip. Mourato et al., (2010) provides a similar valuation based on making these trip 'investments' for the RSPB site visits by school children discussed above for educational benefits. Based on these costs of the investment expended in the pursuit of ecological knowledge on nature based trips to RSPB reserves by schools, total values were estimated to range from just under £850,000 to just over £1.3 million per year. These costs are grounded in market prices and, therefore, amenable for use in an accounting context.

Another way to, potentially, get to a value for ecological knowledge is to value it as a services that builds stocks of human capital. Human capital is formalised by, among other ways, educational attainment, as this is a significant factor in future income. There is a large economic literature on returns to education but very little of this seeks to isolate the portion of human capital that can be attributed to ecological knowledge attained in formal schooling. Mourato et al., (2010) also outline an approach to valuing ecological knowledge in this way, based on weighting the ecological component of Geography, Biology and Science GCSE and A-Level courses. A value is then derived by comparing future incomes for students that gain these qualifications with those leaving school at 16 with no qualifications. In fact they estimate quite substantial Values for Ecological Knowledge as part of Geography, Biology and Science GCSE and A-Level courses that exceed £2 billion / year (in 2010). Again, these are transaction values and suitable for use in the SEEA EEA context.

For the ecosystem service of nature based entertainment via films, documentaries, books etc., values could be estimated from film ticket prices, nature channel subscriptions and costs of other media. Such as books, posters, etc. The contribution of the biodiversity and habitat related ecosystem service could be recovered using resource rent approaches, essentially removing the production and distribution costs associated with these goods from their market price. There has also been some debate in the conservation literature on trying to recover resource rents from nature film producers using a voluntary tax mechanism (e.g., see Wunder & Sheil, 2013). Whilst contentious, this would provide a useful value to apply as a lower bound to valuing this service if it was ever taken up.

The type of valuation approaches described above are, largely, grounded in the instrumental values for cultural ecosystems services as described in Figure 1. However, it has long been argued that biodiversity should be perceived as a good by and of itself (Mace et al. 2012) and that people will value biodiversity based on the relations they have to it (as described by the IPBES relational values in Figure 1). This may particularly be the case for charismatic species and ecosystems that people ascribe non-use value to (especially in relation to their existence now – i.e. existence value - and for the future generation – i.e. bequest value). In this sense, biodiversity can be seen as a source of value for individuals linked to maintaining habitat for these species and the ecosystem themselves. These reflect non-use values derived from certain cultural ecosystem services, rather than intrinsic or non-anthropocentric values highlighted in Figure 1 (the contribution of these values to total economic value is discussed further in TEEB, Chapter 5, Pascual et al., 2010).

People are found to have non-use values for a range of species and habitats and these values have, predominantly, been elicited through stated preference valuation methods (see Annex 1 for examples). As Henandez-Morcillio et al., (2013) observe, these values and normative imperatives are reflected, to some degree, by society's decisions to establish protected areas. However, these stated preference approaches represent the full willingness to pay (welfare value) of individuals for an ecosystem service, rather than the value that would occur in a well-functioning market. As such they are not compatible with the SEEA and the national accounting approach grounded in transaction / exchange values.

Sumarga et al (2015) propose an alternative approach to estimating the value of habitat that avoids this accounting issue associated with stated preference approaches. They focus on one specific aspect of aspect of biodiversity, namely orangutan habitat on the island of Kalimantan, Indonesia. They select orangutans as an indicator for biodiversity given it is a flagship conservation species. They then apply a defensive expenditure method to value orangutan habitat (i.e., an existence value). This is based on amortising the costs of reintroducing orangutans into forests over the orangutan's expected lifetime. This value can then be expressed as the avoided cost of reintroduction and linked to the population (or stocks) of orangutans in given areas. However, as the authors clearly note, this is clearly a gross underestimation of the value of the habitat that will support many more species. Nonetheless, this approach appears to resonate with peoples preferences for biodiversity (e.g., the WWF adopt a Panda as a flagship species as the symbol for public engagement).

The aggregation of the final service values (use and non-use) produced by ecosystems has been labelled TEV (total economic value). But the aggregate instrumental TEV of a given ecosystem service flow (or combinations of ecosystems at the landscape level) is not equivalent to the Total System Value (TSV). The continued functioning of a healthy ecosystem is more than the sum of its individual components. The operative ecosystem yields or possesses primary, 'glue' or infrastructure value. Some combination of ecosystem structure and composition is necessary to ensure the 'healthy' functioning of the system (Gren et al 1994; Turner et al 2003). The role that the system plays in holding everything together has, in principle, an economic value, thus the TSV (total system value)> TEV. This is discussed in more detail in the 'Biodiversity as an asset' section of this document.

Another argument which supports the proposition that TSV>TEV is that there may be something extra to the value of an ecosystem over and above TEV, which can be described as 'shared values'. This value is expressed at a collective /community level where landscapes for example possesses symbolic, historical and cultural meaning (key services captured in Table 1). These cultural values relate to, for example, wild nature and ecosystems, and ancient landscapes/forests (Fish et al 2011). An ample example of such shared values is the European network of the protected areas Natura 2000 network. Through democratic process a shared value was articulated in the EU which is embodied in protecting the numerous ecologically and culturally valuable species and habitats across the EU, institutionalised through Birds and Habitat directive, respectively. An indication of these values might be viewed in the accounting context as the expenses that the EU puts into maintaining the network. This reflected the approach proposed by Eftec et al., (2015) for Corporate Natural Capital Accounting, where biodiversity maintenance costs are recorded. Similarly held values could be seen in other countries through legislation aimed at protection of ecologically and culturally important habitats and species worldwide. However, it is important to note that the costs related to environmental goods and services might not have a relationship with the value of the benefits these are associated with (Barbier 2007). This approach also represents a tautology in the way that the value of the ecosystem services provided by biodiversity and habitat is the amount society is prepared to spend on maintaining them. Nevertheless, it does capture a lower bound.

Biodiversity as an asset maintaining capacity for future ecosystem-service delivery

Within the ecosystem accounting community discussions are ongoing on how accounting for the capacity of ecosystems to deliver ecosystems services can be achieved. There are clear conceptual links here to ecosystem condition accounting, a broader concern than purely biodiversity condition. As a contribution to this debate, this paper also provides tangible guidance on viewing and measuring **biodiversity as an asset** that enhances the capacity of ecosystems to continue delivering services into the future. Valuation options for this asset are also explored. This will allow decision-makers to better consider the trade-offs that emerge between different land-use and management options, and their effects both on present-day ecosystem service flows and on biodiversity assets underpinning the capacity of ecosystems to continue delivering services into the future.

Description of the asset

As outlined earlier in this paper, biodiversity makes important contributions to the present-day delivery of a wide range of ecosystem services. These contributions relate not only to cultural services, including those discussed in the previous section, but also to various provisioning and regulating services, addressed in detail by other papers in this series. A growing body of literature suggests, however, that these direct present-day contributions represent just the 'tip of the iceberg' if we broaden our perspective on the relationship between biodiversity and ecosystem services to also consider the role that biodiversity might play in maintaining and enhancing *"the capacity of the ecosystem asset to continue to generate ecosystem services into the future"* (SEEA EEA 2012), particularly in the face of uncertain environmental change (Isbell et al 2017).

This broader perspective also shifts the scope of measurement and valuation from a relatively narrow focus on those particular elements of biodiversity underpinning present-day supply of ecosystem services, towards consideration of the likely relevance and importance of diversity as a whole, across all levels of biological organisation – i.e. genetic, species and ecosystem diversity. In doing so, it becomes clear that accounting for changes in biological diversity *per se* (as opposed to changes in particular service-providing species, or in gross structural or functional attributes of ecosystems) should be motivated as much by utilitarian concerns, relating to long-term maintenance of ecosystem-service flows, as it is by the more familiar intrinsic or existence value perspective most often invoked in this context.

Two major means by which biodiversity contributes to maintaining capacity for future ecosystem-service delivery are worth distinguishing (as illustrated in Figure 3):

1. From a provisioning-services perspective, elements of biodiversity (e.g. particular species) which may not provide services at present could be needed to provide these same services, or new services not yet envisaged, in the future. This is the concept of "option value" (Weitzman 1992; Faith 2018). For example: a currently rare fish species might greatly increase in abundance and replace a commercial species impacted by climate change, thereby maintaining this important food supply; or a tropical tree species, providing no obvious service at present, might prove to be the only source of a drug capable of combating a major new human disease.
2. From a regulating-services perspective, the diversity of organisms constituting an ecosystem may be vital to the long-term maintenance of fundamental ecosystem

processes (or 'ecological functions') underpinning services such as carbon sequestration or water purification, particularly in the face of significant environmental fluctuation and/or directional change (e.g. climate change). Various terminology is used in treatments of this general concept across different bodies of literature – e.g. "insurance value" (Baumgartner 2007; Augeraud-Véron et al. 2017), or as a component of "ecosystem resilience" (Oliver et al 2015). However the common theme running throughout these perspectives is that the presence of a diversity of organisms (e.g. multiple species) performing a given functional role within an ecosystem boosts the capacity of that system to maintain functionality, despite environmental changes affecting individual elements of this diversity in different ways.

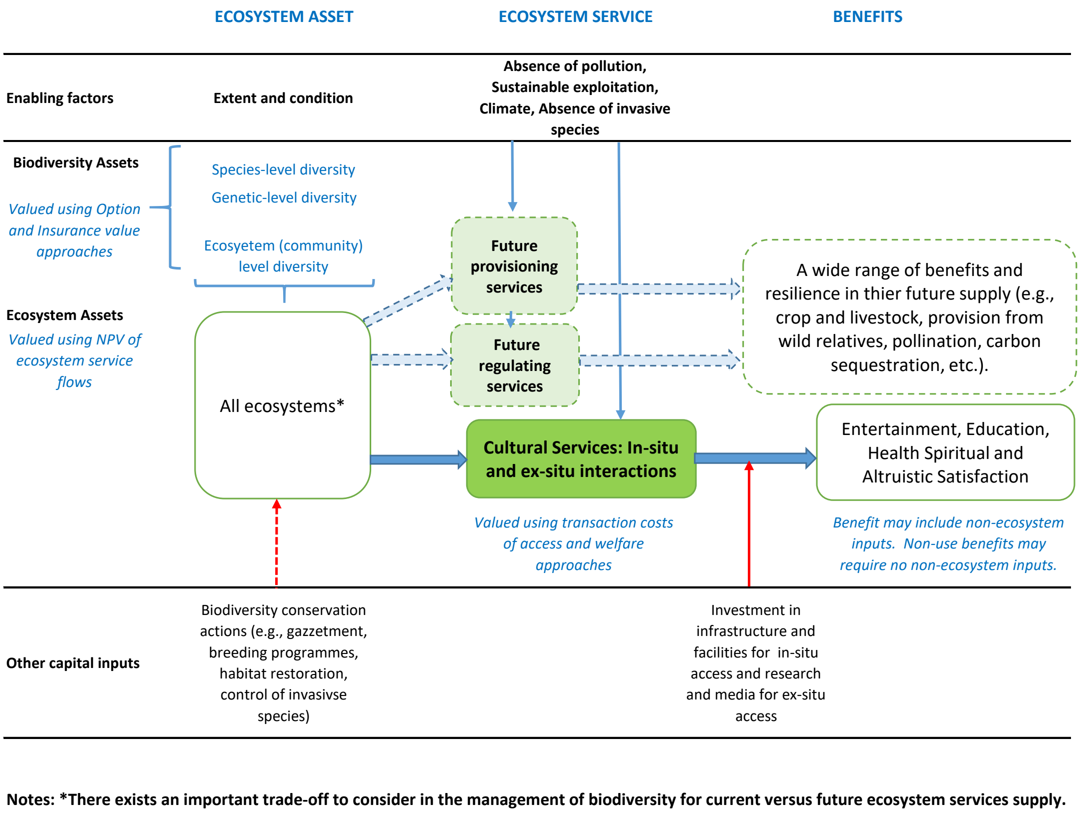

Figure 3: Logic chain for biodiversity as an asset that maintains capacity for future ecosystem-service delivery.

Physical measurement of biodiversity as an asset for future ecosystem-service delivery

Translating this conceptual understanding (of biodiversity's likely contribution to maintaining capacity for future ecosystem-service delivery) into an operational capability for measurement and valuation is clearly fraught with challenges. Techniques for measuring or modelling changes in the capacity of an ecosystem to deliver services into the future as an explicit function of changes in the current stock of biological diversity within that system, while holding considerable promise, are very much in their infancy (e.g. Hisano et al 2018; Sakschewski et al 2016). In the near to medium term it is likely that any assessment of this capacity will need to be based on the implicit assumption that the overall level of biological diversity present within an ecosystem is a reasonable indicator of this component of ecosystem capacity. Such an approach would allow emerging efforts in accounting for ecosystem capacity to take full advantage of the many advances made in developing robust indicators of biodiversity change by the biodiversity-science/biodiversity-conservation community over the past couple of decades.

Wherever possible, however, the types of indicators employed for this purpose need to go well beyond those focused purely on assessing local species richness (alpha diversity), as adopted in several previous accounting exercises. A growing body of research in this field is pointing to the importance of considering beta and gamma diversity in this context – i.e. differences in biological composition between locations, contributing to collective levels of diversity retained across whole regions (e.g. Burley et al 2016). Accounting for the *complementarity* of species assemblages is the core motivation here. In this sense, complementarity (beta diversity) regulates how the richness (alpha diversity) of local species assemblages combines to generate the gamma diversity of the whole, larger system (Colwell & Coddington, 1994). This concept is totally scalable, for example in relation to the species assemblages located in the root systems and canopies of individual trees, or the pattern of species assemblages at landscape level. Given different species, and species assemblages, will perform different ecological processes and have varying degrees of resilience to different pressures, accounting for complementarity is a key ecosystem accounting concern if ambitions for resilient multi-functional landscapes are to be realised.

The importance of accounting for beta and gamma diversity, or in other words the complementarity of species assemblages across whole landscapes and regions, has major implications for how ecosystem accounting would need to approach the assessment of biodiversity as an asset maintaining capacity for future ecosystem-service delivery. It is unlikely that this can be achieved effectively by simply factoring an assessment of local biodiversity occurring within ecosystem assets (i.e. alpha diversity) into the derivation of ecosystem condition accounts. Given that ecosystem assets are defined as discrete spatial occurrences (individual polygons or patches) of ecosystem types, the total biodiversity value (i.e. gamma diversity) of a larger reporting area cannot be derived simply by summing the local biodiversity, or ecosystem condition, scores of the ecosystem assets contained within that unit. This is because, unlike most other ecosystem attributes and values being assessed by accounts, spatial scaling of biodiversity is strongly non-additive. This means that any assessment of the collective state of biodiversity within a reporting area containing multiple ecosystem assets must consider not only the state of biodiversity within each of these assets, but also complementarities in species composition (i.e. beta diversity) between these assets. The same applies to consideration of the effects of habitat configuration, especially connectedness / fragmentation, on biodiversity persistence and ecosystem resilience. This will be a function not only of connectedness within each ecosystem asset but also, and often more importantly, of connectedness between these assets. Proper consideration of both complementarity in species assemblages (beta diversity), and the effects of spatial connectedness of habitat, can be achieved only through whole-landscape approaches to biodiversity assessment (e.g. Ferrier and Drielsma 2010).

Biodiversity research also highlights the importance of considering changes in overall diversity at levels of biological organisation other than the species level (i.e. genetic and ecosystem diversity) and addressing dimensions of diversity other than taxonomic diversity (e.g. functional and phylogenetic diversity; Oliver et al 2015). Rapid advances have been, or are being, made by the biodiversity-science community in developing indicators to more effectively accommodate these multiple levels and dimensions of biodiversity, and it is vital that the benefits of this work now flow across to the development and implementation of ecosystem accounts (UNEP-WCMC 2016).

While this general approach would enable quantification, at least in relative terms, of the biodiversity asset (or stock of biodiversity) underpinning an ecosystem's capacity to deliver services into the future, the challenge of valuation of this capacity is likely to remain unresolved, particularly if this valuation is expected to be expressed in monetary terms. This is clearly an issue which will need to be considered as part of any ongoing deliberations around the treatment of ecosystem capacity more generally in the SEEA ecosystem accounting framework.

Valuation of biodiversity as an asset for future ecosystem-service delivery

Diversity itself can be of value to the process and flow of final ecosystem services, in terms of ecological redundancy capacity. In this dimension of value a number of concepts have been proposed using metaphors and labels such as insurance value, resilience and stability value. These notion are linked to components of TEV, including option, bequest and existence value and relate mainly to instrumental values but with a 'cross-over' into intrinsic value when the full meaning of existence value is explored.

Option value in economic terms (not in financial planning terms) has a long history going back to the 1960s. As first conceived it was defined by three characteristics: uncertainty over the future uses for the asset concerned; irreversibility if loss occurs, either on technical or cost replacement feasibility: and non-storability of the asset. Biodiversity loss fits with the first two characteristics but the advent of seed/gene banks can mitigate to some extent the third factor. Later expositions of the option value concept highlighted the uncertainty and risk preferences linkage. In this view option value is seen as an uncertainty risk premium, known as option price. Finally, if irreversibility is seen as the key issue the value of risk avoidance becomes important and has been labelled quasi-option value or the irreversibility effect (e.g., Conrad 1997).

The stock of biodiversity found in particularly highly diverse biodiversity hotspots can also embodies an option value related to bioprospecting and biomimicry. The Nagoya Protocol on Access and Benefit-sharing (ABS) adopted by the Convention on Biological Diversity in 2010 might offer an indication of the option values related to biodiversity. In principle, companies are likely to pay a lower bound of the value that they are expecting to derive from a given biodiversity resource in the ABS contracts. A number of ways have been proposed to account for the value of bioprospecting. If we take profits as a tax base for royalties, Pravachol (cholesterol) generated US$1.5 billion in profits; Zocor (cholesterol) and Mevacrov (cholesterol) yielded US$3.6 billion and US$1.1 billion respectively. A 3% hypothetical profits sharing agreement would have been worth US$200 million. Some estimates have put the contribution of an area of land to drug discovery at between US$21 and US$ 9,177 per hectare, while the NPV of an extract in a screening programme was estimated at US$487 on average (Harvey and Gericke 2011). However, such values should be clearly contextualised as being narrow underestimates.

Another concept that has been prominent in the biodiversity conservation debate is the idea of insurance value or resilience value. Its general meaning seems to be that resilience is a desirable property of ecosystems in terms of combating stress and shock which could lead to ecosystem state change and irreversible loss of ecosystem service flows (Holling 1973; Scheffer et al 2001). A more diverse system, with more ecological redundancy capacity, may be more stable/adaptable in the face of exogenous stress and shock. It has been demonstrated that biodiversity at multiple trophic levels is needed for ecosystem multi-functionality in the case of grasslands. High species richness in multiple trophic groups has more positive effects on ecosystem services than richness in an individual trophic group (Soliveres et al 2016).

But it is possible to see the notion of biodiversity as an insurance value in two ways. The overall resilience value includes but is larger than the insurance value. The latter can relate to the concept that insurance is an action or institution that mitigates the influence of uncertainty on human welfare. So strictly in insurance terms, Baumgartner and Strunz (2014) define the economic insurance value of resilience as the reduction of an ecosystem user's income risk from using ecosystem services under uncertainty. The value depends on ecosystem properties, economic context and on the ecosystem user's attitude to risk. Their findings indicate that the insurance value is positive for high levels of resilience but not for low resilience states.

Maler et al (2009) see resilience as a stock with a distinct asset value which can be degraded or enhanced over time. Walker et al (2010) provide an empirical application of this stock of resilience approach. The application concerns agriculture and water salinity risks in Australia. Agricultural expansion with tree loss risks depleting the stock of non-salinated soils, measured as the depth of soils for which saline intrusion is not a problem. The benefits of this expansion/depletion risk process need to be traded off against the costs if stocks fall below some threshold level. This approach of course requires a large existing data/knowledge pool, especially when stocks with thresholds are affected by multiple interacting variables (Bateman et al 2011).

Key issues for research

This paper highlights two key measurement challenges that need to be overcome when accounting for and valuing biodiversity and habitat related ecosystem services:

1) As an asset from which we derive a flow of benefits characterised by the relational values we hold for biodiversity, either underpinned by directly interactions with the stocks of biodiversity and habitat or on the basis of bequest and existence cultural ecosystem services (non-use values) (we want to be able to measure these physically or by physical proxy and in monetary terms).
2) As an asset that underpins the future capacity of ecosystems to provide services (we want to be able to measure capacity as a physical metric and to capture these insurance and option values).

Physical measurement

A key challenge and priority for the SEEA-EEA revision process is to build consensus on the measurement approach for biodiversity as an asset at landscape scales (i.e., with respect to the concept of complementarity) in relation to point 2 above. The biodiversity assessment literature provides a number of ways to quantify this at Ecosystem Accounting Area scale (a number are highlighted in UNEP-WCMC 2016). Whilst it is acknowledged that some of the underlying modelling approaches are complex to implement there are several support tools that have evolved in recent years. For example, The Local Ecological Footprinting Tool (LEFT) (University of Oxford, no date) provides a flexible spatial platform for implementing a suite of analytical tools at 30m grid resolution, including Generalised Dissimilarity Modelling (i.e., calculating gamma diversity) using records extracted from the GBIF database, although limited to an analysis window of 4 degree latitude by 4 degrees longitude. Several other modelling approaches are highlighted on the biodiversity Indicators partnership website: https://www.bipindicators.net/. Developing platforms that can implement these approaches in a user friendly manner for ecosystem accounting (and other applications) is a priority. There are several other, scale dependent ecosystem condition parameters that could be tackled as part of a joint programme, for example calculation indices for fragmentation of habitats is important as it interrupts the flows of organisms, matter and generic materials through the landscape. A key challenge for ecological science challenge exists here and significant investment in primary biodiversity monitoring data may well be required if such efforts are to prove successful in improving decision making.

Valuation

Capturing the full range of values that biodiversity embodies in the accounting framework is likely to be difficult due to number of reasons. Natural science knowledge as well as economic theory that would help to value the role biodiversity plays in underpinning ecosystem services and in supporting ecological redundancy is lacking.

Some valuation knowledge and practice is available for estimating the (mostly non-use) values associated with important habitats and charismatic species, particularly using stated preference valuation techniques. These methods are thought to be the only economic valuation techniques capable of measuring non-use values. Since these are based on a welfare value concept, however, including such values in SEEA would not be consistent with other (transaction) values in the accounts. An alternative solution for inclusion of these values in the SEEA would be via Satellite wealth based accounts, which would also be highly useful for informing public policy (Badura et al 2017 EU INCA report). The nature based recreation paper also discusses the use of simulated exchange values to purge consumer surplus from welfare values.

With respect to the application of the stated preferences to charismatic habitats and species a few further issues arise. For example, It has been shown that people tend to value charismatic species over other species or the ecosystems that support these species (e.g. Morse-Jones et al. 2013; Senzaki et al. 2017) and that use of 'iconic' species in stated preference studies might lead to dramatically higher estimated WTP values than if the value for the habitat would had been estimated (Jacobsen et al. 2008). Further, it is unclear how much beneficiaries gain from the habitat service provision if it was a global public good such as Brazilian rainforest. It has also been shown that demand for biodiversity is related to countries' wealth and hence impacts on the value that individual countries would hold for given good (Jacobsen and Hanley 2009).

The use of expenditure values associated with biodiversity and habitat maintained, e.g., annual budgets for managing protected area estates also offer an opportunity for setting a lower bound for related ecosystem services. However, linking these values to specific benefits is challenging and needs to be further explored. The fact that such approaches will also, likely, result in gross underestimation of society's value for these ecosystem services also substantially limits the potential to provide a long term solution to this measurement challenges.

Extensions and applications

Overall there is clearly need for further research with respect to the links between thematic biodiversity accounts (for measuring and valuing biodiversity as an asset) and ecosystem service accounts and the valuation of ecosystem assets based on Net Present Values of these services. Whilst a focus on final ecosystems services is clearly needed to avoid double counting of benefits, accounting approaches or auxiliary accounting tools need to be developed that either communicate the dependencies and trade-offs between the use of ecosystem services and the depreciation of the biodiversity asset base, or integrate such depreciation into ecosystem service values.

Making the links between the SEEA-EEA and decision support should also be key research area for the environmental accounting community. Admiraal et al. (2013) proposes an interesting approach using portfolio theory (which seeks to maximise returns whilst minimising risk) that could be explored in an accounting context. This would be achieved by using information on biodiversity to guide investment in a portfolio of ecosystem assets to generate policy acceptable returns on investment (determined from monetary ecosystem services account), whilst managing risk and reducing volatility in services supply (by maintaining a suitable portfolio of biodiversity).

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Annex 1: Relevant valuation literature for existence and bequest values

Reference	Journal	Year	Value category	$Value$	What is Valued	Whose values (sample)	Method
Amirnejad et al (2006)	Ecological Economics	2004	Existence Value	$30.12 / household / year	North forests of Iran	population in provicial cities in Iran (n=950)	CVM
Strand et al (2017)	Ecological Economics	2012-2013	non-use value	$4 - $100 / household / year	Amazon rainforest	over 200 environmental valuation experts from 37 countries on four continents	Delphi method where experts predict their country's CVM results
Navrud and Strand (2018)	Environmental and Resource Economics	2012	non-use value	EUR 28 / household / year	Amazon rainforest	49 European environmental valuation experts	Delphi method where experts predict their country's CVM results
Randall and Mercer (1997)	Land Economics	1992	non-use	$21 & $31 per household, one time contribution	Tropical Rainforests globally	542 US households	CVM
Turpie (2003)	Ecological Economics	2001	existence value	$10.4 / respondent / year (conservation scenario) $36.7 per year per respondent (Climate change loss scenario)	Vegetation in South Africa likely to be lost due to climate change and other pressures	814 respondents from Western Cape region (568 from Cape Metropolitan Area)	CVM
Horton et al (2003)	Environment Conservation	2000	Non-use value	$45.60 per household per annum for 5% protection of Amazonia $59.29 for 20% protection	Brazilian Amazon	Norwich in UK & Lazio, Lombardy and Tuscany in Italy (total 442 individuals were approached and 407 interviews were completed)	CVM
Morse-Jones et al (2012)	Ecological Economics	2009	non-use values of wildlife conservation in distant locations	Mean marginal WTP (MWTP) per household per annum: Unique and Charismatic (Gorilla) £15.90 Unique and Non-Charismatic (Frog, Toad, Lizard, or Bird) £9.77 Non-unique and Charismatic (Lion) £12.78 Non-unique and Non-Charismatic (Frog) −£0.87a Premium for conserving in two sites £4.44	WTP for (i) conserving wildlife in the Eastern Arc Mountains of Tanzania, and (ii) a larger good, that includes conserving wildlife in the Eastern Arc and in the Cameroon Highlands.	999 complete questionnaires, from UK residents	CE

Sumarga et al (2014)	Ecosystem Services	2010	SEEA compatible: the avoided release costs	up to 17 euro/ha/year	Orangutan habitat	N/A	avoided costs
Garrod and Willis (1997)	Ecological Economics	1995	non-use value	£18.5-£56.4 per year per household	biodiversity improvements in remote forestry commission UK forests	650 UK households (over 146 sampling points throughout Britain)	CE
Jakobsson and Dragun (2001)	Environmental and Resource Economics	N/A	non-use value	$29 per household per year for the conservation of Leadbeater's possum and $118 per household per year for all endangered flora and fauna in Victoria	Conservation of Leadbeater's possum and all endangered flora and fauna in Victoria, Australia	around 1290 respondents from Victoria, Australia	CVM
Kontoleon and Swanson (2003)	Land Economics	1998	existence value	mean (median) US$14.8 (US$10) one off payment	qualitative scenario afforded in situ conservation of the 500 pandas within their natural habitat.	305 non-chinese OECD respondents	CVM
Wattage and Mardle (2008)	Wetlands Ecology Management	N/A	non-use value (and use value too in study)	$1.134-1.386 for non-use ($2.52 total)	Muthurajawela marsh and Negombo lagoon (MMNL) wetland system	local population (358 Questionnaires)	CVM
Rolfe and Windle (2012)	Environmental and Resource Economics	2009 - 2010	Both use and non-use values	The average WTP across Australian households is $21.68 per household per annum for 5 years. Study particularly looks at the effect of distance on the values.	the values to protect the health of the Great Barrier Reef (GBR) at the national level	1919 surveys across Australia's major cities	CE
Yao et al. (2014)	Ecological Economics	2009-2010	non-use value	N/A - study derives marginal WTP values for different attributes of biodiversity enhancements	biodiversity enhancement programme in New Zealand's planted forests	209 New Zealand households	CE