Experimental Biodiversity Accounting as a component of the System of Environmental-Economic Accounting Experimental Ecosystem Accounting (SEEA-EEA)

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Executive Summary

Biodiversity (the diversity of ecosystems, species and genes) plays an essential role in supporting human well-being through maintaining functioning ecosystems that in turn deliver ecosystem services such as food, the regulation of our climate and aesthetic enjoyment.

The System of Environmental-Economic Accounting 2012 Experimental Ecosystem Accounting (SEEA-EEA) provides a framework to measure and link ecosystem service flows supported by biodiversity and other ecosystem characteristics (e.g. soil type, altitude) with the economy and other human activities. It also allows comparison and integration of data on ecosystem services with other economic and social data. Biodiversity Accounts, one of a number of accounts in the SEEA-EEA framework, can help understand the relationship between biodiversity within ecosystems and economic development and planning activity by cutting across this data in a spatially explicit manner.

Accounting for aspects of biodiversity is complex and as such experimentation of biodiversity accounting by countries is less advanced than water or carbon accounting, for instance. This technical guidance document has been prepared in the context of the Advancing the SEEA-EEA project. It is aimed at practitioners who wish to collect and organize data to understand the status and trends of ecosystem and species diversity and incorporate this into the SEEA-EEA framework for national accounting. This document considers the concept of biodiversity in relation to ecosystem functioning and condition and its resilience in the context of national accounting. It reviews some (but not all) of the established approaches to assessing the extent and condition of ecosystem diversity and for measuring species diversity, and examines their suitability for accounting purposes. This document sets out the data mobilisation process for national scale data and considers the use of global datasets and models to inform Biodiversity Accounts and other accounts in the SEEA-EEA framework, such as Ecosystem Condition Accounts. To assist countries in implementing biodiversity accounting, this technical guidance document presents experimental Biodiversity Accounts drawing on case studies from around the world. These are presented using a three tiered approach based on data availability. Other experimental accounts for Ecosystem Condition Accounts and Ecosystem Capacity Accounts are also shown. This document also highlights issues for resolution and recommendations for testing, refining and validation in relation to biodiversity accounting.

Why account for biodiversity?

To deliver sustainable development, national accounting systems need information on the foundation for sustainable economic growth provided by ecosystems and their services. Biodiversity accounting through the SEEA-EEA provides the methodology to help understand the contribution of biodiversity (ecosystem and species diversity) to human well-being and the economy by explicitly considering its role as a determinant of ecosystem condition essential for the generation of ecosystem service flows.

Biodiversity data is incorporated into the SEEA-EEA framework via a supporting Biodiversity Account. The information within the Biodiversity Account provides the basis for generating an indicator of condition, which is recorded within the Ecosystem Condition Account. The spatial nature of the accounts allows statistics on biodiversity to be examined against economic and social statistics in a spatially consistent manner and infer causal relationships.

Biodiversity accounting also provides opportunities for the harmonisation of national level biodiversity data alongside other reporting mechanisms, such as the Sustainable Development Goals (SDGs) and the Convention on Biological Diversity (CBD) through the implementation of National Biodiversity Strategies and Action Plans (NBSAPs).

Assessing ecosystem and species diversity

Assessments of biodiversity generally consider ecosystems and species due to the cost and complexity of assessing genetic diversity. However, that is not to say that genetic diversity is not important and could not be integrated into an accounting framework in the future.

Ecosystem extent can be estimated using land cover, land use, habitat and other ecosystem data from satellite remote sensing, that is data from earth observation satellites. Within the SEEA-EEA framework, these data inputs provide spatial information for delineating ecosystems (as assets) on the basis of common characteristics. This provides information on ecosystem diversity, which can inform the Biodiversity Account. While countries may have their own ecosystem classification standards and methods for mapping them, there is currently no internationally accepted ecosystem classification that can be directly mapped from remote sensing.

This technical guidance document focuses mainly on supplementing information on ecosystem diversity with information on species diversity. Species diversity data is targeted due to its availability and because species measures can also be used to approximate the status of biodiversity, as biodiversity indicators. Development of a Biodiversity Account using species data should move beyond simple counts of the number of species (the species richness) and include the population size of each species (the species abundance) as this provides more information on the status of biodiversity. However, measures of species diversity are resource intensive and have methodological challenges. A complete inventory of a country's species is not possible and so species to be included in a Biodiversity Account will need to be prioritised. When selecting species the greater the representation of taxonomic groups (e.g. plants, birds, mammals etc.), the better the account will estimate biodiversity. In addition, some species (e.g. keystone species) are better indicators of biodiversity or ecological condition than others.

Suitability of assessment approaches for biodiversity accounting

Biodiversity data needs to meet the following criteria to be suitable for informing biodiversity accounting. It should:

- Exist at a spatial resolution suitable for accounting. This allows data to be mapped to individual ecosystem units so stocks of biodiversity can be assigned to ecosystem assets.
- Be temporally relevant. This informs net changes in the stock of biodiversity between opening and closing accounting periods.
- Be comparable to a common reference condition. This allows comparison ecosystem assets against a benchmark indicative of a balanced state and aids aggregation of different types of biodiversity data.
- Be possible to aggregate it collectively to provide an overall indicator of the condition of biodiversity relevant to ecosystem functioning (e.g., via the Simpson Index or aggregation using a common reference condition). The change in the value of this indicator between accounting periods provides an indication of the net biodiversity balance.
- Be comparable over space and time. This allows direct comparison of biodiversity stocks in different ecosystem units.

When measuring ecosystem diversity, remote sensing and associated mapping of land cover, use, habitat or other ecosystem characteristics can meet most of these criteria for informing biodiversity accounting with regards to assessing ecosystem diversity. The main limitation to this is establishing the correspondence between these characteristics and ecosystems.

The Red List of Ecosystems will in due course meet these criteria by generating measures of ecosystem condition based on risk of ecosystem collapse. The spatial resolution will be high enough for national accounting (anticipated to be at least equivalent to the 250m resolution at which Ecological Land Units are mapped). The first global assessment (scheduled for 2025) will provide a baseline which can be used as a reference condition subsequently. Assessments will likely be repeated on a 5-year basis. Further, the application of the quantitative Categories and Criteria will ensure consistency and comparability between countries and over time.

In regards to organising species diversity data, three approaches were examined for their suitability. The IUCN Red List of Threatened Species measures extinction risk. Application of the IUCN Red List Categories and Criteria ensures consistency in assessment over space, over time, and between assessors. Methods are available to allow disaggregation of the Red List Index to national levels. Downscaling of the global Red List to national levels can be complemented with national red lists, where these exist. It is suggested that both the global Red List and national red lists are used to enable different information to be picked up.

The Norwegian Nature Index (NNI) uses indicators from a variety of species groups and major ecosystem types that measure deviation from a reference state. The NNI produces a single 'value' that provides information on ecosystem condition. The methodology involves a series of aggregations first within spatial units and then across spatial units. The NNI incorporates expert judgement, monitoring-based estimates, and model-based estimates, so the method can be used in both data rich and data poor areas.

The Living Planet Index (LPI) aggregates species population trend data from different sources and across multiple spatial scales. The methodology involves a series of aggregations in order to avoid bias by well-known taxonomic groups and well-studied locations. With systematic monitoring of species abundance, the data would lend itself for incorporation in a Biodiversity Account. For those countries that lack this systematic data, the methodology can still yield a single 'value' to provide information on ecosystem condition.

Implementing biodiversity accounting

Getting started

A key starting point for biodiversity accounting is to identify biodiversity-related policy priorities to help determine what information, including on plants, animals and to a lesser extent fungi, should be compiled. Guidance on the selection process is provided below. This step will also establish the resolution of data (both spatial and temporal) necessary to address these priorities.

Establishing an inventory of all existing monitoring data will identify any 'data-gaps' for national biodiversity accounting. Countries should consider their reporting obligations to regional processes and biodiversity-related conventions/agreements, such as the CBD or Ramsar. Identifying data gaps could inform a protocol for further data gathering (e.g., via monitoring or modelling approaches).

The first step in the biodiversity accounting process is to delineate ecosystem assets spatially on the basis of similarities in ecological and ecosystem characteristics (i.e. Ecosystem Units). This can be considered the foundation of a Biodiversity Account in the broadest sense, as it provides information on ecosystem diversity. Capturing further information on ecosystem diversity on the basis of ecological variations in/within classes of Ecosystem Units specified can improve these accounts further. Additional supporting Biodiversity Accounts can then be developed for Ecosystem Units. In order to facilitate analysis and reporting, information on biodiversity can be aggregated across Ecosystem Units to larger scales (e.g., across administrative boundaries or ecological features, such as watersheds).

Due to the complexity of accounting for all components of biodiversity, the Biodiversity Accounts suggested in this document focus mainly on developing information on ecosystem diversity and the construction of 'species accounts'. Based on the policy priorities species that deliver important ecosystem services should be selected, for example, economically important species (e.g. game species) or species associated with ecosystem functioning (e.g. keystone species). More than one Biodiversity Account may be required in order to answer the full range of biodiversity-relevant policy questions. For instance, information on biodiversity relevant to ecosystem functioning may require a different accounting structure than information on species extinction risk. In creating the species account analysts should consult with ecologists to ensure meaningful data is collated and collected.

A number of experimental approaches to Biodiversity Accounting that build on the SEEA-EEA are reviewed or suggested in this document. These approaches vary in complexity and resource requirements and are presented as a three tiered approach ('Tier 1', 'Tier 2', and 'Tier 3'):

- 'Tier 1' accounts capture information on the ecosystem characteristics used to define different classes of Ecosystem Units (or important areas of biodiversity habitat), their extent and can be weighted using input indicators of species diversity.
- 'Tier 2' accounts capture information on species richness, extinction risk and potentially other characteristics (e.g. species health) for ecosystem and other accounting reporting units.
- 'Tier 3' accounts capture information on species abundance within ecosystem and other accounting reporting units.

Whilst primary monitoring data is the ideal for assigning biodiversity information to ecosystem units this is unlikely to be available at the spatial resolution required for ecosystem accounting. A number of approaches exist for upscaling or downscaling data on biodiversity; these include habitat modelling, land use modelling, species-area curves and expert judgement approaches. A portfolio of these approaches may be required to inform biodiversity accounting. Any application of these approaches should be supported by regular updates of primary monitoring data.

Limitations and issues to resolve

This review has found that a majority of potential global datasets in their present state do not provide the temporal or spatial resolution necessary to inform national biodiversity accounting. Furthermore, developing Biodiversity Accounts that are globally comparable is likely to be challenging, particularly when relative measures of biodiversity are employed. This is because a consistent reference condition is required.

Whilst a single biodiversity indicator can provide an overall indication of ecosystem condition (potentially for an Ecosystem Capacity Account), it is unlikely to be useful in informing the link to ecosystem service delivery (via an Ecosystem Service Account). This is because there may be many aspects of biodiversity that will be of importance to different ecosystem services. Consequently, a broad suite of biodiversity indicators is likely to be required. For those aspects of biodiversity considered an ecosystem service in their own right, (e.g. for their existence or aesthetic enjoyment) information in the Biodiversity Account can inform ecosystem service delivery directly.

Ultimately the value of the contribution of biodiversity to ecosystem service provision would be extremely useful to record in the ecosystem accounting framework (i.e. via an Ecosystem Service Account). There exist various market and non-market based valuation techniques to generate a lower bound for this value. However, this will only be possible for a subset of ecosystem services. Accordingly, any valuation of biodiversity should be considered an underestimate and is unlikely to capture all intermediate services (intra- and inter-ecosystem flows) that are dependent on biodiversity and are important for overall ecosystem functioning.

Recommendations for testing, refining and validating

More testing is required of suitable spatial scales for biodiversity accounting. This should be supported with further testing of the modelling and other approaches for generating spatially explicit information on biodiversity via various downscaling and upscaling approaches. Protocols for validation and calibration of these approaches should also be explored.

Issues around selecting the appropriate scale also have significant implications for aggregation of biodiversity information. Further research into and testing of methods to aggregate ecosystem and species data and condition indicators across ecosystem units is required. This should consider the implications of ecotones (in terms areas of high biodiversity on ecosystem borders) and the diversity between different ecosystem units across spatial areas.

The asset accounts for biodiversity recommended in the SEEA-EEA allow for causes of addition and reduction in the stocks of species diversity to be recorded. There are obvious benefits to establishing such a clear causal relationship. However, this will require additional data collection and may often be difficult to complete in a balanced manner. As such the possibilities for undertaking this would benefit from testing in a specific case study, possibly via linkages to land ownership or land use.

Biodiversity is considered as an indicator of condition in the Ecosystem Condition Account. Improvements and reductions in the indicator levels can also be recorded in the condition account. However, there exist multiple drivers of biodiversity loss and so a supplementary account for drivers of ecosystem condition could be a possibility for testing. This would also provide a suitable structure for capturing considerations such as habitat fragmentation, invasive species and ecotones.

The link between biodiversity and ecosystem service delivery is complex. There will often be time lags between changes in biodiversity stocks and resulting changes to the level of ecosystem services provision. Furthermore, capturing information on the importance of biodiversity to ecosystem functional redundancy and resilience is challenging due to non-linear and threshold effects. Given the importance of biodiversity to ecosystem functioning and sustaining ecosystem service provision, considering ecosystem functional redundancy and resilience is a key issue to be addressed in the ecosystem accounting framework. Further research is required in this regard, potentially via an account that captures information on the capacity of ecosystems to deliver services (i.e. an Ecosystem Capacity Account).

To conclude, the Biodiversity Accounts presented in this technical guidance document are experimental accounts and are not necessarily prescriptive of a recommended course of action. They require testing, refining and validation by countries in different contexts and with levels of data availability in order to determine their practicality to be integrated into national accounting and to inform decision-making.

Purpose, Scope and Structure

This technical guidance document on experimental biodiversity accounting is prepared in the context of the System of Environmental-Economic Accounting 2012 Advancing Experimental Ecosystem Accounting project (SEEA-EEA). It is aimed at practitioners who wish to collect and organize data to understand the status and trends of ecosystem and species diversity and incorporate these into the wider SEEA-EEA framework (2014) for national accounting.

Biodiversity Accounts can provide information on the trends in plants and animals within specific ecosystem areas over time. Accounting for biodiversity in this manner is necessary because biodiversity is important for ecosystem functioning and condition, as well as the provision of ecosystem services and their contribution to human well-being. The SEEA-EEA provides the framework to measure and link ecosystem service flows supported by biodiversity and other ecosystem characteristics (e.g. soil type, altitude) with the economy and other human activities. This can, in turn, allow comparison and integration of data on ecosystem services with other economic and social data. An ultimate aim of such an integrated analysis is to reveal and reinforce the importance of the relationship between people and their environment (SEEA-EEA, 2014). More broadly, Biodiversity Accounts can help understand the relationship between biodiversity within ecosystems and economic, development and planning activity by cutting across this data in a spatially explicit manner.

This technical guidance document focuses on how to measure the status and trends of biodiversity temporally and spatially. It concentrates mainly on supplementing information on ecosystem diversity with information on species diversity. It does not examine genetic diversity. This technical guidance document builds on existing literature on biodiversity accounting produced as part of the SEEA process (e.g. SEEA-EEA, 2014; SEEA Central Framework, 2014; McDonald 2011; Ivanov et al., 2013), and ongoing initiatives relating to the measurement of biodiversity at the national level (e.g. Norwegian Nature Index; Cardinale et al., 2012). It also showcases case studies from countries who have experimented with biodiversity accounting.

This technical guidance document begins with a review of the literature. Section 1 introduces biodiversity in the context of national accounting. Section 2 presents a review of some of the established approaches to assess biodiversity (ecosystems and species) that are relevant to the SEEA-EEA accounting framework. Section 3 examines the suitability of approaches for organising ecosystem and species information for accounting purposes. Section 4 considers the data mobilisation process for national scale data and considers the use of global datasets and models to inform Biodiversity Accounts and Ecosystem Condition Accounts. The second part of the technical guidance document explores how Biodiversity Accounting can be implemented. To assist countries in implementation, this technical guidance document is presented as a three tiered approach based on data availability. Section 5 presents example Biodiversity Accounts covering plants and animals for each of the three tiers. In addition, Section 6 presents example Ecosystem Condition Accounts and Ecosystem Capacity Accounts. Finally, general conclusions are summarised in Section 7, including issues for resolution and recommendations for testing, refining and validation of the approaches suggested. Annex 1 presents a literature review of the relationship between biodiversity and i) ecosystem functioning and the delivery of ecosystem services, ii) ecological productivity and, iii) resilience.

1. Biodiversity in the context of national accounting

Key points

- The definition of biodiversity used in the System of Environmental-Economic Accounting Experimental Ecosystem Accounting (SEEA-EEA) includes ecosystem, species and genetic diversity.
- Biodiversity represents one element of a country's stock of wealth and provides benefits in terms of ecosystem good and services, which contribute to human well-being and provide inputs into the economy.
- Biodiversity contributes to human well-being in numerous ways. For example, it plays an important role in ecosystem functioning by regulating ecosystem processes, it provides genetic stock for medicinal products and aesthetic enjoyment through viewing wildlife.
- Biodiversity Accounting, through the SEEA-EEA framework, allows trends in biodiversity (and the benefits it provides) to be compared with economic and social activity in a spatially explicit manner. These comparisons can provide valuable insight to answer policy relevant questions. For instance, whether increases in measured consumption are sustainable.
- In the SEEA-EEA framework biodiversity information and data is contained within both the specific Biodiversity Account and the Ecosystem Condition Account due to its role in the delivery of ecosystem services.
- A number of approaches exist for integrating biodiversity into accounts that build on the SEEA, including those led by the Secretariat of the Convention on Biological Diversity (CBD) and the European Environment Agency.

This section explores what is biodiversity, its various roles in ecosystem services and why countries should incorporate biodiversity into their national accounts. It concludes with an introduction to where biodiversity information and data feed into the System of Environmental-Economic Accounting Experimental Ecosystem Accounting (SEEA-EEA) framework.

1.1 What is biodiversity?

There exist many definitions and measures of biodiversity (Mace et al., 2012). The definition of biodiversity used here and elsewhere within the SEEA-EEA (2014) framework is that adopted by the Convention on Biological Diversity (CBD, 1992):

> "Biological diversity means 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"

The relationship between these three components of biodiversity are illustrated in **Figure 1**. A current system of classification for living organisms is the six kingdoms of Bacteria, Protozoa, Chromista, Fungi, Plantae, and Animalia (Cavalier-Smith, 2004). From a pragmatic accounting perspective it is the last three kingdoms covering plants, animals and to a lesser extent fungi (e.g. edible mushrooms) that are the focus of this technical guidance document.

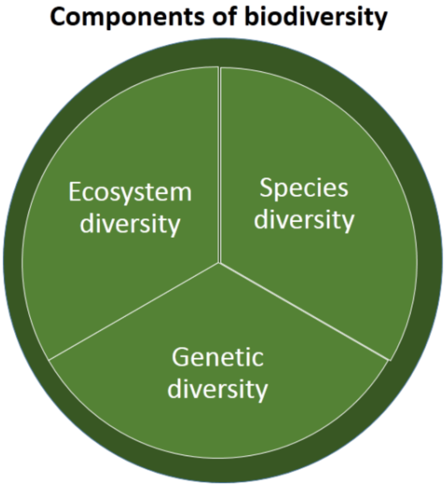

Figure 1. The three components of biodiversity.

1.2 Why is biodiversity important?

The Millennium Ecosystem Assessment (MA, 2005a) highlighted the importance of sustainable use and conservation of biodiversity (ecosystems, species and genes) by stating that "everyone in the world depends completely on Earth's ecosystems and the services that they provide, such as food, water, disease management, climate regulation, spiritual fulfilment, and aesthetic enjoyment" (**Box 1**). Biodiversity plays this essential role in supporting **human well-being** through maintaining the functioning of terrestrial, freshwater and marine ecosystems that underpin the delivery of a range of **ecosystem services** and **benefits** (MA, 2005b).

A considerable body of research exists that examines the relationship between biodiversity and ecosystem services (Naeem & Wright, 2003; Balvanera et al., 2006; Luck et al., 2009). In over 95% of experimental studies a positive relationship between **ecosystem functioning** and biodiversity has been recorded. However, it has been found that, in some cases, less than half the species within the system are needed to maintain most **ecosystem processes** (Schwartz et al., 2000). While this may be the case that fewer species are needed to maintain basic function, in other studies in both forest and grassland systems it has been shown that biodiversity increases ecosystem productivity (Thompson et al., 2012; Potvin & Gotelli, 2008). In addition, individual species are essential for some medicinal compounds both utilised locally and for more commercially produced compounds. Charismatic and iconic species have cultural value or are valued on the basis of aesthetics, characteristics and behaviour or because of the cultural status given to them (Mace et al., 2012; Kellert, 1997; Martín-López et al., 2007). While these individual species may or may not have key roles within ecosystem function, they do support ecosystem services and are vulnerable to ecosystem degradation. Further detail is provided in Annex 1 on the support that biodiversity provides to ecosystem function and resilience.

Box 1: Key definitions

Benefit: Goods and services that are ultimately used and enjoyed by people and which contribute to individual and societal well-being (SEEA-EEA, 2014).

Ecosystem: A dynamic complex of plant, animal and micro-organism communities and their non-living environment acting as a functional unit (CBD, 1992).

Ecosystem asset: Spatial areas comprising a combination of biotic and abiotic components and other elements which function together (SEEA-EEA, 2014).

Ecosystem component: The abiotic and biotic components such as plants, animals and soil that make up ecosystem assets.

Ecosystem condition: The overall quality of an ecosystem asset in terms of its characteristics (SEEA-EEA, 2014).

Ecosystem engineer: Those organisms that directly or indirectly modulate the availability of resources (other than themselves) to other species, by causing physical state changes in biotic and abiotic materials (Berkenbusch & Rowden, 2003).

Ecosystem diversity: The variety of ecosystems in a given place (WWF, n.d)

Ecosystem functioning: The changes in energy and matter over time and space occurring through biological activity, such as primary production, nutrient uptake, decomposition, and evapotranspiration (Mori et al., 2013).

Ecosystem process: This is synonymous with 'ecosystem function' (Reiss et al., 2009)

Ecosystem service: An activity or function of an ecosystem that provides benefit to humans (Mace et al., 2012)

Genetic diversity: The variation in the amount of genetic information within and among individuals of a population, a species, an assemblage, or a community (UN, 1992).

Good: The objects from ecosystems that people value through experience, use or consumption, whether that value is expressed in economic, social or personal terms. Note that the use of this term here goes well beyond a narrow definition of goods simply as physical items bought and sold in markets, and includes objects that have no market price (e.g. outdoor recreation) (Mace et al., 2012).

Human well-being: A context- and situation-dependent state, comprising basic material for a good life, freedom and choice, health and bodily well-being, good social relations, security, peace of mind, and spiritual experience (MA, 2005c).

Keystone species: A species whose impact on the community is disproportionately large relative to its abundance (MA, 2005c).

Resilience: The level of disturbance that an ecosystem can undergo without crossing a threshold to a situation with different structure or outputs. Resilience depends on ecological dynamics as well as the organizational and institutional capacity to understand, manage, and respond to these dynamics (MA, 2005c).

Species diversity: Biodiversity at the species level, often combining aspects of species richness, their relative abundance, and their dissimilarity (MA, 2005c).

Biodiversity can be a measure (both directly and as a proxy) of **ecosystem condition** (European Union, 2014). Biodiversity is a complex concept which encompasses multiple measures of species (Purvis & Hector, 2000). The status of species in terms of both species diversity as well as population sizes can reflect the condition of the underlying ecosystem (i.e. the quantity and quality of the habitat for these species). There will also be individual species that perform irreplaceable ecological functions, and if these are lost then the condition of the ecosystem (in terms of its ability to provide some services) will decline (Ripple et al., 2014; Coleman & Williams, 2002). This is often the case with **keystone species** (generally predators) or **ecosystem engineers**, since they control major cycles of nutrients, water, carbon and energy flows, which underpin ecosystem services. It is, however, recognised within the scientific community that further work is required to understand the role of biodiversity within ecosystem service delivery better. Nonetheless, the existing evidence of the role of biodiversity in supporting ecosystem function, which in turn provides a matrix in which important individual species exist, and the support biodiversity provides to ecosystem productivity all highlight its importance.

Biodiversity plays different roles in ecosystem services but distinguishing these can be complicated as our understanding of physical and biological processes underpinning ecosystem services is poor (MA, 2005b; Díaz et al., 2006; Mace et al., 2012). Mace et al. (2012) identify three roles. As a regulator of ecosystem processes, which derive the flow of materials and energy through and between ecosystems, and underpin ecosystem services. As a final ecosystem service, biodiversity contributes directly to human well-being by providing a stock of genetic material that provides medicines, crops, livestock and biofuels. As a **good**, biodiversity provides direct benefits to society on the basis of its continued existence and its experiential aspects (e.g., appreciation of wildlife and scenic places). **Figure 2** sets out the flow from ecosystems to societal benefits and the contribution of biodiversity.

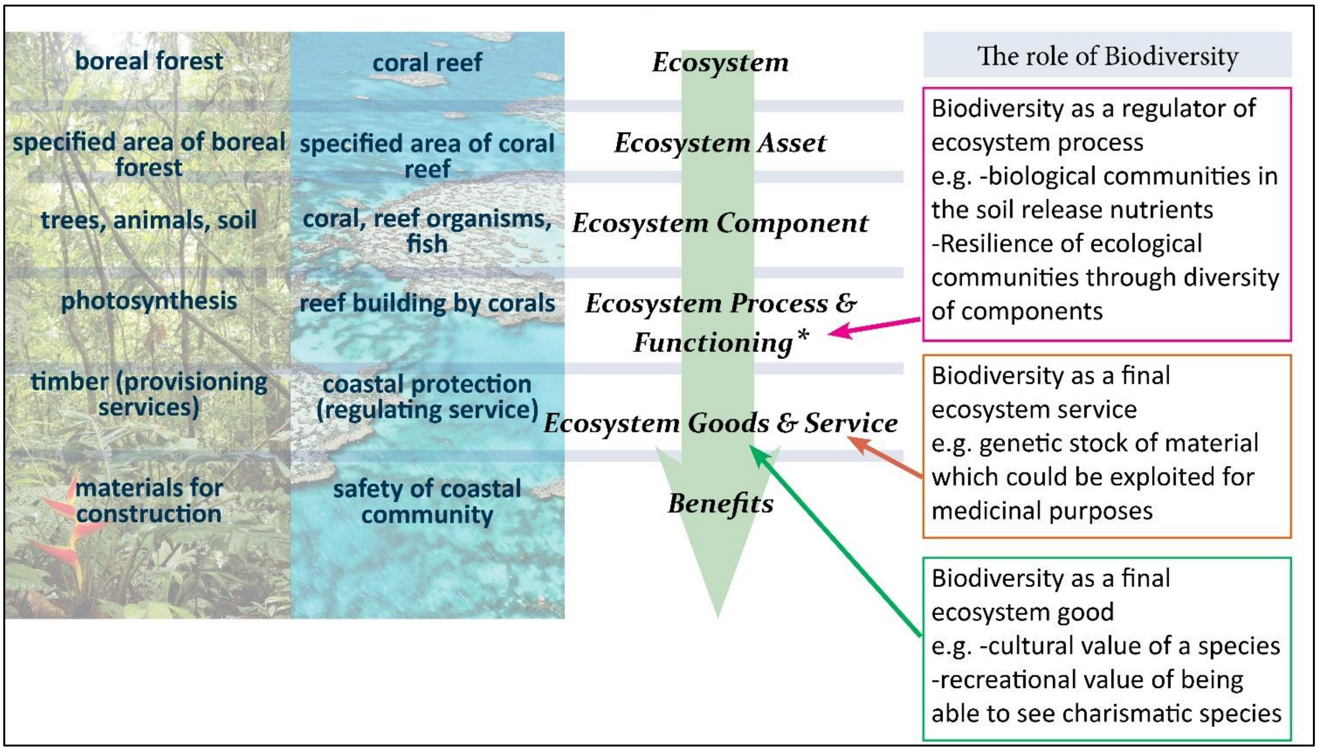

Figure 2. Schematic diagram of the roles of biodiversity in ecosystem service delivery: i) as regulator of ecosystem processes (e.g., nutrient cycling); ii) as a final ecosystem service (e.g., genetic stock for medicines); and, iii) as a good (e.g., viewing scenes of species interactions, such as birds at a wetland). *Represent intermediate services (intra- and inter-ecosystem flows).

There also exists considerable uncertainty surrounding future ecosystem services that biodiversity could provide. Consequently, biodiversity is important for maintaining the option to access these, as yet unknown, ecosystem services (e.g., undiscovered medicines) as well as its role in the **resilience** of ecosystems. Ecosystem resilience is important as it describes the capacity of ecosystems to continue to provide ecosystem services under changing future conditions (e.g., climate change or other shocks) (TEEB, 2010a).

1.3 Why account for biodiversity?

Governments around the world rely on economic data to inform their development policies. The System of National Accounts (SNA) is the national standard for measuring economic progress and compares annual changes in production (in markets where prices are available). However, the SNA does not provide all the information to determine whether current levels of economic activity can be sustained. In order to understand this more fully, Governments require the full range of information on the resource base (i.e. wealth) upon which much economic activity is (directly or indirectly) reliant. Studies such as the MA (2005a) and The Economics of Ecosystems and Biodiversity (TEEB, 2010a) concluded that ecosystems provide important services that contribute to human well-being and provide inputs into the economy. Therefore, sustainable management of ecosystems and biodiversity has important implications for social welfare and economic productivity in the long run (i.e., to achieve sustainable development).

In order to deliver sustainable development, national accounting systems need to fully account for the foundation for sustainable growth provided by ecosystems and their services. The SEEA-EEA (2014) provides an experimental framework for producing national statistics on the environment and its relationship with the economy. As indicated in Section 1.2 biodiversity is a key factor in both the functioning and resilience of ecosystems (Balvanera et al., 2006; Tilman et al., 2006) and their ability to provide ecosystem services. However, human activities resulting in overexploitation, pollution, land use shifts and climate change, for example, can decrease the resilience of natural ecosystems (Bertness et al., 2015). Further, the functioning of ecosystems and the future provision of ecosystem services may be under threat by the current rate of biodiversity loss (Luck et al., 2003). Accordingly, an account of biodiversity is crucial within the SEEA-EEA framework, in order to provide information on the delivery of final ecosystem goods and services as well as intermediate services (inter- and intra-ecosystem flows) that are provided by or are underpinned by biodiversity.

Biodiversity accounting can improve decision making by organising biodiversity information in a spatially explicit format consistent with other statistical frameworks (e.g. the SNA and SEEA Central Framework). This allows statistics on biodiversity to be examined against economic and social statistics in a spatially consistent manner. Trends in stocks of plants and animals can then be assessed against those associated with economic development and social activity and infer causal relationships. This can, in turn, provide the basis for more sustainable management of biodiversity and maintained delivery of important ecosystem services in the future.

Whilst beyond the remit of the SEEA-EEA, other ethical approaches would dispute the instrumental value of biodiversity, and instead adopt a non-utilitarian philosophical framework, which may be derived from closely-held moral, religious or cultural beliefs (Oksanen, 1997). Under this perspective biodiversity has an intrinsic value that is worth protecting, regardless of its instrumental value to humans (Justus et al., 2009). A Biodiversity Account would contain information that would no doubt be of interest to those who perceive biodiversity as intrinsically valuable in its own right, this reflects the moral argument for the conservation of biodiversity (Turner et al., 2003).

Biodiversity accounting data should be policy relevant and in order to facilitate communication to help inform policy responses the biodiversity data could be presented spatially (in map form). For example, at sub-national levels assessing trade-offs between ecosystem services is an important consideration in decision making (Raudsepp-Hearne et al., 2010; Rodriguez et al., 2006). It is especially critical to biodiversity where optimization of many more utilitarian services may come at the expense of biodiversity (Nelson et al., 2009) and would be an important function of accounting (Bond et al, 2013). In addition to the spatial dimension, the account also needs to be referenced temporally, and presented in tabular form, in order to identify trends in attributes of biodiversity, such as species richness, for particular areas over time. This can, in turn, provide insights to inform more sustainable management of natural resources. Bond et al. (2013) provide examples of how a spatially-explicit accounting system, which integrates biodiversity information and data, could inform policy and decision making (see **Box 2**).

Box 2: Examples of how biodiversity information can inform policy and decision making (extract from Bond et al., 2013)

- Land use and profitability can inform land use planning and conservation planning.
- Land use information also provides the link between biodiversity and industry allowing policy makers to identify high (and low) impact industries (Prugh et al., 2010).
- Profitability of different land uses is a key variable as the opportunity cost in making land use decisions at the local level.
- Land use and profitability accounts in conjunction with Biodiversity Accounts facilitates the analysis of where in the landscape high profits and food can being produced for a given level of biodiversity (Polasky et al., 2008).
- Trade-offs analysis, especially between ecosystem services, is an important consideration in decision-making (Raudsepp-Hearne et al., 2010; Rodriguez et al., 2006). It is especially critical to biodiversity where optimization of many more utilitarian services

may come at the expense of biodiversity (Nelson et al., 2009) and would be an important function of accounting.

- Proposed environmental expenditure accounts (UNCEEA, 2012) in conjunction with biodiversity stock accounts could provide a systematic return on investment information to inform cost-benefit analysis (UNCEEA, 2013).
- Spatially linking Biodiversity Accounts to mapped threats can assist in identifying where to invest in threat management for the greatest return for biodiversity at least cost (Carwardine et al., 2012; Evans et al., 2011).

Beyond the accounting framework, the CBD's Strategic Plan for Biodiversity (2011-2020) sets a vision where biodiversity is valued, conserved, restored and wisely used and ecosystem services maintained, sustaining a healthy planet and delivering benefits essential for all people. Accounting for biodiversity and its values is now built into global commitments associated with the Strategic Plan (especially Aichi Target 2) and is highly likely to feature as one of the proposed post-2015 Sustainable Development Goals (SDGs) and its proposed target 15.9. Many countries throughout the world are in the process of reviewing their National Biodiversity Strategies and Action Plans (NBSAPs) to meet these commitments. Biodiversity accounting therefore provides a framework for tracking progress towards these commitments and informing the sustainable management of biodiversity.

1.4 Links with other accounts in the SEEA-Experimental Ecosystem Accounting framework

Ecosystems are considered as **assets** within the SEEA-EEA (2014) and are measured from two perspectives, that of extent and condition and that of ecosystem services. Biodiversity data is incorporated into the SEEA-EEA framework via the supporting Biodiversity Account. The information within the Biodiversity Account provides the basis for generating an indicator of condition within the Ecosystem Condition Account. This section explains the rationale for these two accounts and their interlinkages with other SEEA-EEA accounts. **Figure 4** illustrates these linkages and presents a 'bottom-up' approach. Under this approach ecosystems, as assets, are characterised on the basis of their extent (via the Land Cover Account), their condition (via the Ecosystem Condition Account and supporting account) and their capacity to provide ecosystem services (in the Ecosystem Capacity Account).

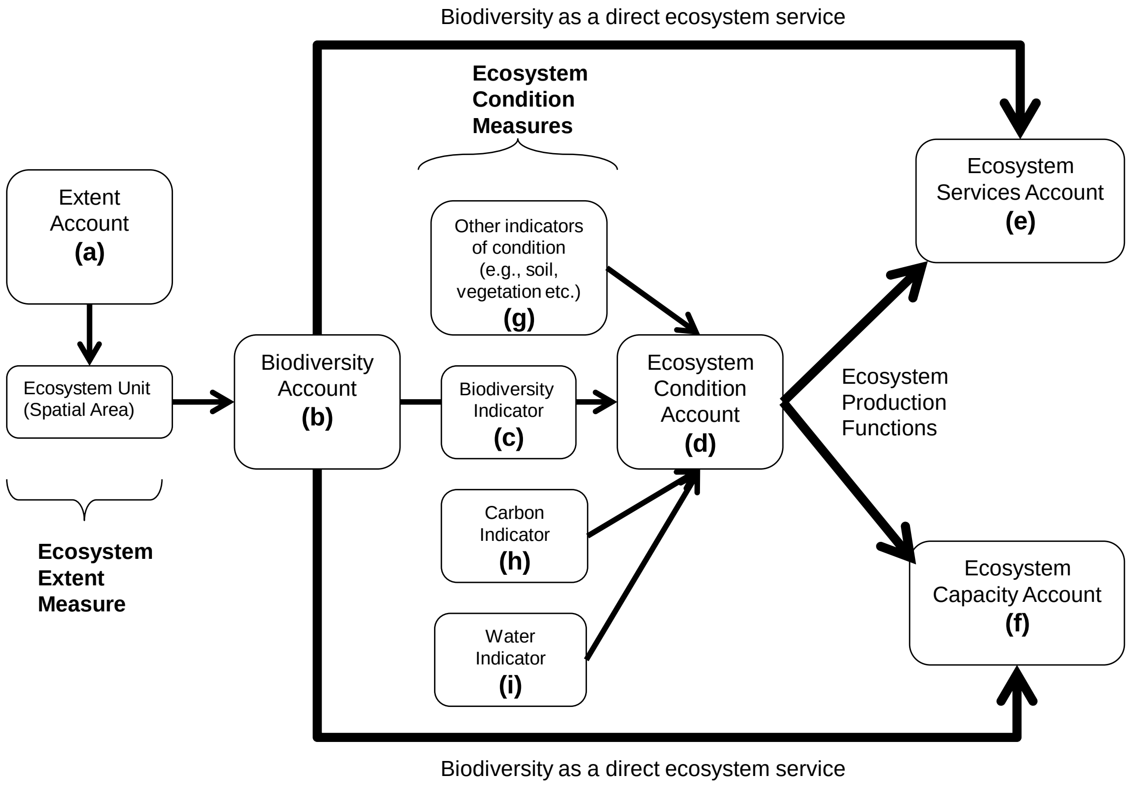

Figure 3. Location of biodiversity accounts within the SEEA-EEA and linkages with other accounts within the framework. Biodiversity information and data is contained within both the specific Biodiversity Account and the Ecosystem Condition Account due to role the biodiversity in ecosystem service provision.

For accounting purposes, ecosystems (as assets) are spatially defined as ecosystem units. The information on the spatial area of an Ecosystem Unit (EU) is recovered from the Extent Account (Figure 3(a)). The Extent Account provides the structure for defining the spatial boundaries for ecosystem units on the basis of similar ecosystem characteristics. Once defined these spatial units provide the structural framework for the Biodiversity Account(s), as well as other accounts within the overall ecosystem accounting framework. Spatial units are discussed in more detail in SEEA-EEA Tech. Guide 8 (2015) and Section 5 of this document.

Once a spatially explicit system of ecosystem units has been determined, the supporting Biodiversity Account (Figure 3(b)) can be populated. This provides a temporally and spatially explicit structure for capturing biodiversity measures for ecosystem units. The focus of this technical guidance document is on supplementing information on ecosystem diversity with information on species diversity, although the importance of genetic diversity in metapopulation dynamics is acknowledged (e.g., as Stanford et. al, 1996 discuss with respect to fish). As depicted in Figure 3, the Biodiversity Account, *inter alia*, underpins the Ecosystem Condition Account (Figure 3(c)) as the status of biodiversity within an ecosystem is intrinsic to the condition of that ecosystem (Rapport, 2007).

The Ecosystem Condition Account captures data on the compositional characteristics of an ecosystem unit that are fundamental to its integrity, functioning and ability to produce ecosystem services. Therefore, in order to inform the condition account, it will be necessary to determine an indicator of biodiversity (Figure 3(d)) as a characteristic of ecosystem condition from information in the Biodiversity Account. Whilst all measures of trends in biodiversity will provide a reflection of ecosystem condition, the analyst should select those measures of biodiversity that are most relevant to the functioning of that ecosystem and the services it provides (see Section 2) for inclusion in the Ecosystem Condition Account.

Figure 3 shows that biodiversity will be one of the characteristics that will be a measure of ecosystem condition. As set out in the CBD definition, ecosystems are a functional unit of abiotic and well as biotic components. For example, carbon and water availability, in addition to other factors (e.g., soil), will also be fundamental to many ecosystem functions. Therefore, indicators of these characteristics are also captured in the Ecosystem Condition Account ((g), (h), (i) in the Ecosystem Condition Account).

The Ecosystem Service Account (Figure 3(e)) captures the flow of services from the ecosystem unit into the economy. Ideally these accounts should quantify output of ecosystems in physical (and possibly monetary) terms. For biodiversity that is valued directly, either for its conservation or commercial value, relevant information (measures) recorded in the Biodiversity Account may enter the Ecosystem Capacity Account directly (see thick black arrows from (b) to (e) in Figure 3). Biodiversity also underpins many of the intermediate services (inter- and intra-ecosystem flows) that transfer energy and nutrients both within and between ecosystem assets. These intermediate services are essential to the output of many final ecosystem services from an ecosystem unit and also from other ecosystem units which benefit from the exchange of energy and materials. For these ecosystem services, ecosystem production functions provide a link between the characteristics recorded in the Ecosystem Condition Account and Ecosystem Services Account.

Finally, the Ecosystem Capacity Account (Figure 3(f)) is conceptually similar to the Ecosystem Service Account but captures the capacity of ecosystem units to deliver a sustainable yield of an ecosystem service (in physical and potentially monetary units). Again, the condition of biodiversity within an ecosystem unit will be an important determinant of the sustainable yield of final ecosystem services but also in the delivery of intermediate services. It should be noted that the concept of a capacity account is still requires development via research and discussion in the broader context of ecosystem accounting. The intention is that the capacity account will provide information to understand the relationship between ecosystems as assets and the sustainability of services they provide. Given their highly developmental nature, it is unlikely that they will be compiled in the initial phases of establishing ecosystem accounts.

The different ways in which people value biodiversity are by no means mutually exclusive. Biodiversity is an important characteristic of ecosystem condition and, therefore, important to the range of services ecosystems provide (via its supporting role). However, it is also an ecosystem service that is valued directly (e.g., on the basis of being able to view iconic species or for its existence). This means that it is not a case of making a choice of where biodiversity sits within the accounting framework (i.e. feeding into Ecosystem Condition Accounts or as a direct provider of ecosystem services). It is important in both these aspects. It should be noted that this does not lead to double-counting or overestimating the benefits from biodiversity in the Ecosystem Service Account or Ecosystem Capacity Account. This is because ecosystem service flows related to characteristics recorded in the Ecosystem Condition Account (e.g., food production, pollination and climate regulation) are distinct and different from the ecosystem services that are provided directly by biodiversity (e.g., existence and experiential services such as iconic species viewing) recorded in the Biodiversity Account.

There are a number of global and regional initiatives underway that are exploring ecosystem accounting, generally building on the SEEA-EEA, which seek to integrate biodiversity into the ecosystem accounting framework (see **Box 3**). However, biodiversity accounting is currently lagging behind the development of other cross-cutting accounts such as water and carbon, highlighting the need for more work in this area.

Box 3: Approaches for integrating biodiversity into ecosystem accounting

The CBD's Ecosystem Natural Capital Accounts: A Quick Start Package (ENCA-QSP; Jean-Louis Weber, 2014) provides a framework for integrating biodiversity into national accounting systems. The ENCA-QSP does not record stocks of biodiversity (unlike the Biodiversity Accounts suggested in SEEA-EEA, 2014). Rather the ENCA-QSP captures biodiversity indicators in an accounting table of ecosystem health. This allows for annual changes in the ecological integrity of ecosystems to be captured. The focus of the ENCA-QSP with respect to biodiversity is very much orientated towards providing information on ecosystem health and understanding the relationship between the use of ecosystems by society and the impact on biodiversity. The European Environment Agency (EEA, 2011) have also produced an experimental framework for ecosystem accounting. The framework includes 'Ecosystem Capital Biodiversity Accounts', which record changes in indicators of species and biotope health between accounting periods. Again, no specific consideration is given to compiling 'stock' accounts on of biodiversity. This approach has been built upon in the experimental accounts for Mauritius (Indian Ocean Commission, 2014). Associated with the EEA work, Ivanov et al. (2013) developed spatially explicit accounts for species and habitats of conservation importance using data reported by EU countries under Article 17 of the EUs Habitat Directive. These accounts presented information on the number of species and habitats per unit area, their conservation status and changes in number or conservation status through time. Elsewhere, ecosystem accounts are being tested at the sub-national level by the Australian Bureau of Statistics (ABS), where Biodiversity Accounts of threatened bird species have been produced for Victoria (Bond et al., 2013). Whilst not directly related to the SEEA-EEA, The Netherlands have constructed national accounts of biodiversity which have been developed based on a system of species abundance weighted habitat areas under the Natural Capital Index project (ten Brink & Tekelenburg, 2002). This approach has also been developed for Scotland as a Natural Capital Asset Index (Albon et al., 2014).

2. Review of approaches to assess biodiversity that are relevant to the SEEA-EEA accounting context

Key points

- Assessments of biodiversity generally consider ecosystems and species.
- There is currently no internationally accepted ecosystem classification.
- Ecosystem extent can be mapped by establishing a correspondence between an ecosystem classification and spatial data on land cover/land use.
- Using land-cover, obtained from remote sensing, to map ecosystems supports international comparability.
- Biodiversity Accounts will likely focus on species diversity due to the availability of species data and because species measures can also be used to approximate the status of biodiversity, as biodiversity indicators.
- Species richness is a frequently used measure of species diversity but species abundance is a more sensitive measure. These concepts capture the ‘quantity’ component of biodiversity.
- The concepts of alpha, beta and gamma diversity capture the ‘variation’ component of biodiversity. National accounts should attempt to capture gamma diversity and various techniques are being developed for factoring beta diversity into the measurement of biodiversity change across large spatial extents.
- Measures of species diversity are highly sensitive to sampling methodology and effort. There are issues surrounding the spatial, temporal and taxonomic aspects of data collection and the Biodiversity Accounts require specialist knowledge to deal with these various technical issues.
- Collection of biodiversity data is resource intensive and species inventories will therefore be restricted to a subset of species. The species that are chosen will depend on purpose of the Biodiversity Account based on the policy priorities.
- Biodiversity Accounts will better approximate species diversity the more representative the sample of species is of taxonomic groups (e.g. plants, mammals, birds).
- Species that are good indicators of ecological condition or biodiversity should be prioritised for inclusion in the Biodiversity Account.
- Biodiversity indicators depend on the quality and availability of biodiversity measures.

This section presents a general overview of approaches to assessing ecosystem and species diversity that are relevant to the context of the SEEA-EEA. It sets the basis for Section 3 which considers the suitability of these approaches for ecosystem accounting.

2.1 Measuring components of biodiversity

Figure 1 illustrated the three components of biodiversity: **ecosystems**, **species** and **genetic diversity** (Box 1). As biodiversity is multi-dimensional any assessment of it should aspire to approach it as such. Neither measures of species, ecosystems nor genetic diversity alone will provide an accurate assessment of the whole of biodiversity (see **Box 4**). Section 2 begins by considering how aspects of ecosystem diversity are measured and then focuses on species diversity measures. Whilst it is acknowledged that genetic diversity can have important ecological effects (Hughes et al., 2008), as well as in relation to resilience, genetic resources and evolutionary potential, approaches to assess genetic diversity are not addressed in this document.

Box 4: Key definitions

Alpha diversity: The biodiversity of an individual location or the within-community diversity.

Beta diversity: The complementarity of two measures of alpha diversity.

Commission errors occur when a species is mistakenly thought to be present in an area (Rondinini et al., 2006).

Ecosystem extent: The size of an ecosystem asset, commonly in terms of spatial area (SEEA-EEA, 2014).

Gamma diversity: The collective biodiversity across a landscape (a combination of alpha and beta diversity).

Geographic ranges: The broad geographic boundaries of the area where a species is known to occur, usually delineated using extreme localities where a species is known to occur and their ecological preferences (Rondinini et al., 2006).

An **index** is a specific type of indicator that comprises a number of measures combined in a particular way to increase their sensitivity, reliability or ease of communication, e.g., the Red List Index for birds shows changes in extinction risk over time obtained through a specific formula. Disaggregation and traceability are important (Brown et al., 2014).

An **indicator** uses measures to communicate something of interest. They are purpose and audience specific (Brown et al., 2014).

Measure (or measurement): The actual measurement of a state, quantity or process derived from observations or monitoring, e.g., species counts, biomass or area of habitat (Brown et al., 2014).

Metric: A set of measurements or data collected and used to underpin each indicator (Brown et al., 2014).

Omission errors occur when a species is mistakenly thought to be absent in an area (Rondinini et al., 2006).

Point locality data: The locations where a species has been recorded, originating from actual observations or specimen collection (Rondinini et al., 2006).

Predicted distribution data: Areas where a species is likely to be present as modelled from the suitability of environmental conditions (Rondinini et al., 2006).

Proxy: A measurement that can be used to represent the value of a different measure in a calculation.

Species abundance: The total number of individuals of a taxon or taxa in an area, population or community (or where counts are not feasible, other measures such as biomass and percentage cover) (MA, 2005d).

Species richness: The number of a species with a given sample, community, or area (usually from a particular taxa e.g., plant species richness) (MA, 2005d).

Umbrella species: Species that have either large habitat needs or other requirements whose conservation results in many other species being conserved at the ecosystem or landscape level (MA, 2005d).

2.2 Measuring ecosystem diversity, extent and condition

Ecosystem diversity (Box 1) at the national scale can be determined in different ways depending on the level of ecosystem classification needed and the spatial resolution required to capture variation in a country's ecosystems. Therefore, it is essential to crosswalk such ecosystem classification and the associated spatial units to a consistent global classification, to ensure comparability between countries. The ecoregions datasets created by the World Wildlife Fund for Nature (WWF) can help to inform this; these identify relatively large regions containing distinct assemblages of natural communities and species, with boundaries approximating the original extent of natural communities prior to major anthropogenic change (i.e., they incorporate biogeography). They identify 867 terrestrial ecoregions, nested into eight realms and 14 biomes (Olson et al., 2001), overlapped by 426 freshwater ecoregions (Abell et al., 2008), and complemented by 232 coastal and shelf ecoregions nested into 62 provinces and 12 realms (Spalding et al., 2007). More recently a global ecosystem stratification approach based on landform, geology, and climate (but not incorporating continental patterns of floristic composition) has been used to create a map of 3,923 Ecological Land Units (ELUs) at 250m spatial resolution produced by ESRI and the United States Geological Survey (USGS) (Sayre et al., 2014). ELUs are broadly comparable to the Formation level of the International Vegetation Classification hierarchy. Both ecoregions and ELUs will doubtless inform the development of globally consistent ecosystem classification and mapping in due course but currently there is no internationally agreed ecosystem classification system. At the European level, the European Nature Information System (EUNIS) habitat classification (Davies & Moss, 1999) aims to provide a unified classification to help link national classifications and support compliance with the European Habitat Directive obligations (EEA, 2014).

Ecosystem extent can be mapped by establishing a correspondence between an, often not-spatially explicit, ecosystem classification and spatial data on land cover/ land use (**Box 5**). Land cover is a function of vegetation and other biophysical properties such as climate, soil and hydrology but also of land use, which means land cover cannot be directly equated to an ecosystem classification, On the other hand using land cover also supports international comparability and may be the only consistent data available over large areas. Different ecosystem mapping initiatives use different criteria for land cover or land use and map different levels of ecosystem classification. For example, the WWF ecoregions based on biogeography and with its nested realms and biomes. At the European level for example, various ecosystem mapping initiatives combine the Corine land cover data with different habitat classifications, such as EUNIS (ETCSIA, 2013) or others (Ivanov et al., 2013).

Box 5: Land cover and land use

Land cover is the observed (bio)-physical cover on the Earth's surface (Herold et al., 2006). **Land use** is the arrangements, activities and inputs by people to produce, change or maintain a certain land cover type (Di Gregorio & Jansen, 1998). Therefore, while land cover is a readily observable characteristic of the land surface, land use is a description of the use of the land cover for human activities such as food production, recreation or industry. Land cover is directly observable using remote sensing methods but land use is normally inferred with the addition of contextual information on human population, infrastructure and agriculture.

The level of classification and the spatial resolution that can be attained in mapping ecosystems extent is dependent on the available land-cover data. Land cover/ land use data can be obtained from satellite remote sensing (**Box 6**). This term is used to describe observations from earth observation satellites that can map and monitor the Earth's surface at regular, routine intervals in an automated fashion. Remote sensing has long been used as a tool for global land cover mapping but advances in technology have allowed higher spatial resolutions and more advanced and frequent measurements. Remote sensing is increasingly used to map habitats and predict species distributions (e.g. Haines-Young, 1996; St Louis et al., 2006 & 2009). Secades et al. (2014) presents an overview of the possibilities that remote sensed data provide to biodiversity monitoring in the context of the Aichi Biodiversity Targets.

Box 6: Mapping habitat types at the national scale for Tanzania

A suite of land cover products at various spatial resolution for Tanzania is set out in the figure below. GlobeLand 30 (2000 and 2010) has a spatial resolution of 30m while GLC2000 is 1km and the Tanzania National Forestry Resources Monitoring and Assessment (NAFROMA) data are vector polygons of land cover interpreted by national experts. While GLC2000 is a widely used global land cover product (Bartholomé & Belward 2005) it may not be appropriate for habitat definition at the national scale especially in countries with ecosystem diversity at finer spatial scales than the grid cell size (1km). The GlobeLand30 product (Chen et al., 2015) can be used to infer habitat at a much finer spatial resolution (30m) while also allowing change to be estimated from 2000 to 2010. However, it is only limited to 10 land cover classes compared to the 22 classes of GLC2000 and 25 classes of NAFORMA.

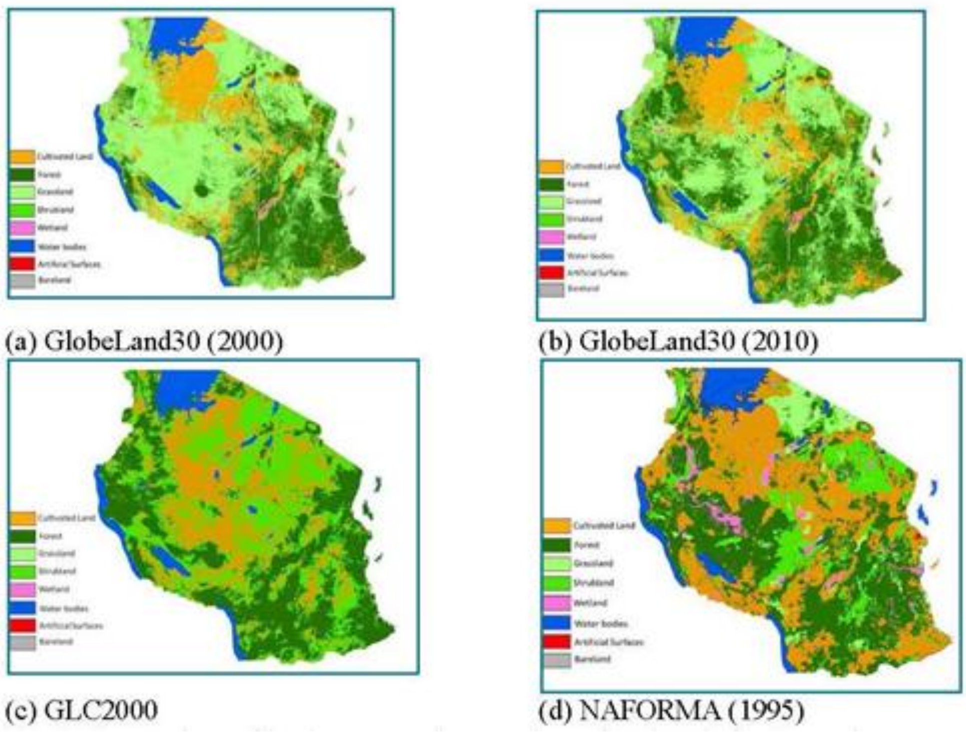

Figure. Comparison of land cover products at the national scale for Tanzania (note: land cover classes have been aggregated to the level of habitat for comparison purposes) (UNEP-WCMC, unpublished)

Within the SEEA-EEA framework, these land data inputs also provide spatial information for delineating ecosystems (as assets) on the basis of common land cover characteristics. Capturing this information on land cover within an accounting framework provides the basis for terrestrial ecosystem accounting. Tracking land cover change in and around habitats such as riparian zones, natural grasslands, tropical rainforests and mangrove forests, which provide key ecosystem services, can indicate trends in the ability of these habitats to continue providing these services. Monitoring the loss of such habitat through the analysis of land cover change is an essential first step to evaluating the potential loss of the services they provide.

Land cover has been mapped globally using remote sensing at a variety of spatial resolutions using two major global classification systems - those of the International Geosphere Biosphere Programme (IGBP) and the Food and Agricultural Organisation's (FAO) Land Cover Classification System (LCCS) (Loveland & Belward, 1997; Bartholomé & Belward, 2005). Monitoring temporal changes in ecosystem extent with the help of land cover data can be challenging because global land cover change data that are consistent over space and time are lacking and there is uncertainty in mapping spatial ecosystem changes especially in highly diverse environments (Gross et al., 2013). A global land cover change product at 30m spatial resolution was released in September 2014 by the National Geomatics Centre of China (Chen et al., 2015). GlobeLand 30 provides a more detailed picture of global land cover patterns and land cover changes induced by human activities from 2000 to 2010. Yet, while countries may have their own ecosystem classification standards and methods for mapping them, because land cover classes cannot be directly equated to ecosystems, there is currently no internationally accepted ecosystem classification that can be directly mapped from remote sensing.

Measurement of **ecosystem condition** (Box 1) requires not only information on changing ecosystem extent, but also on changing biotic and abiotic ecological process within the ecosystem. A promising approach to delivering such metrics is the application of the categories and criteria for risk to ecosystems (Keith et al., 2013), adopted by IUCN in 2014, which is intended to yield a global Red List of Ecosystems by 2025. This assesses risk of ecosystem collapse, based on standardised categories, criteria and thresholds for rates of decline in ecosystem distribution (A), ecosystems with restricted distributions with continuing declines or threats (B), rates of environmental (abiotic) degradation (C), rates of disruption to biotic processes (D), and quantitative estimates of the risk of ecosystem collapse (E). Ecosystem red listing is underway or has been completed in 14 countries and several regions, including Norway and Venezuela plus case studies in China and New Zealand (under Version 1; Rodríguez et al., 2011); and the Continental Americas, Madagascar, Morocco, Senegal, plus more than 20 case studies from Australia, East Asia, New Zealand, South Africa, the Aral Sea, the Caribbean, and Europe (under Version 2; Keith et al., 2013). This criteria testing has highlighted a number of challenges (Boitani et al., 2014; Keith et al., 2015), including: a) defining ecosystem types, their salient processes, and differentiating them from other ecosystems; b) defining when an ecosystem has collapsed; and c) assessing how spatial and temporal scales affect ecosystem threat assessments. These are ongoing areas of research.

Mapping the diversity, extent and condition of marine ecosystems poses additional challenges to mapping terrestrial ecosystems because of the variable physical, chemical and biological characteristics of the sea surface. There are a range of direct and indirect remotely sensed observations of marine ecosystems which can be used to characterise their physical, biogeochemical and biological conditions. Direct observations include sea surface height, salinity, ocean colour, sea surface temperature, sea ice thickness and extent, wind speed and direction. Indirectly habitats such as coral reefs have been successfully mapped in terms of structure and species composition (Mumby et al., 2004). The extent and condition of coastal habitats such as saltmarshes (Belluco et al., 2006) and mangroves (Giri et al., 2011) are continually mapped using a variety of direct and indirect remote sensing techniques. The foraging and migration patterns of marine mammals, sea birds and turtles can be tracked by satellite positioning systems and tagging devices in particular for the conservation of endangered species (Block et al., 2011).

2.3 Measuring species diversity

This guidance document mainly considers species diversity. There are four reasons for focusing on species (McDonald, 2011):

i) the conservation and sustainable use of species is addressed through a number of biodiversity related Multilateral Environment Agreements and national policies on biodiversity;

ii) species are relatively conspicuous;

iii) there is considerable research on species, decades of science on the measurement of species, and many countries have long-term monitoring programs for species; and

iv) species are often used as a **proxy** for biodiversity in general and are vitally important for ecosystem function.

The main measures for assessing species diversity are species richness and species abundance. In addition, biologists are interested in how these measures alternate across space, which is reflected in the concepts of **alpha**, **beta** and **gamma diversity** (Boxes 4 & 7). The distribution of rare and common species can also be informative to know and in a conservation context, measures of phylogenetic uniqueness, endemism and threat will also be of relevance. The methods used to gather information for assessing species biodiversity vary greatly.

Species richness is the most often used measure of species diversity (Hill & Burgess, 2015). However, species richness has notable sampling issues surrounding its measurement. From a spatial perspective, if sampling locations are not chosen at random then the resulting species richness data is very likely to be biased. Species richness also does not allow for differentiation between the species found in high and low quality habitats. From a temporal perspective, species assemblages undergo natural fluctuations across various time scales and it is important to take into consideration these natural fluctuations (Magurran, 2011). From a taxonomic perspective, not all species have an equal likelihood of being detected, resulting in a bias towards more conspicuous or charismatic species.

Species richness is based on species distribution data, which can contain both **commission errors** and **omission errors**. Three types of species distribution data exist: **point localities**, **geographic ranges** and **predicted distribution data**. The strengths and weaknesses of these methods are reviewed by Rondinini et al. (2006) and all include trade-offs. In summary, point locality data can minimize commission errors but are generally sparse and discontinuous, while geographic ranges generate large commission errors and may be too coarse to identify trends due to the assumption of homogenous species distributions. Predicted distribution data make explicit assumptions about species distributions and both types of errors and depends on the resolution of the raw data; as the data improves, so too will the outputs of the model. While it will not be possible to eliminate commission and omission errors, it is important to include an estimation of their extent.

Measures of species richness are highly sensitive to sampling methodology and sampling effort: with increased effort, more species are likely to be detected (Colwell & Coddington, 1994). Plotting the number of species against effort gives the species accumulation curve. A commonly used proxy for effort is area, which gives the species-area curve. The graphs will show an asymptote as the number of species approaches the ‘true’ species richness of an area. This asymptote can be estimated through various methods to give the species richness (Colwell & Coddington, 1994).

Measuring species richness alone can mask declines in populations of individual species, accounts equally for rare and common species and does not recognise when a decrease in certain species is offset by an increase in others. In contrast, the abundance of a species in danger will decrease long before the species is no longer detected in an assemblage, so **species abundance** is more sensitive to anthropogenic pressures than species richness. At a national and sub-national level, measures of abundance often refer to the number of individuals of a species within a specified area, the density. However, species abundance suffers from the potential issues discussed above such as sensitivity to sampling effort as well as issues surrounding the counting of individuals within a species.

Most importantly, it should be noted that measuring abundance is resource intensive (McDonald, 2011). As a result, it is not plausible to directly measure all species within a country on an annual basis. Therefore measures of species biodiversity are generally limited to a subset of the total number of species. What species are assessed depends on the priorities and concerns of the stakeholder conducting the assessment. For example, conservationists are particularly interested in the species abundance of rare and threatened species. However, it may be informative to assess the distribution of both rare and common species. While rare species may be the focus of conservation, common species are more responsible for the functionality of communities and are more likely to be significant contributors to ecosystem services (Mace et al., 2005). Therefore, from an accounting perspective the selection of species could be a combination of the following:

- representation of different species groups, trophic groups and the different roles of species in ecosystem functioning, this could include endemic species;
- important species for ecosystem functioning (e.g., **keystone species**);
- species that are highly specialised to specific ecosystems;
- threatened species that face risk of extinction in the wild that provide existence ecosystem services (MA, 2005b);
- economically important species that may provide ecosystem services directly (e.g. game species related to tourism);
- culturally important species that may provide ecosystem services directly (e.g. sacred plants/animals);
- socially important species that may provide ecosystem services directly (e.g. medicinal plants); and

- species that directly deliver particular ecosystem services (e.g. pollinators).

It is important to note that in order to represent species biodiversity, the subset should sample from across all taxonomic groups (i.e., a mammal and an amphibian is generally a better indication of biodiversity than two mammals). In addition, some species are considered better proxies (commonly referred to as surrogates) of biodiversity and ecosystem condition than others (Caro, 2010). For example, **umbrella species** are defined as such because their requirements include those of many other species as a result of sharing the same habitat) and their status can serve as a proxy for the status of multiple species (Roberge & Angelstam, 2004). **Keystone species** that are important to ecosystem function can be particularly useful as proxies for ecosystem condition. A final useful aspect to consider is species **extinction risk** due to its importance in conservation, which will be discussed in greater detail in Section 3.3.

Box 7: Measuring biodiversity: quantity and variation

When measuring species biodiversity, it is important to distinguish between quantity and variation. So far species biodiversity has been considered in terms of quantity (richness and abundance), rather than variability as emphasised by the CBD's definition (CBD, 1992). This variability is captured in the concept of alpha, beta and gamma diversity. At a national scale, holistic measurements of biodiversity should also attempt to capture gamma diversity. Simply aggregating estimates of local, cell-by-cell change in alpha diversity will not address the beta diversity component of gamma diversity (Faiths et al., 2008; Ferrier, 2011). Various techniques have and are being developed for factoring beta diversity into the measurement of biodiversity change across large spatial extents – working either with discrete classifications of ecosystem, or community, types (e.g., Drielsma et al., 2014; Faiths et al., 2008; Leathwick et al., 2010; Turak et al., 2011) or with continuous representations of spatial turnover in species composition (e.g., Allnutt et al., 2008; Ferrier et al., 2004).

Alpha and beta diversity is commonly measured through the use of the Shannon Index or the Simpson Index when abundance data is available. The Shannon Index describes the proportional contribution of each species within the community, whereas the Simpson Index describes the likelihood of encountering different species within a community (Sherwin et al., 2006).

2.3.1 Biodiversity indicators and indices

While the state of biodiversity can be given in many detailed figures, it is often instead communicated through the use of biodiversity **indicators** (see Box 4). The purpose of an indicator is to convey more information than the underlying data that is used in its construction (BIP, 2011). For example, a change in the population of a species that is highly specialised to a specific ecosystem could be a simple indicator of the ecological condition of the ecosystem. Indicators can be used for monitoring both ecosystem and species biodiversity however, in this document we will focus on indicators of species biodiversity.

Biodiversity indicators can be simple **measures** (e.g., change in a species population over time) or more complex **indices**. For example, using species abundance data, the proportion of specialist species can be used as an indicator of functional homogenization and biodiversity loss (The Community Specialization Index, see Clavel et al., 2011). A successful indicator should be: scientifically valid, based on available data, responsive to change in the issue of interest, easily understandable, and relevant to user's needs (BIP, 2011). The more representative the measures of species biodiversity captured in the indices, the better the indices will approximate species biodiversity.

The use of indicators and indices will be limited by the quality and availability of biodiversity measures used to populate them (**Figure 4**). The most appropriate biodiversity indicators/indices will include approaches to minimise bias from the over-representation of particularly well sampled species. For example, the Living Planet Index (LPI) partially compensates for over-representation of species from certain biogeographic realms by giving equal weight to data-poor and data-rich realms (for more information on the LPI see Section 3.4.2).

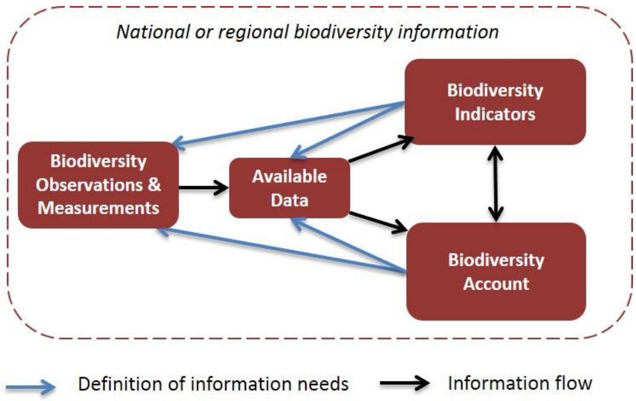

Figure 4. Biodiversity indicators depend on biodiversity measures and the availability of that data (adapted from a diagram produced in the EU BON Project).

At a global level, progress towards the 2020 Aichi Biodiversity Targets is monitored using a suite of global indicators. This suite of indicators is coordinated by the CBD-mandated Biodiversity Indicators Partnership (BIP). The BIP consists of more than 40 organisations who collaborate to develop new biodiversity indicators, strengthen existing ones, and to communicate these to decision-makers and other relevant parties. The BIP Secretariat, hosted at UNEP-WCMC, assists with national indicator development (BIP, 2010). Most recently they been working with countries in four regions to develop biodiversity indicators as part of revising their NBSAPs.

The BIP also provides support in other capacities. For example, to the Ad Hoc Technical Expert Group (AHTEG) on Indicators for the Strategic Plan for Biodiversity 2011-2020. The AHTEG is convening in June/July 2015 and is tasked with identifying a small set of measurable potential indicators that could be used to monitor progress at the global level towards the Aichi Biodiversity Targets, in particular, for targets that are currently not well addressed (CBD, 2015). The outcomes of the AHTEG will likely be of interest to the ecosystem accounting community not least because one of the Aichi Biodiversity Targets without indicators is Target 2 on national accounting. In regards to the wider relevance to ecosystem accounting, some of the indicators developed by the BIP have methodologies, or data that lend themselves to the possibility of developing opening and closing accounts of biodiversity stocks at national levels, and as such have the potential to feature in accounting systems. For example, countries can use the existing suite of indicators to develop their own fit-for-purpose indicators, such as the Uganda Living Planet Index (BIP, 2013). Some of these indicators will be discussed in Section 3.

In the accounting process, it is the data underpinning the indicator that is used. **Table 1** provides examples of biodiversity indices that have been developed using national biodiversity data. The suitability of some of these indices in an ecosystem accounting context are explored in the next section (Section 3).

Table 1. Biodiversity measures and associated biodiversity indices

Measure	**Index**	**Applied in**	**Reference**
Species abundance	Biodiversity Intactness Index	South Africa	Scholes & Biggs, 2005
	Living Planet Index	Uganda, Canada	Loh et al., 2005; Collen et al., 2009
	Marine Trophic Index		Pauly & Watson, 2005

	Wild Bird Index	UK	Gregory et al., 2005
	Species Specialization Index		Clavel et al. 2011
Extinction risk	The IUCN Red List	Multiple (see Section 3)	Butchart et al., 2004, 2005, 2007
Ecosystem extent and condition variables (species abundance, species richness or average ecosystem structure)	Natural Capital Index	Netherlands	ten Brink & Tekelenburg, 2000
Any ecological measure	The Norwegian Nature Index	Norway, Costa Rica	Certain et al., 2011

3. Critical evaluation of biodiversity assessment approaches for national accounting

Key points

- For ecosystem accounting biodiversity data needs to be: at a suitable spatial resolution; temporally relevant and; comparable over time and space, and to a reference condition. Plus collectively can be aggregated to provide an indication of condition that is relevant to ecosystem condition.
- Remote sensing for land cover mapping meets most of the suitability requirements for informing biodiversity accounting with regards to using land cover to assess ecosystem diversity.
- The Red List of Ecosystems will in due course meet the suitability criteria by generating measures of ecosystem condition based on risk of ecosystem collapse.
- The IUCN Red List of Threatened Species measures extinction risk. Application of the IUCN Red List Categories and Criteria ensures consistency in assessment over space, over time, and between assessors. Methods are available to allow disaggregation of the Red List Index to national levels.
- Downscaling of the global Red List to national levels can be complemented with national red lists, where these exist. It is suggested that both the global Red List and national red lists are used to enable different information to be picked up.
- The Norwegian Nature Index (NNI) provides a useful biodiversity indicator of ecosystem condition. The indexes underlying the NNI contain information that could be used to inform a Biodiversity Account.
- The NNI maps species diversity to different habitat types and can inform accounting for ecosystem and species diversity at a range of temporal and spatial scales. However, this is entirely dependent on the underlying data availability.
- The NNI approach is scalable, lends itself to aggregation, is flexible and can be used in different country contexts.
- The Living Planet Index (LPI) can yield a useful biodiversity indicator of ecosystem condition from heterogeneous data sources.
- The collection of the LPI on a biannual basis makes it useful for accounting.
- The utility of the LPI methodology for use in biodiversity accounting depends on the consistency of the underlying data. In theory this could be disaggregated to sub-national scales and be mapped to different habitats (e.g., as in Uganda).

This section discusses established methodologies for organising information and data on ecosystems and species and their potential for informing biodiversity accounting. The evaluation considers both the conceptual aspects of the methodologies and the potential for the associated reporting mechanisms and the underlying data to inform Biodiversity Accounts directly. Due to the difficulty of assessing all three components of biodiversity (shown in Figure 1) a pragmatic approach has been adopted in this technical guidance document. This section focuses mainly on the suitability of species data for a 'species account' to supplement information on ecosystem diversity.

3.1 Suitability criteria for informing biodiversity accounting

Biodiversity data needs to meet the following criteria to be suitable for informing biodiversity accounting:

- Data exists at a spatial resolution suitable for accounting. This allows data to be mapped to individual ecosystem units so stocks of biodiversity can be assigned to ecosystem assets.
- Data is temporally relevant. This informs net changes in the stock of biodiversity between opening and closing accounting periods.

- Data can be compared to a common reference condition. This allows comparison ecosystem assets against a benchmark indicative of a balanced state and aids aggregation of different types of biodiversity data.
- Collectively data can be aggregated to provide an overall indicator of the condition of biodiversity relevant to ecosystem functioning (e.g., via the Simpson Index or aggregation using a common reference condition). The change in the value of this indicator between accounting periods provides an indication of the net biodiversity balance.
- Data can be comparable over space and time. This allows direct comparison of biodiversity stocks in different ecosystem units.

These criteria pose significant challenges when considering biodiversity data. It requires dialogue between analysts and data providers to ascertain the flexibility of these criteria based on the data availability in their country.

3.2 Suitability of approaches for measuring ecosystem diversity, extent and condition for biodiversity accounting

Assessing the types of ecosystems and their extent in the area of interest is necessary in order to define ecosystem units. This is also an important step in the biodiversity accounting process as it provides information on ecosystem diversity. The SEEA Land Cover Accounts (SEEA-EEA Tech. Guide 2, 2015) provide a broad system for achieving this. Additional approaches are discussed in Section 2 and SEEA-EEA Tech. Guide 1 (2015).

Land cover data can help assess ecosystem diversity, extent and condition (the limitations of equating land cover classes to ecosystems being recognised and discussed in Section 2.2 and 5.2). Land-cover data can be obtained from remote sensing at resolutions that are limited by the spatial, spectral and temporal resolutions of the imaging sensor used. The spatial resolution of satellite remote sensing imagery available varies from 1km (AVHRR), 250 m (MODIS), 30 m (Landsat), 20 m (SPOT), down to a few meters (e.g. IKONOS, QuickBird). Not all data has global coverage. Some examples of global land cover mapping initiatives using satellite remote sensing are: IGBP land cover map based on AVHRR data (Loveland & Belward, 1997), GLC2000 using SPOT4 (Bartholomé & Belward, 2005), Globcover using MODIS, and Globeland30 based on Landsat (Chen et al., 2015). The use of remote sensing for detecting and mapping change is limited mainly by temporal resolution (revisit frequency), the availability of historical data and cost, especially for the highest resolution data. Landsat provides the longest and most consistent historical archive, with data going back to 1972, and covers the globe. Its data is also freely available and often used for mapping regional, national or local level land cover and land cover change (e.g. Fuller et al., 1994; Yuan et al., 2005).

Remote sensing images can be classified according to land cover different classification systems within the limits of the spectral resolution of the sensor. The higher the differentiation in, for example, vegetation types required, the more specific the band widths that are required to be able to detect differences. Most optical sensors are affected by atmospheric conditions, which vary over time, but correction algorithms help create time series of directly comparable data. Persistent cloud cover can be an important limitation in some areas, such as in the humid tropics. New radar sensors can help overcome such limitations, but the imagery is generally more expensive and more difficult to interpret.

Remote sensing for land cover mapping can meet most of the suitability requirements for informing biodiversity accounting with regards to using land cover to assess ecosystem diversity. The main limitation being establishing the correspondence between land cover classes and ecosystems.

Further, The Red List of Ecosystems will generate measures meeting the requirements outlined in Section 3.1 in due course. It will have a high enough spatial resolution for national accounting, anticipated to be at least equivalent to the 250m resolution at which Ecological Land Units are mapped. Assessments are planned to be re-evaluated every five years, the finest frequency which can support ongoing re-assessment of risk of ecosystem collapse. Through incorporation of continental patterns of floristic composition, the Red List of Ecosystems will also have high thematic resolution, approximating the Macrogroup level of the International Vegetation Classification's hierarchy, and be able to be crosswalked to national classifications as appropriate. The first global assessment (scheduled for 2025) will provide a baseline which can be used as a reference condition subsequently. Finally, the application of the quantitative Categories and Criteria will ensure consistency and comparability between countries and over time.

To support provision of these measures, the Red List of Ecosystems is managed, developed and communicated through a Red List of Ecosystems Thematic Group in the IUCN Commission on Ecosystem Management. Its three objectives are to assess all ecosystem types of the world by 2025, and update these assessments at regular intervals subsequently (e.g., every five years); to provide technical support (e.g., training, guidelines, peer review, advice) to the development of ecosystem red lists at the sub-global levels; and to support the use of the Criteria and Categories in the assessment of individual ecosystem types of particular interest to stakeholders. A Committee for Scientific Standards promotes the application of high scientific standards, is responsible for quality control, and ensures impartiality of assessments and that the intent of the Red List of Ecosystems Criteria is not compromised. Global assessments led by IUCN are reviewed to ensure integrity in the ecosystem description, correct application of the Red List of Ecosystems Criteria, and sufficient support by robust scientific data. National level ecosystem red lists are similarly encouraged to seek independent peer review for their assessments. A IUCN Red List of Ecosystems Guidebook will help to guide assessments at national, regional and global levels (Rodríguez et al., 2015).

3.3 Suitability of approaches for measuring extinction risk for biodiversity accounting

The IUCN Red List of Threatened Species is described below in relation to the five suitability criteria for biodiversity accounting listed above.

3.3.1 IUCN Red List of Threatened Species

The IUCN Red List of Threatened Species (http://www.iucnredlist.org; IUCN 2014; hereafter "Red List") measures species extinction risk. The objective of the Red List is to inform and catalyse action for biodiversity conservation by providing information and analyses on the world's species, especially on their status, trends, and threats. It was established in 1964 (Smart et al. 2014), and now assesses extinction risk for 76,199 species. The process to develop the IUCN Red List Categories and Criteria (Mace et al. 2008), the standard against which the Red List is implemented, was initiated with the "Road to Extinction" conference in 1984 (Fitter & Fitter 1987). This led to intensive effort through the IUCN Species Survival Commission to draft categories and criteria (Mace & Lande 1991), finally approved in 2001. The Criteria encompass population size reduction (A), small and declining geographic range (B), small population size and decline (C), very small or restricted population (D), and quantitative analysis (E). All assessments follow the current versions of the IUCN Red List Categories and Criteria, Version 3.1 (IUCN 2012a) and the Guidelines for Using the IUCN Red List Categories and Criteria (IUCN SPSC 2014).

The Red List is produced and managed by the IUCN Global Species Programme, the IUCN Species Survival Commission, and the IUCN Red List Partnership (BirdLife International, BGCI, CI, Microsoft, NatureServe, Kew, Sapienza University of Rome, Texas A&M University, Wildscreen, and ZSL). The Red List Unit, based in Cambridge, UK, is responsible for managing the data underlying the Red List, maintaining the Red List websites, verifying Red List assessments, creating derived data products and analyses, and providing training to Red List Assessors. A Standards and Petitions Sub-Committee is responsible for ensuring the interpretation and application of the Red List Categories and Criteria and for ruling on petitions against the listings of species on the Red List. The entire process requires multiple stage review and quality control[1] and incorporates a transparent challenge process for any petitions against published assessments.

IUCN authenticates and publishes online the global extinction risk assessments of all species evaluated for the Red List, including species with low risk of extinction which are listed as Least Concern. Submission of assessments must follow the appropriate guidelines, meet the required supporting information standards, and be independently reviewed to assure objectivity before final checking by the Red List Unit. Guidelines setting out the process for conducting national and regional level species assessments are also published by IUCN (2012b). Regional Red List assessments conducted by IUCN, such as those undertaken in Europe with support of the European Commission, are authenticated according to the regional assessment guidelines and featured on the website as special initiatives. National red lists (Zamin et al. 2010) are led by national-level institutions, including government agencies or non-governmental organisations, and collated by the National Red List Alliance, currently chaired by the Zoological Society of London. There are more than 26 regions, 113 countries, and 45 sub-national entities that have already developed Red Lists (National Red List, 2012). National red lists do not necessarily follow IUCN standards, but where they do, national

[1] http://cmsdocs.s3.amazonaws.com/keydocuments/Rules%20of%20Procedure%20for%20Red%20List_2013-2016.pdf

assessments of national endemics have the potential to inform in turn the global Red List, where these species are unassessed at the global level (e.g., for many plants, fungi, and invertebrates). For accounting purposes it is suggested that countries do follow the guidance for national Red Lists and seek help from IUCN in its application if needed. For brevity, information about the breath of the Red List and training materials available can be found in Annex 2.

Where entire taxonomic groups have been comprehensively assessed against the IUCN Red List Categories and Criteria two or more times, Red List Indices can be derived to yield indicators of change over time in the aggregate extinction risk of these species groups (Rodrigues et al. 2007). The methodology was first published by Butchart et al. (2004), overcoming challenges (Cuaron 1993) to previous attempts by considering only those species changing categories between assessments owing to genuine increases or decreases in extinction risk, and excluding changes resulting from new information or revised taxonomy (Brooks & Kennedy 2004). A revised methodology (Butchart et al. 2007) has now been applied across multiple taxonomic groups (Butchart et al. 2005, 2010, Hoffmann et al. 2011). It can be used to provide a counterfactual against which impacts of conservation action can be monitored (Hoffmann et al. 2010, 2015). The dates of earliest data points vary among taxonomic groups (e.g., mammals 1996, birds 1988, amphibians 1980); uncertainty is generated around the indicator based on those species assessed as Data Deficient (Butchart et al. 2010). The Red List Index can be disaggregated geographically to allow reporting at national and regional levels (Han et al. 2014, Rodrigues et al. 2014) and also disaggregated to report trends for driven by different threats (Salafsky et al. 2008), such as invasive species (McGeoch et al. 2010), utilisation (Butchart 2008) or pollution (Tittensor et al. 2014), or by taxonomic or functional group (e.g., pollinators; Regan et al. 2015). Guidelines for calculating Red List Indices at the national or regional level are available (Bubb et al. 2009), and an increasing number of national applications have been published (CBD, 2014) (see **Table 2**).

Table 2. Countries known to have conducted repeat National Red List (NRL) assessments (includes some countries known to have produced national-level Red List Indices or RLIs) (CBD, 2014)

Countries with repeat NRL assessments	Year of assessments
Ecuador	2000, 2011 (endemic plants)
Finland	2001, 2010 (all taxa)
Norway	2006, 2010 (all taxa)
Spain	2008, 2010 (vascular plants)
Sri Lanka	2000, 2007, 2012 (all taxa)
Sweden	2005, 2005, 2010 (all taxa)
Switzerland	2001, 2010 (birds)
Thailand	1990, 2005 (all taxa)
Vietnam	1996 (published again as a full Red Data Book with mammals in 2004), 2007 (plants)

The Red List therefore clearly meets the five suitability criteria discussed above. The global dataset of 76,199 species is fully coded to national occurrence, and methods are available to allow disaggregation of the Red List Index to national levels (Rodrigues et al. 2014). Downscaling of the global Red List to national levels can be complemented with national red lists, where these exist (Zamin et al. 2010). Using both the global Red List and national red lists enable different information to be picked up, for example, endemic species and other species that have not yet been globally assessed but would be noted in a national assessment.

The temporal resolution of re-assessments is on the order of five years; while relatively coarse for national accounting purposes, this is the finest resolution possible given the fact the changes in species extinction risk unfold over periods of many years. The full global dataset is classified to a standard habitat classification scheme[2], which can be crosswalked to ecosystem classifications and so provide a link between species and ecosystem accounting. The earliest year of comprehensive assessment is designated as the reference baseline; subsequent assessments are backcast to this point if changes resulting from changing knowledge (rather than change) are identified. Finally, the application of the IUCN Red List Categories and Criteria ensures consistency in assessment over space, over time, and between assessors.

3.4 Suitability of approaches for measuring species abundance for biodiversity accounting

The Norwegian Nature Index and Living Planet Index methodologies are described below in relation to the four data suitability criteria for biodiversity accounting.

3.4.1 Example 1 - The Norwegian Nature Index methodology

The Norwegian Nature Index provides an overview of the state of biodiversity in Norway's major ecosystems using more than 300 indicators. Indicators are chosen from a variety of species groups for each ecosystem, and measure deviation from a reference state, which is intended to represent ecological sustainability (Norwegian Environment Agency, 2012). This makes it particularly suitable as a biodiversity indicator of ecosystem condition. The Nature Index is included in the Norwegian official set of indicators for sustainable development, reported annually by Statistics Norway and by the Ministry of Finance in the National Budget (Norwegian Institute for Nature Research, 2015).

The Nature Index incorporates expert judgement, monitoring-based estimates, and model-based estimates, so the method can be used in both data rich and data poor areas. It relies on a network of scientific experts, each of whom is responsible for one or more biodiversity indicators. Within the index framework, natural systems are termed "major ecosystems" and are categorized into a set of nine broad natural system types, i.e., mountain, forest, open lowland, freshwater, mires and wetland, coast pelagic, coast bottom, ocean pelagic, and ocean bottom. All indicators and the overall Nature Index have values between 1 (for the reference state) and 0 (very poor state). The first edition of the Nature Index was published in 2010 and values were calculated for 1950, 1990 and 2000 as well as 2010. It should be noted that whilst the Nature Index provides a methodology for aggregating indicators of biodiversity change across ecosystems, it does not provide a solution for generating such indicators. As such the applicability of the approach to an accounting framework is dependent on the existence of biodiversity indicators for all the different ecosystems within any proposed accounting application.

To account for the fact that not all taxa, functional groups, or geographical areas can be studied to the same degree, indicator values are weighted. The results of the index can be presented at several aggregated levels and the choice of resolution depends on the underlying question addressed. A summary for how the overall index is calculated is shown in **Figure 5** and the full method is described in Certain et al. (2011).

The Nature Index illustrates a good example of a using national indicators to assess the status and trends of biodiversity for different ecosystems but also demonstrates how to get a single 'value' that can provide an indicator of ecosystem condition. The spatial resolution and temporal relevance of the final Index is subject to data availability. With sufficient input of resources to make data available, the framework provides the flexibility to organise biodiversity at a range of scales in a format suitable for aggregation.

The Nature Index methodology has also been tested on Costa Rican forest ecosystems in collaboration with experts from Costa Rican research institutions (Barton et al., 2014). In this study 11 indicators were used. This reveals the flexibility of the framework for use in different accounting contexts.

[2] http://www.iucnredlist.org/technical-documents/classification-schemes/habitats-classification-scheme-ver3

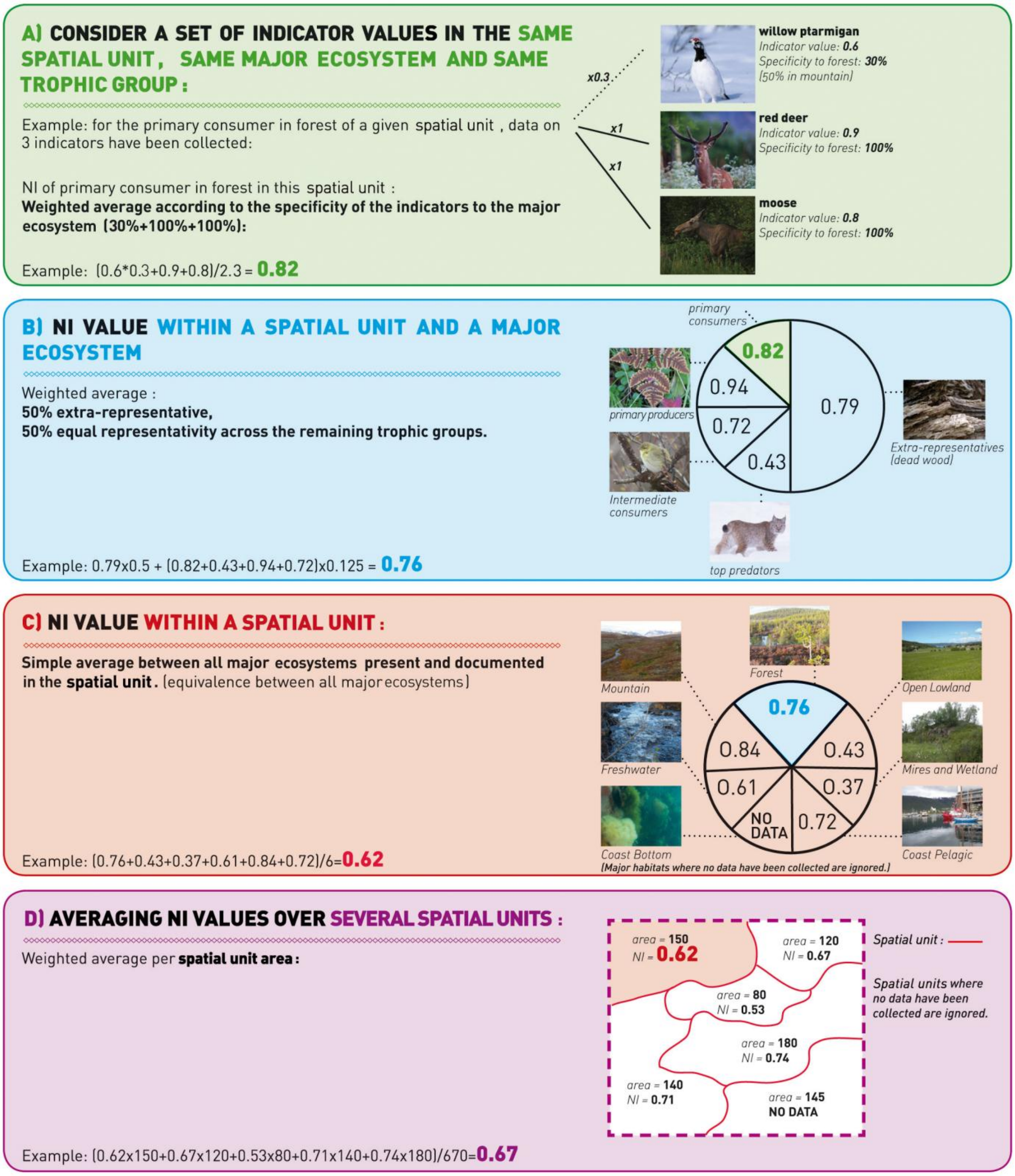

Figure 5. Example of the aggregation process from a set of indicators in a single spatial unit (A) to averaging several spatial units (D) that is used in the calculation of the Nature Index, including the weights used (Certain et al., 2011).

3.4.2 Example 2 - Living Planet Index methodology

The Living Planet Index (LPI) is an indicator of change in biodiversity that uses measures of change in species population abundance. The strength of the LPI methodology is that it can aggregate population trend data from different sources that is across multiple spatial scales. The species population data in the central database used to calculate the index are collected from a variety of sources including: scientific journals, government reports as well as wildlife and other natural resource management authorities records (McRae et al., 2008).

Data consists of measures of either population estimates or proxies for population size, e.g., density measures, for example the number of birds per kilometre of transect. All population time series have a minimum of two data points, collected by methods that are comparable across years, so that trends can be determined. Data is then aggregated by species and then into sub-indices depending on whether they are temperate or tropical and their system (terrestrial, freshwater or marine). The global index is then the geometric mean of the three system sub-indices. The final trend line represents the average change within the entire collection of population samples within the study period, giving equal weight to each species, whether common or rare, and to small and large populations (McRae et al., 2008).

Application of the LPI method to create an indicator that is representative of biodiversity at a national scale is usually constrained by data availability and can be biased by well-known taxonomic groups and well-studied locations. The more equal the distribution of data across the country, the finer the spatial resolution the Index can be aggregated to for accounting purposes. Application of current LPI data at the national scale requires an examination of data coverage to test for certain biases, and interpret it accordingly. At the national level it may also be possible to complement existing data within the central database with new studies that help to fill gaps (McRae et al., 2008).

In the global LPI, the baseline used is 1970 and is produced bi-annually. However, the temporal period used and the frequency of update are both dependent on data availability. With sufficient data, any temporal period and reference point can be used and the Index can be updated as and when data becomes available.

The LPI method has been used to create a national Index in Uganda as part of a ‘State of Uganda’s biodiversity’ report (Pomeroy & Tushabe, 2008; NEMA, 2011). The Living Planet Index for Uganda combines the trends from the species population indices of Uganda’s forests, freshwaters and savannahs.

Data within the LPI is obtained from variety of spatial scales and is often heterogeneous. With systematic monitoring of species abundance, the data would lend itself for incorporation in a Biodiversity Account. For those countries that lack this systematic data, the methodology can still yield a useful indicator of ecosystem condition. If there is insufficient data to generate an Index to represent biodiversity as a whole, than various informative sub-indices can be generated from the available data. For example, if there was only data for ecosystem specialists, than an Index of specialist species could be generated (e.g., reef dependent species) that could be used as an indicator of ecosystem condition.

4. Data mobilisation for biodiversity and ecosystem condition accounting

Key points

- The scale at which biodiversity information is collected must be relevant to its intended use.
- As a first step, an inventory of existing biodiversity information at the national scale should be prepared, drawing on monitoring schemes and reporting obligations to regional processes and biodiversity-related conventions/agreements.
- Modelling, expert judgement and other approaches may be useful for upscaling or downscaling information on biodiversity to inform biodiversity accounting.
- Confidence in estimates of species diversity from modelling, expert judgement and other estimation approaches can be improved with 'ground-truthing'.

This section describes the data mobilisation process to inform biodiversity and associated aspects of ecosystem condition accounting. It discusses possible sources of national species data, as well as the potential use of global datasets and modelling if data at the national scale is limited.

4.1 Mobilising biodiversity data at national or local scales

Figure 6 provides a systems view of the components and activities to mobilise biodiversity information at national or local scales. The framework promotes the view that information mobilised for biodiversity accounting should start by defining information needs of the users and the key questions this information will be used to answer. Ideally, Biodiversity Accounts will be constructed on the basis of comprehensive in-country monitoring programmes that are undertaken on a regular and consistent basis.

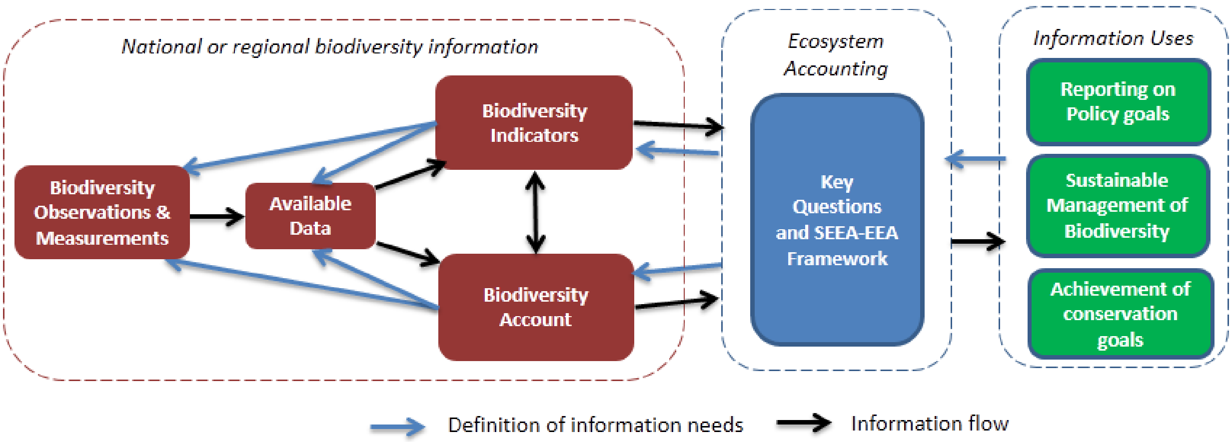

Figure 6. Framework for functions and activities for mobilising biodiversity data for ecosystem accounting. The blue arrows indicate a specification of the information required for the principle information used (green boxes). The black arrows represent the return flow of information via the accounting system (adapted from a diagram produced in the EU BON Project).

The blue arrows in Figure 6 represent stages in the definition of information needs, starting with the definition of key questions and conceptual frameworks from the identification of users' needs. These in turn guide the selection and scoping of what information should be recorded in the Biodiversity Accounts and which indicators of biodiversity are required for the Ecosystem Condition Account. This sets the parameters for measuring biodiversity (e.g., is data on species abundance, richness or threat status required and at what spatial and temporal scale) and associated processing activities (e.g., GIS, spreadsheets, modelling etc.).

With respect to spatial and temporal scales for collecting information, it is important that this addresses the information uses presented in the green boxes in Figure 6. This will be context specific. In some circumstances biodiversity monitoring will be required at a fine scale. For example, for the management of small protected areas to achieve conservation goals. In other circumstances, biodiversity information will be required at a broader scale that can be related to economic activity. This could be administrative areas, for instance. With respect to temporal scales, annual monitoring is ideal from an accounting perspective. However, this will often be impractical. A pragmatic approach may be to coordinate monitoring programmes with relevant policy cycles (e.g., environmental or sectoral).

To account for ecosystem diversity it is important that biodiversity information is captured for all habitats or land classes that are considered within the accounting context. However, the resources devoted to different ecosystem types should clearly reflect the policy questions the accounting information is intended to inform. For instance, a country may be particularly interested in improving biodiversity in their wetlands to maintain (or even increase) the delivery of regulating ecosystem services to reduce agricultural pollution, for example. Therefore, it would make sense to devote more resources to monitoring wetland areas.

Determining species of national importance is one approach for prioritising the use of species data for accounting purposes. Such an approach may also help to make the case for continued (or even additional) resources for monitoring programmes. The selection of species data will depend on how the information coming out of the account will be used. A list of example priority species is provided in Section 2.3.

4.1.1 Identifying national-level species data

A pragmatic first step in constructing accounts of biodiversity is to compile an inventory of existing species data at the national scale. **Box 8** describes possible sources of these data.

Box 8: Sources of species data for use in national biodiversity accounting

Monitoring and reporting of the status and trends of species at the national level takes place at multiple spatial scales:

National and sub-national monitoring schemes
Compiling an inventory of existing national biodiversity monitoring schemes is a good starting point to determine possible sources of species information and data. Examples include, butterflies (e.g., Israel's Butterflies Monitoring Scheme (Peer, 2015)), birds (e.g., New Zealand's Garden Bird Survey, which is a nationwide citizen science project (Spurr, 2012)) and bats (e.g., the North American Bat Monitoring Programme (USGS, 2015). Local scale schemes where data is collected through citizen science may also be useful.

Other sources of biodiversity information include: national NGOs, universities and museums; National Red List Assessments (see Section 3.3.1); and for members of the Global Information Facility (GBIF), the GBIF node, which is the dedicated institution that coordinates and manages biodiversity data in a country (GBIF secretariat, n.d.).

Some species data may be outside the country, for instance, Kew Royal Botanic Gardens in the UK houses botanical and mycological collections from around the world. Further initiatives, such as the Crop Trust, oversee multiple global gene banks to safeguard crop diversity.

Reporting to regional processes
Separate processes through which information related to the indicators could be retrieved, are through regional reporting obligations, such as the EU's obligation on member states to produce reports on the State of the Environment (SoE). This SoE reporting takes places every five years and

assesses the European environment's state, trends and prospects in a global context (EEA, 2015). It brings together a wide range of environmental information to assess trends in terrestrial, freshwater, marine and coastal biodiversity, including the conservation status of species and habitats. A number of other EU instruments are relevant to biodiversity and can provide data to support national biodiversity accounting, e.g., the Habitats Directive (since 1992), which, together with the Birds Directive, forms the cornerstone of Europe's nature conservation policy; the Water Framework Directive (since 2000); and the more recently implemented but more holistic Marine Strategic Framework Directive (since 2008).

National governments that are Contracting Parties to one or several Regional Seas Conventions also collate and contribute data relevant to biodiversity accounting as part of their reporting obligations. For instance, OSPAR's Quality Status Reports (2000, 2010) are milestones for evaluating the quality status of the North-East Atlantic and for taking forward OSPAR's vision of a clean, healthy and biologically diverse sea. These reports are based on 10 years of joint monitoring and assessment by OSPAR Contracting Parties of the marine environment and provides the scientific basis for these holistic reports.

Reporting to international conventions/agreements
Possible sources of biodiversity data of relevance to calculating Biodiversity Accounts include the templates used to report progress in implementing biodiversity-related conventions/agreements. These are a subset of Multilateral Environmental Agreements (MEAs) that form a key part of the International Environmental Governance (IEG) landscape, providing a legal framework through which countries negotiate and cooperate on the protection of biodiversity. Biodiversity-related conventions provide Parties with a space to convene and produce overarching plans and strategies to guide coordinated national action. Examples of useful data are:

- **Trend in the status of threatened species**: Parties to the Ramsar Convention are asked to report on species of fauna in Ramsar sites that are of particular concern (e.g., unique, rare, endangered or bio-geographically important).
- **Species abundance**: Several conventions/agreements ask Parties to report on populations of species. For example, of Appendix I Species under the Convention on Migratory Species and of the main populations of fauna in the Protocol Concerning Specially Protected Areas and Wildlife (SPAW Protocol) annexes, Red List and national protected species.

4.1.2 Using global data

If national level data is limited, a further source of data is to consider suitable global datasets. For example, the IUCN Red List of Threatened Species provides a potentially useful database on ranges of a large number of species (i.e., not just for threatened species). The ENCA-QSP (Jean-Louis Weber, 2014) provides a good discussion on the application of IUCN data for informing ecosystem accounting (this will be relevant to Biodiversity Accounts of species and as a proxy for ecosystem condition). The IUCN website[3] provides spatially explicit datasets on a number of species and their habitat preferences, which would facilitate mapping to different ecosystem units. However, it is unclear how often these are revised and updated.

The IUCN also provides tables on the change on the status of species (at the global scale and only for some taxonomic groups). However, this needs to be supplemented with further data to obtain the status of species at the national scale (as discussed in Section 3). Nevertheless, this could provide the basis for informing the opening and closing stocks of species status and provide proxies for ecosystem condition (Jean-Louis Weber, 2014). This data would also have the potential to inform Biodiversity Accounts for species of cultural significance (e.g., status of important species for

[3] http://www.iucnredlist.org/technical-documents/spatial-data

conservation). As the ENCA-QSP observes, aggregated information on the threat to species could be useful as a first estimate of a biodiversity indicator for the Ecosystem Condition Account (Jean-Louis Weber, 2014).

A number of studies have sought to map biodiversity at a global scale. For example, drawing on the SEEA-EEA approach, Dickson et al. (2014) present the world's first global map of key ecosystem assets, including biodiversity. The analysis identifies clear synergies between biodiversity and maintaining terrestrial organic carbon stocks, notably in tropical areas. Dickson et al. (2014) present a global map of terrestrial species richness compiled using the IUCN data on species occurrence (i.e., spatial maps of species presence or absence), which are overlaid to provide an overall indication of species richness. However, the IUCN database on species occurrence is compiled from studies on species ranges that span a period of several decades.

For the marine environment, Tittensor (2010) presents a global data layer of species richness for a subset of marine taxa, although the supporting studies are over a period of around a decade. As such, it is unlikely that these datasets will have the temporal resolution necessary to provide the time series data necessary for biodiversity accounting. Whilst the temporal resolution could be improved, this would require a large global investment in a structured international monitoring programme.

Ideally, species diversity data should be at a spatial resolution that allows data to be assigned directly to ecosystem units. This implies the resolution of species diversity data should broadly correspond to resolution of the ecosystem units or for specific habitat areas within the seascape. This will depend on the landscape context on which the accounts are compiled. It is unlikely that global spatial datasets will achieve this level of resolution with confidence. For example, Dickson et al. (2014) employ a 50km x 50km grid of basic spatial units and Tittensor (2010) use a 880km x 880km grid. Whilst a finer scale grid could be overlain on the source data, the resolution adopted reflects an acceptable level of uncertainty in assigning species richness spatially. Jetz et al. (2012) discuss the issues relating to the coarseness of global data on species distribution further. They propose a framework for developing and refining species distribution data further and generating a quality 'Map of Life' for species. However, this work has yet to be progressed.

A second source of biodiversity data discussed in the ENCA-QSP (Jean-Louis Weber, 2014), is the global Biodiversity Information Facility (GBIF)[4]. This database comprises information on species observations across the planet. However, there are many distributional gaps in these data that can introduce bias. Furthermore, as this data relates to point observations on species so processing or modelling would be required to generalize species abundance and/or richness estimates for ecosystem units for which studies are absent.

4.2 Other methods of estimating biodiversity data

Surveying the entirety of species ranges is often not possible (Mant et al., 2014). Therefore models or expert judgement are often used to assess the most likely distributions and abundances of species based on key environmental drivers. The ENCA-QSP (Jean-Louis Weber, 2014) suggests ecological niche modelling may be useful in this regard, where ecological characteristics (e.g., altitude, land cover type etc.) are mapped to known occurrences of species (e.g., as determined from the GBIF database). This allows species presence in non-sampled areas to be inferred on the basis of similarities in ecological characteristics. Linking this approach with maps on land cover, land use and / or habitat that are updated on a regular basis would allow opening and closing stocks of species diversity to be determined on the basis of associated changes of ecological characteristics of ecosystem units. However, this should be supported by comprehensive monitoring programmes in order to calibrate the approach. Determining a reference condition could also be achieved in the basis of modelling a set of 'ideal' ecological characteristics (i.e., based on minimal human disturbance).

A number of models exist that can perform ecological niche modelling using various statistical approaches. For example, the software package Maxent can be used to estimate species richness (e.g., Phillips et al., 2006). Other statistical approaches can include the use of Generalized Linear Models or Spatial Linear Models (e.g., Jetz & Rahbek,

[4] http://www.gbif.org/

2002) and Generalized Dissimilarity Models (Ferrier et al., 2007). **Box 9** describes how these types of approaches can be used for informing ecosystem accounting for in Peru.

Box 9: Use of biodiversity models for biodiversity accounting in Peru (Grantham, 2015, pers com.)

Background

The following describes briefly the biodiversity accounting approach that Conservation International, the Government of Peru and CSIRO are currently experimenting with in San Martin, a region within Peru. In this approach biodiversity is being accounted for both as an input to the *Ecosystem Condition Account* and some additional *Biodiversity Accounts* to complement this.

Peru is a megadiverse country with vast biodiversity values. San Martin has had the highest level of deforestation in the country and is an important biodiversity priority, while also being very important for agriculture, forestry and indigenous issues. Policies that these data might inform include protected areas planning, forest zoning, conservation concessions, forestry concessions, payments for ecosystem services, and green development strategies.

Methods

Biodiversity is being measured in two ways: i) using Generalized Dissimilarity Modelling (GDM; Ferrier et al., 2007); and ii) as individual species distributions.

GDM is a statistical technique for modelling compositional dissimilarity – i.e. the proportion of species in a given biological group (e.g. reptiles) occurring at one location that do not occur at a second location – as a function of the environmental characteristics of these two locations. Inputs to the GDM model are species occurrence records (currently based on data aggregated by the Global Biodiversity Information Facility for plants and invertebrates, and with vertebrates value-added by Map of Life) and various environmental layers[1] that correlate with patterns of biodiversity distribution. The primary outputs from the model are nonlinear transformations of the environmental layers that define a multidimensional environmental space within which environmental distances between locations correlate as closely as possible with observed dissimilarity in species composition between these locations. This transformed environmental space is used to predict the compositional dissimilarity between any two locations within the region, regardless of whether biological data are available for these locations. This compositional-dissimilarity model is then related to a biodiversity condition layer (comprising of habitat fragmentation, degradation and, for vertebrates, defaunation) so that a prediction of the proportional change in collective species (gamma) diversity for the region can be made for different biological groups. Currently, an existing global model at 1km^2 resolution is being used. Time and funding permitting, the model will be revisited post July 2015 with potentially new biological data and a few extra, locally available, finer-resolution environmental layers (e.g., soils and climate).

For individual species distribution the approach being trialled uses existing species models that were previously developed by Natureserve using Maxent (Phillips et al., 2006). Models of predicted habitat distribution are being used and the change within these areas based on forest change and condition (fragmentation, degradation, and defaunation for vertebrates) is being measured. The criteria for which species might be included are yet to be finalised but it is likely to be at least Red List species and perhaps other important species (e.g., restricted range).

Core accounting tables

For biodiversity as a condition measurement (Table 1) the GDM-based score of the proportion of overall species diversity retained within each ecosystem type is being derived. The score is between 0-1 so easily integrated with other metrics for a composite index.

Table 1. Potential structure of the Ecosystem Condition Account (not all ecosystems are shown)

Time 2009		**San Martin**							
Ecosystem Assets									
		Extent		**Condition Scores**					
		Current Area (ha)	% Original	Biodiversity	Fragmentation	Degradation	Hydrological Function	Soil Regulation	**Composite Index**
Forests	Bosque Humedo de Colina Alta								
	Bosque Humedo de Colina Baja y Lomada								
	Bosque Humedo de Montana								
	Bosque Humedo de Terraza Alta								
	Bosque Humedo de Terraza Baja y Media								

For Biodiversity Accounts something along the lines of Table 2 and Table 3 is being explored. Table 2 measures the GDM-based score across broad taxonomic groups and over time to gain trends in biodiversity. Table 3 measures the change in specific species in terms of their extent and condition of habitat. As these measures are normalized they can be combined together into a single score.

Table 2. Potential Biodiversity Account measuring change in biodiversity (GDM) over time

Biodiversity group	Ecosystem Accounting Unit		
	Original	**2009**	**2012**
	Score	Score	Score
Plants			
Vertebrates			
Invertebrates			

Table 3. Potential species account measuring change in extent and condition of important species. Condition likely based on fragmentation, degradation and defaunation (the latter for vertebrates only).

Species	Ecosystem Accounting Unit				
	Benchmark	**2009**		**2012**	
	Extent (Ha)	Extent (%)	Condition	Extent (%)	Condition
Species 1					
Species 2					
Species 3					
Species 4					
Final score					

Key lessons

A number of lessons learned from experimenting with Biodiversity Accounts have been identified so far:

- Biodiversity across larger spatial extents is not an additive function of biodiversity within smaller extents, so it is not possible to simply add local biodiversity scores across ecosystems to get an overall measure of biodiversity change.
- Complementary approaches are being explored with: 1) GDM representing the majority of biodiversity; and 2) the individual species being specific biodiversity of interest (e.g., known species that might be lost). The ecosystems themselves are also a surrogate for representing species patterns. This has not yet been explored yet.
- For biodiversity as an ecosystem condition metric, first an overall assessment of condition for biodiversity is developed, and then combined with the species level or

compositional models to generate measures of the effectiveness of habitat, scaled by the respective models. In this context, GDM is being used to measure the change (loss or gain) in overall diversity within an ecosystem. The biodiversity condition metric therefore needs to be a general estimate of habitat condition for biodiversity, varying between 0 and 1, incorporating spatially explicit knowledge of degradation processes. It combines with and adjusts the specific biodiversity measures so that they represent the amount of habitat available for an individual species or the proportion of overall diversity retained, lost or gained, depending on the question being addressed.

- We are not measuring species richness change at the basic spatial unit (BSU). If an estimate of this was of interest, the biodiversity condition index might be used for this purpose.
- The use of ecosystems within the Biodiversity Accounts, which is currently more focused on species, are also being considered, such as the use of species area curves as part of this. Crop wild relatives is also one option for thinking about the genetic level focus of biodiversity.
- A biodiversity change account that incorporates land use to identify drivers of biodiversity change is also being explored.

[1]Climate data were derived from temperature and precipitation surfaces from www.worldclim.org, adjusted for radiative variation due to terrain (DEM: GMTED2010 from https://lta.cr.usgs.gov). Substrate/landform data adapted from www.soilgrids.org and www.worldgrids.org

For marine applications, the Aquamaps tool provides an ecological niche modelling approach to predict marine species richness on a spatially explicit basis (Kaschner, 2007). The key drivers in this model are depth to seafloor, temperature, salinity, primary production, sea ice and distance to land. Unlike terrestrial drivers of biodiversity loss, these drivers are likely to be harder to link to changes in local human activities.

There also exists several proprietary terrestrial models that develop the ecological modelling approach further by linking key human drivers of biodiversity loss to biodiversity estimates. The SEEA-EEA identifies the terrestrial model GLOBIO (Alkemade et al., 2009). Others include PREDICTS (Newbold, 2015). These approaches are based on a meta-analysis of global datasets containing large numbers of existing site-level studies on species distribution and abundance. Each study site is scored for the levels of the key drivers of biodiversity loss, for example, land use, land use intensity, land use history, population density and proximity to roads (proxy for habitat fragmentation) (Newbold et al., 2015). The meta-analysis allows a set of coefficients to be estimated for this set of key drivers, these can then be employed to predict biodiversity estimates at a site for which no original study exists. The relative species diversity estimates reported by the models described above comprise aggregated estimates of species abundance (mean species abundance) and richness (means species richness).

Where data on the key drivers of biodiversity loss for each ecosystem unit is available, these types of models could be used to generate pilot accounts of terrestrial biodiversity. The basic approach would link ecosystem units to the different land uses (and possibly intensities) that are recognised by the modelling software. This is likely to require support from a GIS specialist. The accounts could be improved by additional information on population and transport infrastructure, which is also likely to be available in a GIS format.

These models could be improved further by developing an underlying set of coefficients that are country specific. This could be achieved using a subset of biodiversity studies relevant to the country (or region) in question, possibly supplemented by expert knowledge (as per Scholes & Biggs, 2005, see below).

As the outputs from these models comprise estimates aggregated across species they are useful for informing the Ecosystem Condition Account (in terms of the characteristics of biodiversity). They will also be helpful in understanding resilience in the context of ecosystem assets and their capacity to deliver ecosystem services (Newbold et al., 2015). However, such aggregated estimates are not particularly informative for the core accounts of biodiversity. Adapting these models to report on estimates of different taxonomic groups could provide the necessary resolution in terms of species groups to inform pilot core accounts of biodiversity.

An important issue relating to the use of such models, is the underlying assumption of a linear relationship between the drivers of biodiversity loss and estimates of species diversity. This can be a dangerous assumption, as the relationship between ecosystem integrity and biodiversity will often have thresholds and nonlinear effects. Accordingly, the ENCA-QSP recommends that measures of biodiversity should be based on real monitoring data (Jean-Louis Weber, 2014).

In addition to meta-analysis and ecological niche modelling, simulation models may also be used to predict species diversity in a spatial manner. These 'General Ecosystem Model' (GEM) approaches (e.g., The Madingley Model[5]) are conceptually similar, in that they link biological approaches with human pressures, including land use change. They remain rooted in global datasets but rely on simulation approaches, rather than direct empirical relationships, to predict the structure of ecosystems.

A related approach is to generate relative estimates of species diversity on the basis of expert judgement. This is the approach proposed by Scholes & Biggs (2005) for the development of the Biodiversity Intactness Index (BII). Under this approach, ecological specialists are asked to estimate reductions in populations of broad taxonomic groups for given land cover classes (or other ecosystem classification process) due to land use activities. These estimates were referenced relative to a large protected area of that land cover class. This approach is presented for Southern Africa in Scholes & Biggs (2005) and may provide a useful starting point for piloting accounts of biodiversity.

Primary monitoring data should inform the core information presented in Biodiversity Accounts. However, modelling and expert judgement provide useful approaches to use this point data to infer biodiversity stocks across spatial areas (i.e., via upscaling or downscaling of biodiversity information). These approaches are likely to feature significantly when piloting Biodiversity Accounts in the first instance. Confidence in their application can be increased through time by 'ground-truthing' as part of a validation and calibration programme. This will also increase the inventory of primary monitoring data through time.

[5] http://www.madingleymodel.org/index.html

5. Producing Biodiversity Accounts and core tables

Key points

- The first step is to delineate ecosystem assets spatially on the basis of compositional similarities to provide 'Extent Accounts' for a set of explicit Ecosystem Units.
- The spatial resolution of ecosystem units used in accounting should be driven by information needs. It may also be more appropriate and pragmatic to collect biodiversity data across an area of interest (e.g., a watershed [Ecosystem Accounting Unit]) rather than for each Ecosystem Unit.
- Further testing is required of useful scales for collecting biodiversity information for accounting.
- The Biodiversity Accounts and case studies presented here may be implemented at Ecosystem Units, Ecosystem Accounting Units or Administrative Accounting Units scales, according to the availability and resolution of data.
- 'Tier 1' Ecosystem Extent Accounts follows the Natural Capital Index approach and provides a framework for organising information on ecosystem extent and diversity. The measures of extent can be weighted using measures (or input indicators) of species diversity. This may reduce the resources needed but provides limited information on the actual 'stock' of species.
- 'Tier 2' Species Richness Accounts can be produced by collecting data on species ranges and occurrence. These can be supported with information on their extinction risk and possibly species health. However, these approaches provide limited information on the actual number of species present in an ecosystem.
- 'Tier 3' Species Abundance Accounts, based on species abundance data, provides the most detailed accounts but requires the most resources.
- Relative or even qualitative measures of biodiversity can be used in the Biodiversity Accounts.
- Further testing of downscaling, upscaling or other approaches to spatially allocate biodiversity data to Ecosystem Units, particularly for species richness and abundance, is needed.
- Further testing of methods of aggregating species diversity data, particularly relative measures, across ecosystem units is required. This should include evaluation of how to account for ecotones and gamma biodiversity in biodiversity accounting.
- More than one Biodiversity Account may be required to provide information on different biodiversity-related policy questions.
- More than one indicator of biodiversity may be required to provide information on both progress towards policy targets and on ecosystem condition.

This section builds on Sections 2, 3 and 4. It describes Biodiversity Accounts based on supplementary information on ecosystem diversity and the construction of 'species accounts'. These are illustrated with example accounting tables. The tables are presented in three tiers based on complexity and resource requirements. Considerations of key issues are also discussed and case studies are included where available.

5.1 Getting started - identifying the policy relevant question(s)

Producing spatially explicit accounts of biodiversity allows trends in biodiversity to be compared to policy targets and with economic and social statistics. This provides an opportunity to integrate biodiversity into a wider range of decision-making contexts. The information presented in the accounts should address key biodiversity-related policy questions. These may include species of economic importance (e.g., game species for ecotourism) or ecologically important keystone, umbrella or specialist species that may be at risk of extinction in the wild.

Within the SEEA-EEA, specific consideration is given to biodiversity as a resource whose stock it is important to maintain for future ecosystem service delivery. Therefore, measures of functional group diversity (defined by the MA

[2005] as groups of organisms that respond to the environment or affect ecosystem processes in a similar way) are also of interest to better understand threats to ecosystem functioning and service delivery arising from trends in biodiversity (e.g., pollinator diversity). This information is essential for the sustainable management of biodiversity.

The specific approaches adopted for quantifying biodiversity in ecosystems will be largely dependent on the policy context and the availability of data. A discussion of established approaches is presented in Sections 2 and 3.

5.2 Defining ecosystem units for biodiversity accounting

The SEEA-EEA (2014) describes ecosystems as assets that are a combination of biotic and abiotic components and other characteristics that function together. In order to establish a foundation for ecosystem accounting, it is necessary to define these ecosystem assets in a spatially distinct and explicit manner. Given that the broad purpose of ecosystem accounting is to account for ecosystem services, any delineation of ecosystem assets should be based on a set of common functional ecological components and characteristics (e.g., a contiguous area of montane coniferous forest). However, bringing these ecological concepts into the accounting framework is a key challenge acknowledged by the SEEA-EEA. This is discussed in SEEA-EEA Tech. Guide 1, 2015, which proposes the Ecosystem Unit (EU) as the spatially explicit accounting unit for ecosystem assets. Information from supporting accounts (including Biodiversity Accounts) can then be assigned to these units.

The EU is delineated on the basis of habitat characteristics, including biotic, abiotic and other linking compounds, being consistent across a collection of basic spatial units (BSUs) (e.g., a $100m^2$, hectare or km^2 grid). Consequently, defining a spatially explicit system of EUs requires expert GIS input. The size of the set of different EU classifications will depend on various factors, including: the number of different ecosystems in the landscape, the availability of classification data and the purpose or the intended use of information on EUs. The account of the extent of EUs (Extent Account) and information on their defining characteristics will provide significant information on biodiversity, particularly ecosystem diversity. This can be supported by assigning a specific habitat class codes to EUs that are relevant across the area of interest, as discussed in SEEA-EEA Tech. Guide 1 (2015). Depending on the scale adopted for defining EUs, consideration could also be given to including habitat information as subclasses of the EU type to provide further information on ecosystem diversity between EUs. Whilst the scale of EUs will depend on context, it is likely that they will be described in terms of hectares or km^2.

As a starting point, the SEEA-EEA suggests information from Land Cover Accounts could provide the framework for defining EUs. This may be the most pragmatic option when developing pilot accounts. SEEA-EEA Tech. Guide 2 (2015) sets out recommendations for deriving Land Cover Accounts. This details how individual Land Cover Functional Ecosystem Units (LCEU) can be delineated on the basis of land cover and land use similarities to provide coarse EUs. Again, the LCEU scale will be dependent on context but it is likely the units will be described in terms of hectares, km^2 or possibly 10s km^2. The SEEA Central Framework (2014) land cover classifications are set out in **Table 3**.

Table 3. The 14 land cover classification as described in the SEEA-Experimental Ecosystem Accounting framework (Table 5.12 in SEEA Central Framework, 2014)

	Category
1	Artificial surfaces (including urban and associated areas)
2	Herbaceous crops
3	Woody crops
4	Multiple or layered crops
5	Grassland
6	Tree-covered areas
7	Mangroves
8	Shrub-covered areas
9	Shrubs and/or herbaceous vegetation, aquatic or regularly flooded
10	Sparsely natural vegetated areas
11	Terrestrial barren land
12	Permanent snow and glaciers
13	Inland water bodies
14	Coastal water bodies and intertidal areas

However, as discussed in SEEA-EEA Tech. Guide 1 (2015), the delineation of areas on the basis of land cover will omit substantial detail on the variation in ecosystem functional characteristics within the LCEU. The example provided in SEEA-EEA Tech. Guide 1 (2015) is an area of wetland contained within the larger extent of a contiguous LCEU of 'Grassland'. The wetland will have different ecosystem functioning (and biodiversity) properties that confer specific ecosystem services. Therefore, from an accounting perspective, it would be useful to isolate the wetland as a distinct ecosystem unit. This will allow any management action to realise the full benefits of the ecosystem services provided by the wetland. Consequently, disaggregating the LCEU in to component EUs is likely to provide more useful units for capturing and modelling ecosystem services.

Maintaining the relationship between the EU and LCEU is useful for maintaining the link between biodiversity and the economic activity via the SEEA Central Framework (where land ownership or land use information can be assigned to land cover information). This can allow statistics on biodiversity to be examined against economic statistics associated with both land use and ownership. In this context the LCEU can also provide a useful unit for aggregation, **Figure 7** taken from SEEA-EEA Tech. Guide 1 (2015) summarises this below.

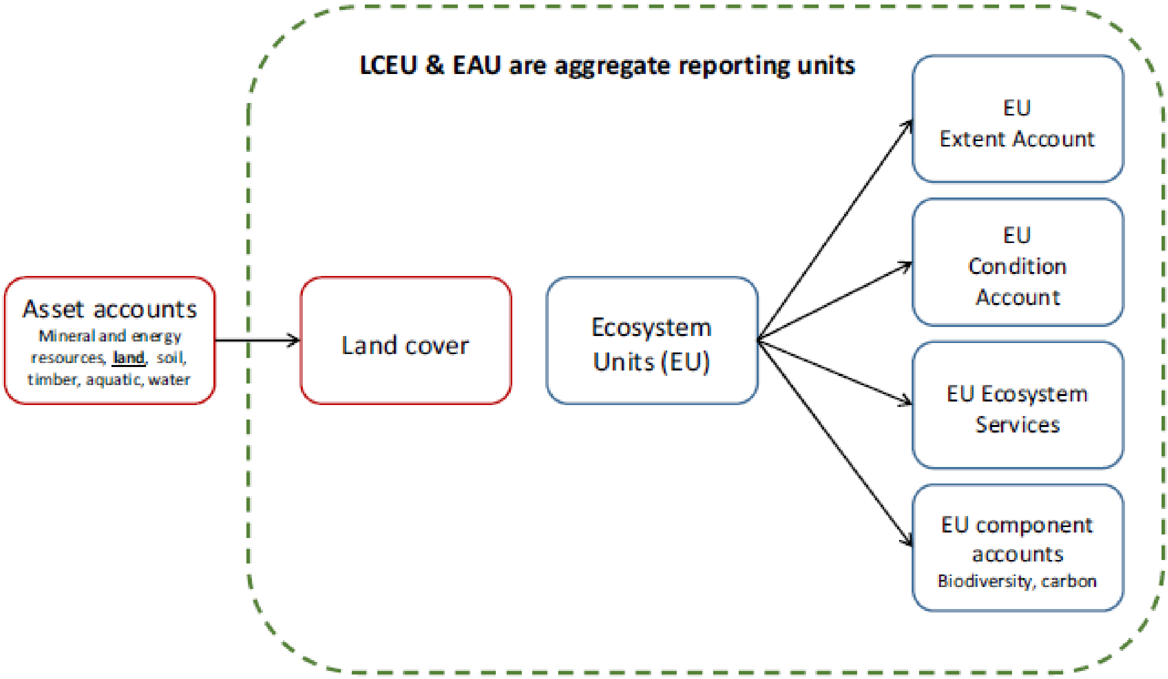

Figure 7. The potential link between Ecosystem Units (EUs) and the SEEA Central Framework. This figure illustrates the EU as an ecosystem asset, which is linked to the land asset in the SEEA Central Framework via its parent land cover class (SEEA-EEA Tech. Guide 1, 2015).

5.2.1 Scale and Reporting Units

Ideally, information on biodiversity should be collated at the EU scale to capture variation in ecosystem functionality. This can then aggregated to larger scales for reporting. However, whilst the wetlands example demonstrates the rationale for developing Extent Accounts of EUs at a fine scale, there are obvious resource implications for this task. Therefore, the decision on the appropriate scale at which to define EUs within the accounting framework should be based on policy need and the particular ecosystem services of interest. For biodiversity accounting the issue of scale is complicated further by the ability of individuals to move between different ecosystem units and the scale at which biodiversity can be reasonably measured or estimated. Again, the decision on spatial scale at which to capture information on biodiversity should be driven by policy needs and the priority species or aspects of biodiversity that are of interest.

In this regard, the SEEA-EEA acknowledges that it may be necessary to work with biodiversity data at broader spatial scales. SEEA-EEA Tech. Guide 1 (2015) suggests two sensible scales for aggregating EUs that would correspond with scales at which management decisions are undertaken. The first is the Ecosystem Accounting Unit (EAU), predicated on the basis of an ecological feature (e.g., watershed or bioregion). The EAU may provide a more appropriate scale for measuring biodiversity as ecological boundaries are more likely to correspond to species range boundaries. Downscaling approaches can then be employed to attribute information on biodiversity to constituent Ecosystem Units (e.g., as done for species and habitat distribution in Ivanov et al., 2013). The second reporting aggregate considered is the Administrative Accounting Units (AAU), based purely on administrative boundaries (e.g., County or State boundaries). The AAU can help understand the links between ecosystem assets and economic and other development activities. The EAU and AAU are likely to be described in terms of 100s km^2 or greater.

Overall, whilst the scale at which biodiversity information is collected will depend on several factors, including context, policy settings and the priority species chosen, further testing and experimentation is required in order to understand the scales most useful for informing biodiversity accounts. This should include the testing of upscaling, downscaling and / or transfer approaches to assign biodiversity information to Ecosystem Units. The Biodiversity

Accounts and case studies presented here may be implemented at EU / LCEU and EAU / AAU scales, according to the availability and resolution of data available.

5.2.2 Data Sources and Limitations

National scale habitat maps may also provide a useful starting point for informing EU boundaries that are most relevant to biodiversity. There are also a number of regional and international land cover classification systems (e.g., Corine in Europe) that could provide the basis for land accounts. These are discussed in Section 2 and SEEA-EEA Tech. Guide 2 (2015). However, as identified in SEEA-EEA Tech. Guide 2 (2015) linking different land cover classifications to the SEEA Central Framework will require translating land cover nomenclature to be consistent with the SEEA land cover classifications. Any such translation needs to be fully documented in order to support the potential for wider comparisons. Ideally, the GIS data on land cover or habitat area should be from a single source, this will avoid inconsistencies in the manner in which areas are delineated.

As noted in SEEA-EEA Tech. Guide 4 (2015), the approach for rivers (as linear features) requires development. The South African approach to river ecosystem accounting was to adopt sub-quaternary catchment areas as ecosystem units. However, this will confound in-stream river habitat with terrestrial riparian habitat. The ENCA-QSP (Jean-Louis Weber, 2014) adopts an alternative approach, where River System Units (RSUs) are specified in linear terms on the basis of homogeneous characteristics (e.g., stretches of canals, river reaches of same Stahler order, etc.). The SEEA-EEA Tech. Guide 4 (2015) states that this approach has been developed in some countries (e.g., Australia) and would benefit from further testing elsewhere.

Whilst the SEEA-EEA identifies that the delineation of marine areas on the basis of area and ecosystem characteristics is an important issue, it provides no specific guidance in this regard. The approach adopted in the ENCA-QSP (Jean-Louis Weber, 2014) is to establish a set of Marine Coastal Units (MCUs) based on bathymetry, legal boundaries and sea habitat mapping. In this regard, the Pegaso Project proposes a set of marine ecosystem units for the Cabrera Archipelago, Spain, based on benthic habitat mapping (Breton et al., 2014).

Given the possible the limitations on the scale at which biodiversity information can be collected, it may be necessary to employ downscaling approaches in order to assign biodiversity data to the EU scale. Further testing of these approaches is required but could be based on species ranges, expectations of abundance distributions or habitat suitability modelling.

It is noted that diversity amongst ecosystems reflects an aspect of biodiversity captured in the CBD definition. Accordingly, once Extent Accounts for different ecosystem units have been established this can be viewed as a Biodiversity Account in its own right, albeit with no direct data on species diversity. Resource accounts that record the ‘stock’ of species within EUs are required to organise data on species diversity.

5.3 Producing Biodiversity Accounts and core tables

To date ecosystem accounting has largely related to terrestrial areas. As such, the Biodiversity Accounts discussed in this technical guidance document focus upon terrestrial accounting units. However, the general accounting structures proposed are believed to be equally applicable to marine areas employing a system of MCUs. Where river ecosystem accounting is of interest, the accounting tables discussed will also serve well for collecting biodiversity information on RSUs. This implies that accounts of biodiversity may need to be compiled individually for terrestrial and marine domains, and possibly rivers as well (as per Jean-Louis Weber, 2014). This reflects the differences in accounting units across these domains. Whilst there are potential links between terrestrial accounts of biodiversity and economic activity via land accounts, linking RSUs and MCUs to the economy is an area for testing and development.

Once a spatially explicit arrangement of ecosystem units has been established, appropriate measures of biodiversity must be selected to inform Biodiversity Accounts for each unit. Jones & Soloman (2013) discuss the issue of what data to use in the Biodiversity Accounts, including issues associated with data availability or associated collection effort. Jones (1996) presents a six-tiered pyramid of approaches for inventorying biodiversity and measuring changes in its stock. Measuring biodiversity to the detail required in the highest level (Level 6 – an inventory of flora and fauna by population) is very resource intensive. Most countries will not have this level of data at the spatial scale required for full national Biodiversity Accounts. A three tiered approach, therefore, is presented in **Figure 8**. This provides a pragmatic approach with 'Tier 1', 'Tier 2', and 'Tier 3', options based on data availability.

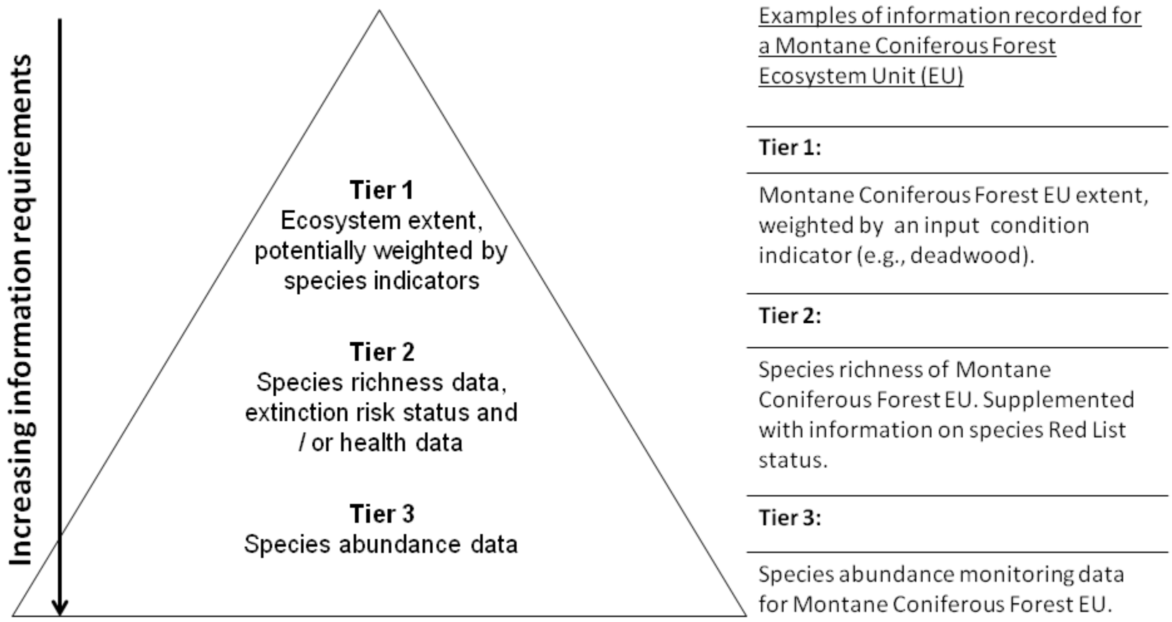

Figure 8. Pyramid illustrating the three levels of detail considered in the different biodiversity accounting contexts discussed. The information collected becomes more detailed and requires more resources to compile towards the base.

It should be noted the accounting structures presented below are intended for guidance and as a basis for experimentation. They are not necessarily prescriptive of a recommended course of action and still need to undergo testing, refining, and validation.

5.3.1 'Tier 1': Ecosystem extent accounting

The SEEA-EEA identifies that the construction of accounts of land cover to provide information on ecosystem diversity across the landscape and, therefore, comprise a biodiversity account in their own right. Consequently, identifying and mapping areas of habitat of high biodiversity value and collating land-based accounts of these areas can provide useful statistics for minimising the impacts of economic activity and development on biodiversity. This approach has recently been employed by Statistics Sweden, as summarised in **Box 10**.

Box 10: Land Accounts for Biodiversity (Statistics Sweden, 2015)

The Swedish Ministry of the Environment commissioned an experimental statistical framework in order to identify and link habitats of high biodiversity value to the economy via land ownership. The scope of the land accounts was limited to habitats reported in accordance with Article 17 of the European Union Habitats Directive and key biotopes. These comprised: Wetlands, Western taiga, key forest biotopes, and meadows and grazing land. These areas of high biodiversity value were linked to ownership entities via the Register of Real Estate Assessment. In some cases it was possible to break down the biodiversity value in habitat classes further, for example wetlands were assigned to high and low natural value classes. The project demonstrates a useful experimental approach that benefits from the use of proxy data to construct accounts for biodiversity (ecosystem diversity) at an aggregate scale.

Whilst the approach presented in Box 10 provides a useful starting point for understanding trends in ecosystem diversity, Statistics Sweden (2015) identified that it would benefit from inclusion of better 'proxy' data for biodiversity. This allows distinctions to be made between near pristine and degraded ecosystems and habitat. The SEEA-EEA Tech. Guide 1 (2015) provides the potentially useful approach of defining Ecosystem Units are described on the basis of the plant community associations. Delineating EUs in this manner, therefore, clearly captures an account of plant and ecosystem diversity across the landscape and is likely to provide a good proxy for species diversity generally.

In related approaches, the Norwegian Nature Index (Certain et al., 2011; Section 3.4.1), Natural Capital Index (NCI) in Holland (ten Brink & Tekelenburg, 2002) and the Natural Capital Asset Index (NCAI) in Scotland (Albon et al., 2014) provide methodologies for organising information on biodiversity. These methodologies are based upon, *inter alia*, indicators of species diversity and their interaction with ecosystem extent measures. It is important to note the distinction here between these *input* indicators, which are derived outside of these methodologies, and the *output* indicators that these methodologies generate. The input indicators are used to weight ecosystem areas in terms of biodiversity quality. **Box 11** presents the NCI approach, where the NCI is calculated for given ecosystems as a product of their extent and condition, where condition is determined as a function of species abundance relative to a reference condition.

Box 11: Natural Capital Index for the Netherlands (ten Brink & Tekelenburg, 2002).

The Natural Capital Index (NCI) was developed in order to provide an indicator to communicate trends in habitat and biodiversity loss in the Netherlands. The approach is based on calculating:

i. extent of ecosystem remaining - as a percentage of a given habitat classification compared to a reference scenario, and
ii. ecosystem condition - as a percentage of species abundance diversity in each habitat compared to a reference condition.

The product of these two measures provides the NCI for that habitat (see figure below). Aggregation over all habitat types provides the NCI for the Netherlands as a whole. The NCI favours species abundance over richness as a more sensitive measure of biodiversity loss. Calculating the NCI on a regular basis (ideally annually) allows opening and closing levels of the NCI (ecosystem and species diversity stocks) to be recorded. This is similar to the Natural Capital Asset Index and Norwegian Nature Index approaches.

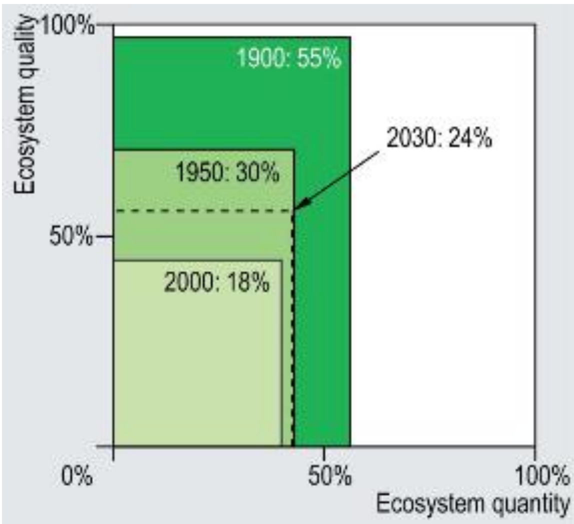

Figure. Aggregated Natural Capital Index showing changes in ecosystem quantity (extent) and quality (condition) between 1900 and 2000.

Under the NCI, NNI and NCAI approaches, the availability of appropriate reference conditions for the input indicators chosen is essential to establish the relative measure of biodiversity condition. The SEEA-EEA suggests the reference condition should comprise reference to a state of minimal human disturbance. In this regard, the ENCA-QSP (Jean-Louis Weber, 2014) suggests identifying an area for the relevant land class in near pristine or very good condition (e.g., a protected area, nature reserve, etc.) and capture a species diversity measure for use as a reference condition. However, in many cases (e.g., much of Europe) it may be impossible to estimate reference conditions due to the long history of human development. In these scenarios, the ENCA-QSP (Jean-Louis Weber, 2014) suggests a socially aspirational target could be adopted, possibly based on specific policy drivers. Alternatively, a reference condition can be established on the principles of accrual accounting, where the reference condition reflects the opening level of the first account compiled. This is similar to the LPI approach (see Section 3.4.2) of setting the reference level to a known base year (in this case 1970). It should be noted that reference condition levels for the same indicator are likely to change across different types of ecosystem units. Furthermore, the selection of a reference condition needs to be considered in light of the specific ecosystem services that an Ecosystem Unit provides. For instance, a pristine biological assemblage may imply the loss of several economically and culturally important ecosystem services.

Once a reference condition has been determined, relative measures for input indicators (X') can be generated using normalisation approaches. For example, normalisation approach based on the distance to the reference condition and centred on 1, of the form:

$$X' = \frac{X}{XRef}$$

Where: X is value of indicator, XRef is the reference value for the indicator and X' is the value of X relative to XRef (i.e., distance from XRef, where 1 implies equality, <1 implies below reference and >1 above reference). A useful summary of this and other normalization approaches is provided in Nardo et al. (2005). Whilst the approach is described for indicator, relative measures of species richness and abundance can be derived in exactly the same manner.

It should be noted the normalisation approach above assumes negative effects are encountered below the reference level and no negative effects occur above it. Furthermore, a value > 1 implies a positive outcome. Certain et al. (2011) describe a set of more additional scaling models. These comprise optimal (implying there are negative effects associated with exceeding or being below the reference level), minimal (where negative effects occur below the reference level only) and maximal (negative effects observed above the reference level only). Certain et al. (2011) also describe how these models can be developed to cater for uncertainty in indicator values via the use of simulation techniques (Monte Carlo analysis).

The NCI, NNI and NCAI approaches to weighting ecosystems using indicators and aggregating this information provides a more informative account of biodiversity than solely accounting for the extent of different ecosystems. **Table 4** provides an example of what an accounting table employing this approach could look like. Additional information on habitat type for Ecosystem Units can be obtained by specifying their habitat sub-classes to provide more detailed accounts of habitat diversity, as discussed in Section 5.2. The set of input indicators selected should also reflect both conservation policy priorities for biodiversity and the role of species in ecosystem condition / functioning. The input indicators included should, ideally, be relevant to the ecosystem unit or EAU in question. For example, in the UK the woodland bird index may be selected as an indicator of animal species diversity for tree covered areas. The 'Tier 1' biodiversity accounting approach has clear parallels with accounting for ecosystem condition (discussed in the following section), in terms of capturing summary information on the characteristics of ecosystems.

In some circumstances a single relevant input indicator may also be sufficient as the overall indicator to communicate the condition of biodiversity in an ecosystem (i.e., an output indicator). Indeed it may be that only a single indicator is available for certain ecosystems (e.g., an indicator for a keystone species). However, in many circumstances a headline composite indicator (HI) is likely to provide a better output indicator for the overall condition of biodiversity within the ecosystem unit. This will better characterise the range of policy priorities for biodiversity, against which progress is being assessed. This type of indicator can be generated by specifying weights (in the range 0 - 1) for each input indicator in the in Table 4 (i.e., w, x and y) on the basis of their importance to policy priorities and / or ecosystem functioning. These weights should sum to 1 (i.e., $w + x + y = 1$). The headline composite indicator may be transferred directly to the Ecosystem Condition Account, providing it reflects a distribution of species expected in a well-functioning ecosystem for the class of the Ecosystem Unit in question.

The selection of weights could either reflect the distribution of species in the ecosystem unit under the reference condition or the relative importance of each species group to the policy question(s) the account is intended to answer. This will require expert ecological judgement to complete. The headline composite indicator (HI) level for each ecosystem unit is then obtained by summing the product of each species indicator and its corresponding weight (i.e., $HI = w*SI1 + x*SI2 + y*SI3$ in Table 4). It is important this procedure is only applied to the normalised measures as they are dimensionless (or scaled).

The weighting of ecosystem extents has the advantage of allowing a ‘common currency’ (biodiversity weighted hectares) to be derived across different types of ecosystem units. This can aid aggregation. The final column in Table 4 illustrates how this measure is obtained.

Table 4. An example 'Tier 1' weighted Ecosystem Extent Account for an Ecosystem Unit (EU), which organises information on diversity between ecosystems and indicators of species diversity within these ecosystems.

EU01 - HABITAT CLASS (e.g., Plains Woodland)*							
	Habitat Area*	Species Indicator 1 (SI1) (Input Indicator)	Species Indicator 2 (SI2) (Input Indicator)	Species Indicator 3 (SI3) (Input Indicator)	Species Indicator... (Input Indicator)	Headline Species Indicator (HI) (Output Indicator)	Stock of Biodiversity (HI * habitat area)
	indicator weight	w	x	y	...	HI = w*SI1 + x*SI2 + y*SI3 ...	
Open							
Additions	*						
Reductions	*						
Close							
Net Change							
Reference							N/A

*Habitat type and could be recovered from information on EUs and habitat sub-classes, as discussed in Section 5.2. Information on changes on habitat area could be recovered from the EU extent account.

Box 12 provides a summary of possible methods for obtaining input indicators for species diversity.

Box 12: Obtaining data for weighted 'Tier 1' Ecosystem Extent Accounting

- The 'Tier 1' Ecosystem Extent Accounts comprise measures of ecosystem extent. This can be supported with information on habitat type and the use of indicators of biodiversity to weight habitat areas.

- The general approach described has been employed by Albon et al., (2014) for producing a Natural Capital Asset Index. They made use of existing indicators of biodiversity, *inter alia*, that were readily available and relevant to different ecosystem services. These include: woodland bird index, tree health and species richness of broadleaf and conifer forests. Similar indicators may already exist in a number of countries.

- Indicators could also be developed on the basis of expert judgement for based on a set of key drivers of biodiversity loss.

Where there is a paucity of data on biodiversity and / or ecosystem classification, a pragmatic approach may be to move directly to the EAU or AAU level. **Box 13** provides an imaginary case study for what such an account would look like at the EAU scale. As a minimum some broad habitat mapping will be required in order to populate such an account.

Generating a 'proxy' stock for biodiversity in terms of biodiversity weighted habitat, allows direct aggregation across EUs and EAUs. This data can be aggregated in total or across different ecosystem units or habitat types. 'Tier 1' weighted Ecosystem Extent Accounts can then be aggregated to the national scale in order to allow progress towards national biodiversity policies to be reported. Providing the underlying suite of indicators and methodology for generating reference conditions remains broadly consistent, 'proxy' stocks can be compared with, and aggregated across, countries. Dividing the total 'proxy' stock for the EAU by the total area of the EAU would generate an average index value in the same fashion as the Norwegian Nature Index (Certain et al., 2011).

The weighting procedure and systematic averaging across ecosystem units is useful for communicating biodiversity trends at large-scales. However, this will obscure the spatially explicit nature of data derived for ecosystem units. As such, the approach may provide limited information on trends in ecosystem services at these scales, which is a key motivation of ecosystem accounting generally.

Box 13: 'Tier 1' weighted Ecosystem Extent Account for an imaginary case study based on an Ecosystem Accounting Unit (e.g., watershed)

The example weighted Ecosystem Extent Account displayed in the table below is for an Ecosystem Accounting Unit (EAU) containing three different ecosystem or habitat types (woodland, moorland and grassland). In addition to this information on ecosystem diversity, information on species diversity is captured for each ecosystem via a range of species diversity indicators. The reference column is included for illustration and would not be required in any final accounting table. The indicators presented are expressed as a proportion of a reference condition for each habitat. Some indicators apply to all habitat types (e.g., species richness) and others are only relevant to specific habitat types (e.g., tree health applies to woodland only). Weights for the different indicators can be specified for each habitat type to express their relative importance (e.g., with respect to policy priorities or ecosystem functioning) to inform the headline indicator. A proxy for the stock of biodiversity is calculated in the final column in a similar fashion to the Natural Capital Index for The Netherlands (i.e. as the area of habitat multiplied by the headline indicator for that habitat). The account reveals that the stock of biodiversity has reduced within this EAU. The account also reveals that, whilst increasing in habitat area, the stock of woodland biodiversity has decreased.

	Open (2000)	Close (2010)	Open (2000)	Close (2010)	Ref (1950-60)	Open (2000)	Close (2010)	Ref (1950-60)	Open (2000)	Close (2010)	Ref (1950-60)
Indicators:	Habitat Area		Bird Populations			Species richness			Tree Health		
Woodland	3.7	3.78	0.9	0.97	1	1	0.85	1	0.99	0.88	1
Indicator Weights				0.19			0.41			0.25	
Grassland	0.7	0.69	0.57	0.57	1	0.8	0.6	1	Not applicable	Not applicable	Not applicable
Indicator Weights				0.21			0.48			0	
Moorland	0.66	0.63	1	0.92	1	0.95	0.88	1	Not applicable	Not applicable	Not applicable
Indicator Weights				0.31			0.31			0	

………

	Open (2000)	Close (2010)	Ref (1950-60)	Open (2000)	Close (2010)	Ref (1950-60)	Open (2000)	Close (2010)	Ref (1950-60)	Open (2000)	Close (2010)	Net Change
	Butterfly population			Bog grass:forb ratio (inverse)			Headline Species Indicator			Stock of Biodiversity		
Woodland	1	0.71	1	Not applicable	Not applicable	Not applicable	0.328	0.277	1	1.212	1.046	-0.165
Indicator Weights		0.08			0							
Grassland	1	0.78	1	Not applicable	Not applicable	Not applicable	0.31	0.242	1	0.217	0.167	-0.050
Indicator Weights		0.31			0							
Moorland	1	0.92	1	1	0.71	1	0.17	0.156	1	0.112	0.099	-0.014
Indicator Weights		0.17			0.21							
									TOTAL:	0.514	0.437	-0.076

…continued…….

5.3.2 'Tier 2': Species richness accounting

The 'Tier 2' approach to biodiversity accounting is based on species richness data, species extinction risk data and other information on their status. Broad approaches to measuring species richness are discussed in Sections 2, 3 and 4. Ideally, data on species richness should be representative of species across all taxonomic groups, particularly if biodiversity in relation to ecosystem functioning is the principle concern. Table 5, presents an example of what a 'Tier 2' species richness account would look like for collating information on different taxonomic groups. Table 5 follows the general form of asset accounts in the SEEA Central Framework (2014), with opening, closing and net changes in biodiversity 'stocks' over the accounting period. Additional rows are included to capture the increases and decreases in species abundance due to natural or human processes over the accounting period.

Table 5. Extension of SEEA-EEA Table 4.7 (2014) 'Tier 2' Species Richness Account proposed for an Ecosystem Unit or Ecosystem Accounting Unit. This example provides a framework for organising information on absolute measures of species richness (relative measures could also be used), with opening and closing stocks for a range of taxonomic groups. The headline indicator captures an overall measure of the condition of the ecosystems biodiversity (i.e., an output indicator).

EU01 - HABITAT CLASS (e.g., Montane Coniferous Forest / EAU01 - EAU TYPE (e.g., watershed)							
	Animals					Plants	Headline Indicator (Output Indicator)
	Mammals	Birds	Reptiles	Fish	Invertebrates (e.g., butterflies bees)		
Absolute Measures							
Opening population							
Additions							
Immigration							
Managed Reintroduction							
Reductions							
Local extinctions							
Closing population							
Net Change							

As identified in SEEA-EEA, across larger areas (e.g., national scale) measures of aggregated species richness may show little change on a temporal basis. Consequently, it may be useful to include additional information on the species extinction risk as additional columns in the 'Tier 2' Species Richness Accounts. This approach to species accounting has gained some traction in applied settings, albeit at sub-national or EAU / AAU levels, and is reviewed in Box 13.

5.3.3 'Tier 2': Species extinction risk

'Tier 2' Species Richness Accounts capture information on species occurrence. However, it may be useful to supplement this with information on species extinction risk. Table A4.1 of the SEEA-EEA suggests a supplementary account for threatened species for ''Tier 2' biodiversity accounting, based on The IUCN Red List of Threatened Species Categories (see **Table 6**). The accounts for threatened species can be constructed for particular EUs, EAUs / AAUs, or for a country as a whole. However, it is important to note that the information on extinction risk is relevant over the scale at which it is compiled (e.g., a global or national Red List) not the scale it is disaggregated to (e.g., an Ecosystem Unit). Nonetheless, there are likely to be many contexts in which such an account could prove informative in assessing progress towards policy goals or identifying where additional resources need to be directed to combat biodiversity loss and safeguard important species. Furthermore, the presence of threatened species within an Ecosystem Unit is likely to represent the presence of ecosystem specialists and likely indicative of favourable ecosystem condition. As an example, the account presented in **Box 14** collates information on the status of species within a National Resource Management Region in Australia, equivalent to the EAU / AAU scale.

Table 6. SEEA-EEA Table A4.2.1 (2014) 'Tier 2' account of biodiversity in terms of threatened species for Ecosystem Unit or Ecosystem Accounting Unit. This account shows how information on species conservation status could be organised, comprising opening and closing stocks of species assessed against The IUCN Red List of Threatened Species Categories.

	IUCN Red List categories								
	Extinct	Extinct in the wild	Critically endangered	Endangered	Vulnerable	Near threatened	Least concern	Data deficient or not evaluated	Total
Opening stock									
Additions									
From lower threat categories									
From higher threat categories									
Discoveries of new species									
Rediscoveries of extinct species									
Reclassifications									
Updated assessments									
New additions to list									
Total additions									
Reductions									
To lower threat categories									
To higher threat categories									
Reclassifications									
Local extinction									
Updated assessments									
Total reductions									
Closing stock									

The entire Red List Index methodology (Butchart et al., 2004; Butchart et al., 2007) depends on coding reasons for change, and purging changes resulting from changing knowledge. This provides information on genuine changes in extinction risk for species. As such national and sub-national Red Listing provides a reliable method that enables the conservation assessment of species across these respective scales. As noted in Section 3.3.1, 113 countries, and 45 sub-national entities that have already developed Red Lists. Establishing a reasonable timescale for re-assessment would provide the necessary data to populate accounting tables of the format of Table A4.2.1 in SEEA-EEA (see Table 6). The National Red List website provides further guidance in this regard (http://www.nationalredlist.org).

The Australian Bureau of Statistics has developed Biodiversity Accounts that include inventories of species and their extinction risk for the Great Barrier Reef terrestrial areas, which align with the SEEA framework. **Box 14** provides an example of the experimental account compiled for one of the Great Barrier Reef regions – the Burdekin National Resource Management Region (NRM) – as a terrestrial EAU.

Box 14: A Species Account for Burdekin National Resource Management Region (NRM) in Australia (Bond et al., 2013)

The table below presents a Species Account based on both species richness and extinction risk for the Burdekin NRM in the year 2000.

	Introduced species	Native species				
		Unprotected	Protected	Rare and endangered	*Total native species*	Total species
Animals						
Vertebrates						
-Mammals	15	2	112	20	114	129
-Birds	10	0	458	33	458	468
-Reptiles	2	0	202	26	202	204
-Amphibians	1	0	51	9	51	52
-Bony fish	4	56	0	0	56	60
-Cartilaginous fish	NA	NA	NA	NA	NA	NA
-Insects	0	11	2	0	13	13
Subtotal	*32*	*69*	*825*	*88*	*894*	*926*
Plants	376	5	3239	91	3244	6320
Subtotal	*376*	*5*	*3239*	*91*	*3244*	*6320*
Fungi	0	0	68	0	68	68
Subtotal	*0*	*0*	*68*	*0*	*68*	*68*
Protista	0	0	148	0	148	148
Subtotal	*0*	*0*	*148*	*0*	*148*	*148*
TOTAL	**408**	**74**	**4280**	**179**	**4354**	**4762**

The second table below also provides a biodiversity asset account for the Burdekin NRM. The large gap between the opening and closing years for the table was chosen on the basis of data availability. Given data exists for two time periods it would be possible to establish a trend line for intervening and future years. The accuracy of trend forecasting would be greatly improved by additional data points. Whilst the table contains the characteristic asset table rows for additions and reductions in stock levels, these are not populated. Columns on the status of various species are also included.

Burdekin NRM	Introduced	Native species				Total species
		Unprotected	Protected	Rare and endangered	*Total native species*	
Opening stock 2000	408	74	4280	179	4354	4762
Additions						
-from lower threat categories (ie increased risk of extinction)	NA	NA	NA	NA	NA	NA
-from higher threat categories (ie reduced risk of exinction)	NA	NA	NA	NA	NA	NA
-discoveries of new species	NA	NA	NA	NA	NA	NA
-rediscoveries of extinct species	NA	NA	NA	NA	NA	NA
-reclassifications(a)	NA	NA	NA	NA	NA	NA
Total additions	NA	NA	NA	NA	NA	NA
Reductions						
-to lower threat categories (ie reduced risk of extinction)	NA	NA	NA	NA	NA	NA
-to higher threat categories (ie increased risk of exinction)	NA	NA	NA	NA	NA	NA
-reclassifications(b)	NA	NA	NA	NA	NA	NA
Total reductions	NA	NA	NA	NA	NA	NA
Net change	*121*	*34*	*353*	*30*	*387*	*508*
Closing stock 2011	529	108	4633	209	4741	5270

(a) Where one existing species is now recognised as two or more distinct species.

(b) Where two or more existing species are now recognised as one species.

The Burdekin NRM account has the typical open and closing stocks for an asset account, although the accounting period is 11 years. As aforementioned, yearly estimates for species counts can be interpolated from establishing a trend line between the accounting periods. Species counts for future years could also be extrapolated using this trend line. However, species counts estimated from interpolation of trend lines will be subject to some error. This could be severe where there is a paucity of time series data, as in the Burdekin example with only two time series observations. Nonetheless, this is likely to be a common issue in establishing any pilot accounting for biodiversity and illustrates that the accounting process is iterative, with accuracy and detail increasing over accounting periods.

Box 15 provides a summary of possible methods for obtaining data on species richness and its extinction risk for biodiversity accounting.

Box 15: Obtaining data for 'Tier 2': Species Richness Accounts

- Set-up or use established monitoring programmes.
- Box 8 in Section 4 lists other sources of species monitoring data that may be available.
- Existing data on species ranges (e.g., from IUCN Red List of Threatened Species data) could be employed to establish a baseline. These could be updated to generate time series data.
- Employ modelling or expert judgement approaches upscale existing point monitoring data

to estimates species occurrence / distribution for key species.

- Where accounts are constructed at the EU scale, it may also be possible to estimate species abundance using a species-area curve (Brooks et al., 2002).

5.3.4 'Tier 2': Biodiversity health accounting

In addition (or as an alternative) a supplementary table on species health may be useful in order to capture information on disease prevalence, toxins in tissues etc. For example, such information would be useful for EUs that provide agricultural-related ecosystem services, such as dry land pastures. **Table 7** provides an example of what such an account could look like.

Table 7. 'Tier 2' biodiversity health account for an Ecosystem Unit or Ecosystem Accounting Unit. This account shows how information on species health can be organised, including opening and closing levels of disease and toxin prevalence in animals and other health measures.

EU01 / EAU01				
	Animals health measures		Microorganism health measures	Plant health measures
	Disease	Toxins		
Opening level				
Additions				
Reductions				
Closing level				
Net change				
Reference level				

5.3.5 'Tier 3': Species abundance accounting

At its most comprehensive, an account of species abundance implies a count of all individuals for each species present in an ecosystem unit to be captured in the accounting table. Such data is likely to be limited to only high priority species. For instance, species of particular cultural, economic or conservation significance (e.g. pandas in China, elephants in Africa, and whales in the oceans) and to areas with large investments in environmental monitoring (e.g. protected areas). For example, an account of this nature may be useful for organising information on large mammals in Kruger National Park due to the economic importance of ecotourism.

Table 8 presents an example of 'Tier 3' Species Abundance Account. Again, Table 8 follows the general asset accounts form, with opening, closing and net changes in biodiversity 'stocks' over the accounting period. Additional rows are included to capture the increases and decreases in species abundance. However, in the particular context of species abundance data, it may be pragmatic to focus purely on opening and closing stocks in the pilot phase. This is because populating these additional rows will require additional data collection and may often be difficult to complete in a balanced manner. A reference level for biodiversity is included in Table 8 in order to provide a benchmark for comparison.

Table 8. Example of a 'Tier 3' Species Abundance Account proposed for an Ecosystem Unit or Ecosystem Accounting Unit based on SEEA-EEA Table 4.7 (2014). This example provides a framework for organising information on species abundance of selected priority species (this will change with context). Opening and closing stocks for individuals and a species diversity index (output indicator) generated from this information are included.

EU01 - HABITAT CLASS (e.g., Plains Grassy Woodland)) / EAU01 - EAU TYPE (e.g., Watershed)				
	Elephants	Rhinos	Lions	Species Diversity Index (Output Indicator)
Absolute measures				
Opening Species Abundance				
Additions				
Natural offspring				
Immigration				
Managed re-introductions				
Reductions				
Death rate				
Emigration				

Closing Species Abundance				
Net Change				
Reference Species Abundance				

A species diversity index (an output indicator of biodiversity) can be generated using the data recorded on individual population levels in Table 8 (as per the final column). A number of methods for deriving such indices based on discrete population counts are discussed in Section 2, for example, the Shannon Index or the Simpson Index (see Sherwin et al., 2006) or Community Specialisation Index (Clavel et al., 2011). Such an output indicator of species diversity can be used in the Ecosystem Condition Account to quantify biodiversity characteristics for Ecosystem Units. However, where data on species populations is limited to species of particular cultural, conservation or economic significance (e.g., as in Table 8), any such index will suffer bias due to the omission of species measures from other taxonomic groups. Indicators generated in these circumstances are unlikely to provide indicators of biodiversity that would reflect trends in ecosystem functioning (e.g., nutrient cycling). Consequently they may be of limited use as measures of ecosystem condition. One way to negotiate this issue would be to focus on keystone, umbrella or specialist species (as discussed in Section 2) or ensure a comprehensive range of taxonomic groups are considered.

Table 9 has been populated in terms of specific species in absolute terms. However, the structure is equally suitable for capturing information on the abundance of individuals in different taxonomic groups. For example, mammals, birds, insects etc. It may also be more practical to generate measures or estimates of species abundance in relative terms (e.g., as per the Norwegian Nature Index and discussed in relation to indicators in 'Tier 1' accounting). **Table 9** presents an example of a 'Tier 3' biodiversity accounting table in this format. An account of this type would be useful for organising information on species abundance for a watershed (EAU) in a country, for example, the Thames Basin in the UK. Whilst not indicated in Table 9, rows for additions and reductions in stocks could also be included in the table.

Table 9. Extension of SEEA-EEA Table 4.7 (2014) 'Tier 3' Species Abundance Account of relative species abundance proposed for an Ecosystem Unit or Ecosystem Accounting Unit. This example provides a framework for organising information on relative species abundance, with opening and closing stocks for a range of taxonomic groups. A headline indicator to capture an overall measure of the condition of the ecosystems biodiversity (Output indicator).

EU01 - HABITAT CLASS (e.g., Montane Coniferous Foest / EAU01 - EAU TYPE (e.g., watershed)							
	Animals					Plants	Headline Indicator (Output Indicator)
	Mammals	Birds	Reptiles	Fish	Invertebrates (e.g., butterflies bees)		
Relative Measures							

Opening population as a proportion of reference population							
Closing population as a proportion of reference population							
Net Change							

A headline indicator (HI) (output indicator) can be generated by specifying (in the range 0 - 1) for each relative species abundance measure in Table 9 on the basis of their importance to ecosystem functioning (or other policy priorities). These weights should sum to 1 and should be determined on the basis of expert ecological knowledge. The HI value for each ecosystem unit is then obtained by summing the product of relative species abundance measure and its corresponding weight (in the same fashion as described for indicators in the 'Tier 1' accounts). It is important this process is only applied to relative measures as they have been normalised, thus rendering them suitable for aggregation within ecosystems.

It should be noted that more than one HI may be required to summarise information in the Biodiversity Accounts. For instance, one indicator may be required to inform progress towards policy goals and another to characterise biodiversity for the Ecosystem Condition Account. Deriving separate indicators for different purposes is dependent on the weighting system employed. To generate a HI for the Ecosystem Condition Account it will be important to capture a distribution of species indicative of a healthy well-functioning ecosystem.

Ideally, species abundance should be compiled in quantitative terms. However, where data limitations apply, species abundance could be specified in qualitative terms. For example, the SEEA-EEA also suggests 'very abundant', 'abundant', 'common', 'rare' and 'very rare' as broad classes for species abundance.

The SEEA-EEA clearly states the accounting period is one year. This may be difficult to achieve in some circumstances due to the intensive resources needed. For example, butterfly population monitoring data may be available on an annual basis. However, other groups of species such as higher plants and bryophytes may not have annual population counts. Where it is unlikely to be possible to capture data on a yearly basis, trend lines could be used to inform accounts for intervening periods. In addition to this, clear protocols need to be established for designating the time of year that different populations are measured. For example, the appropriate time in the year for measuring butterfly populations is likely to be different to that for migratory birds.

Box 16 provides a summary of possible methods for obtaining data on species abundance for biodiversity accounting.

Box 16: Obtaining data for species abundance accounting

- The ideal is to use species monitoring programmes to generate abundance data for priority species at the EU or (more realistically) EAU, AAU or national scale. This will allow the account to be populated in absolute terms. This may also provide the basis for transferring estimates of species abundance to similar EUs within the country.

- **Box 8** in Section 4 lists other sources of species monitoring data that may be available.

- The conceptual approach for the Living Planet Index lends itself to a relative species abundance accounting framework. In many countries there will be established monitoring programmes that can be drawn on for collecting abundance data. There may also data already within the global LPI database that is relevant. A headline index for biodiversity can be established that could provide an indicator of biodiversity relevant to ecosystem condition. The LPI has been derived at the national scale (e.g., Canada and Uganda) and guidance exists for developing this at national and sub-national scales (see McRae et al., 2008). Consequently, it may be possible to generate Biodiversity Accounts using this approach at the AAU scale.

- Modelling approaches based on ecological niche, key drivers of biodiversity loss or simulation approaches could be employed to estimate relative measures of abundance for some taxonomic groups based on upscaling existing or new data from monitoring programmes (see Section 4) to provide abundance estimates for EUs.

- An alternative approach is to generate relative estimates of species abundance for EUs on the basis of expert judgement. This is the approach proposed by Scholes & Biggs (2005) for the development of the Biodiversity Intactness Index (BII). Under this approach, ecological specialists are asked to estimate reductions in populations of broad taxonomic groups for given ecosystems due to land use activities (or other characteristics of Ecosystem Units). These estimates were referenced relative to a large protected area of that land cover class. This approach is presented for Southern Africa in Scholes & Biggs (2005) and may provide a useful starting point for piloting accounts of species abundance.

5.3.6 Aggregation of Species Accounts

Biodiversity data collected for EUs will be useful for management at the local scale, for instance in the case of nature reserves and national parks. However, aggregating biodiversity information to the scale of Ecosystem Accounting Units (EAU) or Administrative Accounting Units (AAU) will be required to provide statistics that reveal the trends in stocks of biodiversity at a scale that can be assessed against economic and social statistics. The nested arrangement of reporting units is summarised in **Figure 9**, using EAUs as an example.

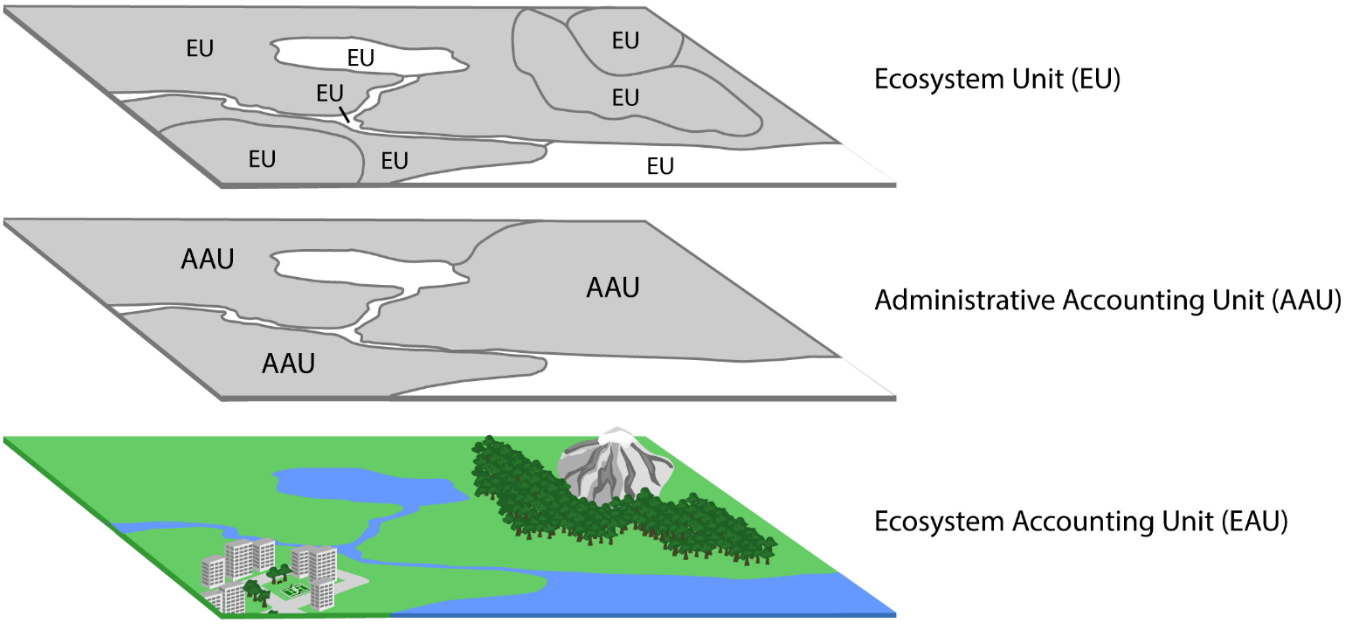

Figure 9. The arrangement of ecosystem accounting and reporting units. Ecosystem Units (EUs) relate to specific areas of ecosystems defined by their Extent Account. These can be aggregated to larger scales for reporting purposes. For instance across administrative areas based on state or county boundaries that define Administrative Accounting Units (AAUs). Alternatively, they may be aggregated on the basis of ecological boundaries, such as a watershed, which define Ecosystem Accounting Units (EAUs).

For accounts collating statistics on absolute measures of species abundance (e.g., number of Elephants as per Table 8), the aggregation procedure is relatively straightforward. All individuals in all EUs within an EAU or AAU can be summed together to obtain a measure of the number of individuals within the EAU. For example, consider the number of a particular bird species (say Grouse) in a watershed area (EAU), within which EUs have been coarsely defined in terms of Land Cover Functional Ecosystem Units (LCEUs). The watershed EAU contains 1 lake (inland water body LCEU), 2 areas of natural forest (tree covered area LCEUs) and 1 area of moorland (grassland LCEU). If zero Grouse are in the loch, 50 in the first forest area, 25 in the second forest area and 200 in the moorland area a total of 275 are present in the EAU. The same process is repeated across EAUs to arrive at a national total.

For species richness accounting, the aggregation approach is more complex because there will be a number of common species across different EUs and care will need to be taken to avoid double counting of species when aggregating to the EAU scale. Again, the same process as for abundance data would be repeated across EAUs to arrive at a national total. However, given this complication it may be pragmatic to construct accounts of species richness at different scales directly, based on spatial data on their distributional range / occurrence.

Statistics on biodiversity at the EAU or AAU scale will be particularly useful for identifying broad patterns between economic activity and biodiversity within accounting units, potential, providing insight in to the trade-offs between the two (i.e., is increasing economic activity such as agriculture associated with significant declines in biodiversity).

Where relative measures are aggregated it should be appreciated that these will not comprise additive measures of biodiversity. The approach adopted for the Norwegian Nature Index is to employ area weighting, which is necessary to ensure that a proportionate measure of species diversity is generated for a given land cover classification in the EAU (Certain et al., 2010). Whilst this allows aggregation and communication of biodiversity information at the national level, the approach is not strictly accurate. This is because the fundamental constituents of lower relative measures (i.e., the different species) may not necessarily be represented in higher values of the same measure. Furthermore, averaging this information may obscure the existence of ecosystem assets with high biodiversity stocks within an EAU. Accordingly consideration could be given to presenting more sophisticated descriptive statistics on biodiversity at higher levels of aggregation. For example, the value of relative measures of biodiversity for a series of area percentiles of an EAU (e.g., for 5% of an EAU the relative biodiversity measure is 0.9 or greater). This is an issue for further research and testing.

A further, potential, issue relates to aggregating relative measures different types of ecosystems or settings (e.g., countries) where different reference conditions are employed. Where the reference condition is one of minimal human impact (assessed with confidence) aggregation will not suffer systematically from error. However, if different reference conditions are employed (for example an aspirational reference condition for an urban area and a minimal human disturbance one for an area of forest), the associated relative diversity measures will not be comparable. The SEEA-EEA provides a discussion of these issues in relation to ecosystem condition.

A final but very important consideration with respect to aggregation is to identify how to capture information on diversity of ecosystems across reporting areas, such as EAUs (i.e., gamma biodiversity discussed in Section 2). This is related to issues associated with ecotones (the border areas between different ecosystems), which are often high in biodiversity (as discussed in the ENCA-QSP, Jean-Louis Weber, 2014). Overall, further testing of aggregation approaches on both pragmatic and theoretical terms is required.

As an intermediate step in aggregation, it may be useful to collate all information on similar EUs within an EAU in a single accounting table (e.g., all areas of forest in an AAU, where the AAU comprised the national scale). In addition, where biodiversity is collected at the EU scale, these can also be aggregated across their parent land cover classes in order to establish a potential link to the SEEA Central Framework (2014).

Once tables have been constructed for all EAUs or AAUs, overall trends in the selected biodiversity measures can be determined at the national scale for different ecosystem classes. This is likely to be particularly useful for communicating progress towards national policy targets. If appropriate, EUs or EAUs can also be weighted in terms of the importance of biodiversity contained within these units. This implies some value judgement is applied to biodiversity in certain areas due to the benefits it provides. For instance, biodiversity contained in EUs or EAUs near urban population centres may be receive a higher weighting due to the cultural services it may provide to a large number of near-by beneficiaries.

6. Producing Ecosystem Condition Accounts and Ecosystem Capacity Accounts at different scales

Key points

- The indicator of biodiversity condition obtained from the Biodiversity Account should reflect the range of species necessary for a healthy, well-functioning and resilient ecosystem.
- Key drivers of improvements and reductions in ecosystem condition could be included in a supplementary table.
- Further testing and research is required into methods to aggregate biodiversity condition indicators across Ecosystem Units, particularly with respect to capturing information on the biodiversity between ecosystems (gamma biodiversity) in Ecosystem Accounting Units.
- Several indicators of biodiversity may be necessary to link the Ecosystem Condition Account to the Ecosystem Capacity Account.
- Information in the Biodiversity Account can inform the Ecosystem Capacity Account directly where biodiversity is considered an ecosystem service in its own right.
- Information on the contribution of biodiversity to ecosystem services can be captured for some services using ecosystem production functions.
- Capturing information on the contribution of biodiversity to intermediate services and ecosystem functional redundancy and resilience is challenging due to non-linear and threshold effects. However, given the importance of this aspect of biodiversity to sustaining ecosystem service provision it needs to be addressed in the ecosystem accounting framework. Further research is required in this regard, however, the Ecosystem Capacity Account provides a useful construct in which to capture and analyse this type information.

This section describes how information on all components of biodiversity is captured within the Ecosystem Condition Account and mobilised from the Biodiversity Accounts. It also describes the linkages between biodiversity information in the Biodiversity Accounts and the Ecosystem Condition Accounts and the information on ecosystem services recorded in the Ecosystem Capacity Account.

6.1 Ecosystem stocks and flows

Ecosystem accounting is based on a system of stocks and flows. Stocks are represented as defined spatial areas comprising ecosystem assets (e.g., Ecosystem Units [EUs] or Ecosystem Accounting Units [EAUs]). These ecosystem assets have characteristics that underpin their functioning and, consequently, their ability to provide a flow of ecosystem services. These services not only include the flows to society and the economy but also intra-ecosystem and inter-ecosystem flows, within and between different ecosystem units (e.g., nutrient and energy cycling). These interlinkages are set out in a basic model in Figure 2.1 of the SEEA-EEA, presented in **Figure 11.**

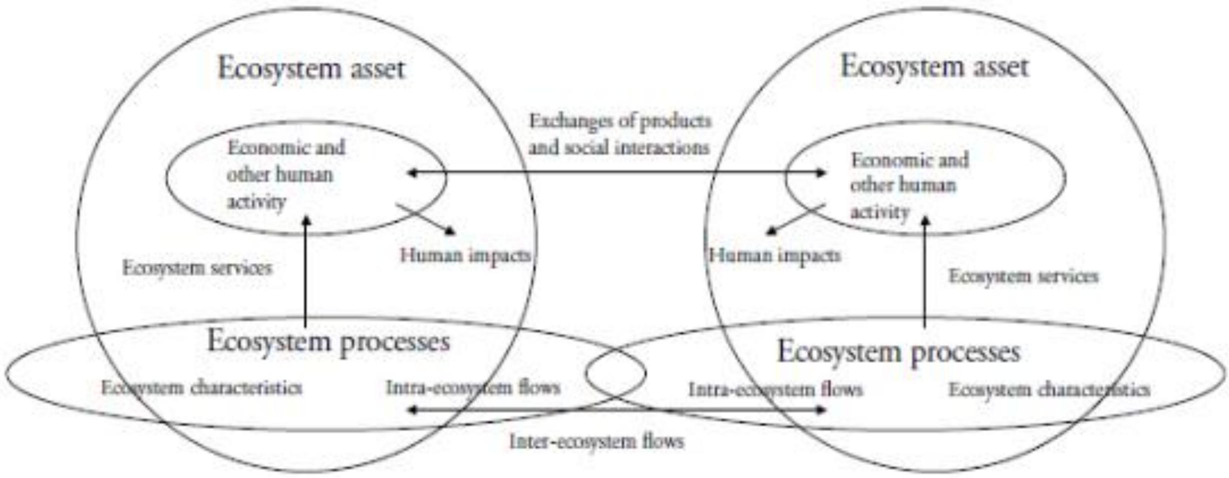

Figure 11. The SEEA-EEA basic model of ecosystem stocks and flows (Figure 2.1 in SEEA-EEA, 2014)

The Ecosystem Condition Account provides the structure for collecting information on the characteristics of ecosystems that drive the flows described in Figure 11**.** The Ecosystem Capacity Account provides the structure for collecting information on the stocks of ecosystem assets (i.e., their capacity to deliver ecosystem services) and changes in these stocks over accounting periods. Consequently, the stocks recorded in the Ecosystem Capacity Account are based on the extent and condition of the ecosystem asset (Aslaksen, 2005). Associated with these accounts is the Ecosystem Services Account, which captures information on the actual flow of ecosystem services into the economy. The linkages between these accounts are described in Section 14.

6.2 Ecosystem Condition Accounts

The purpose of the Ecosystem Condition Account is to collect information on ecosystem characteristics important to ecosystem processes that provide a range of both final and intermediate services. This allows trends in the characteristics of ecosystems (of which biodiversity is one) to be linked to economic and social activity (via the Ecosystem Services Account). This can help reveal both the benefits of well-functioning ecosystems (including biodiversity) to society and promote sustainable ecosystem management.

The capability of ecosystems to provide service flows is dependent on their ecological integrity and good health. Rapport (2007) discusses this metaphor of ecosystem health, expressing a loss of ecosystem functionality from "Ecosystem Distress Syndrome (EDS)" that results in a general loss of ecosystem services. Rapport (2007) identifies biodiversity loss and related concepts, such as shifts in community, loss of endemic species and increases in invasive species to be principal symptoms of EDS. This relationship is borne out by a number of studies that demonstrate the link between biodiversity, ecosystem resilience and ecosystem service provision (see Annex 1). As such, biodiversity is identified as being of particular importance to ecosystem condition in the SEEA-EEA.

Information on the characteristic of biodiversity for an EU or EAU should be recovered from the associated core biodiversity accounting tables. A number of approaches have been discussed in Sections 2, 3 and 4, which can be employed to recover a composite index or indicator of biodiversity condition from the biodiversity accounting tables. In order for these to be useful in the Ecosystem Condition Account they must indicate a comprehensive measure (in terms of taxonomic groups) of species diversity, which would be expected for a healthy, fully functioning ecosystem of the given land cover or habitat type. For example, the suite of indicators compiled for habitat types under the Norwegian Nature Index.

As Mace (2010) identified whilst many rare, endemic and other species contribute to species diversity in an ecosystem, they may not necessarily be important for maintaining the overall functioning of the ecosystem. A composite indicator for such species could be compiled to measure progress toward conservation targets, although this is likely to under represent important functional species groups. This would be the type of indicator recovered from a

Biodiversity Account for threatened species, for example, populated using data from a National Red List. Nonetheless, an aggregated measure of the level of threat to species could be useful as a first estimate of a biodiversity indicator for ecosystem condition (Jean-Louis Weber, 2014).

In order to organise information on the condition of ecosystems (including biodiversity as a characteristic) and their ability to provide ecosystem services, the SEEA-EEA suggests two Ecosystem Condition Account tables. The first (see **Table 10**) provides a framework for capturing an indicator of biodiversity for an EAU or Administrative Accounting Unit (AAU). It should be noted this table relates to an indicator measure recorded at the end of an accounting period.

Table 10. Measures of ecosystem condition and extent at the end of an accounting period for an Ecosystem Accounting Unit (Table 4.3 in SEEA-EEA, 2014)

	Ecosystem extent	Characteristics of ecosystem condition				
		Vegetation	Biodiversity	Soil	Water	Carbon
		Examples of indicators				
Type of LCEU	Area	Leaf area index, biomass, mean annual increment	Species richness, relative abundance	Soil organic matter content, soil carbon, groundwater table	River flow, water quality, fish species	Net carbon balance, primary productivity
Forest tree cover						
Agricultural land[a]						
Urban and associated developed areas						
Open wetlands						

[a] Medium to large fields of rain-fed herbaceous cropland.

The second Ecosystem Condition Account table (**Table 11**) is relevant for accounting at an EU scale. It provides the familiar asset format with opening and closing stocks, together with additions (improvements) and reductions for condition indicators.

Table 11. Changes in ecosystem condition for an Ecosystem Unit (Table 4.4 in SEEA-EEA, 2014)

	Characteristics of ecosystem condition				
	Vegetation	Biodiversity	Soil	Water	Carbon
	Examples of indicators				
	Leaf area index, biomass, mean annual increment	Species richness, relative abundance	Soil organic matter content, soil carbon, groundwater table	River flow, water quality, fish species	Net carbon balance, primary productivity
Opening condition					
Improvements in condition					
Improvements due to natural regeneration (net of normal natural losses)					
Improvements due to human activity					
Reductions in condition					
Reductions due to extraction and harvest of resources					
Reductions due to ongoing human activity					
Catastrophic losses due to human activity					
Catastrophic losses due to natural events					
Closing condition					

Testing the Ecosystem Condition Account is discussed in SEEA-EEA Tech. Guide 3 (2015). It identifies that attributing specific drivers to the improvements and reductions in the condition indicator levels will be difficult for certain ecosystem characteristics. This is likely to be particularly true for biodiversity, where there exist a number of drivers of biodiversity loss. The SEEA-EEA Tech. Guide 3 (2015) suggests a supplementary account for drivers of ecosystem condition as a possibility for testing.

Testing and development of Ecosystem Condition Accounts is currently in progress, for example, for river ecosystem condition accounting in South Africa. In addition, the Australian Bureau of Statistics (ABS, 2015) recently presented experimental Ecosystem Condition Accounts for the Great Barrier Reef (see **Box 17**).

Box 17: An Ecosystem Condition Account for the Great Barrier Reef (GBR) (Australian Bureau of Statistics, 2015)

The Australian Bureau of Statistics (ABS) have compiled an Experimental Ecosystem Account for the Great Barrier Reef Region to organise and link a significant body of scientific work to other accounts produced by ABS. The Ecosystem Condition Account table is populated for the period covering 2007-2008 to 2012-2013. The account considers three broad biomes (terrestrial, river and marine) and associated indexes of the condition of these ecosystems. An accrual approach is adopted, where all indexes are related to the reference condition of the beginning of the first accounting period (2007-2008) on a percentage scale. Indicators relevant to biodiversity are presented for the marine biome only. These reveal general reductions in the condition of coral and seagrass over the accounting periods, suggesting a loss of marine biodiversity. In addition to the condition accounts, ABS also produced physical and monetary accounts of ecosystem services. This allowed ecosystem condition to be compared with ecosystem service provision and associated economic activities, such as agriculture and ecotourism.

Table. Terrestrial and marine ecosystem condition and river loads, Great Barrier Reef region (2007-08 to 2012-13, Index (2007-08=100))

	Terrestrial Condition	River Loads			Marine Condition			
	Average NPP	Solids	Nitrogen	Phosphorous	Coral	Water Quality	Seagrass	Fish numbers
2007-08	100	100	100	100	100	100	100	100
2008-09	97	67	64	57	102	102	97	99
2009-10	91	37	51	58	96	115	94	101
2010-11	110	105	176	197	81	73	53	92
2011-12	98	29	48	47	67	na	53	101
2012-13	94	na	na	na	73	na	78	93

na - not available
NPP - Net Primary Productivity
Source: Summary of data from tables in later chapters

A further consideration on inclusion of biodiversity indicators for the Ecosystem Condition Account is the linkage with ecosystem services and the capacity account. For example, indicators of plant diversity may be important for erosion control services or indicators of soil microorganism diversity for soil fertility services. Whilst a single composite biodiversity indicator may be informative in terms of the condition of an ecosystem, a suite of indicators is likely to be required to establish a meaningful link to the Ecosystem Capacity Account. This is reflected in Box 17 in the selection of three indicators (corals, seagrass and fish) relating to different aspects of marine biodiversity.

Given the indicators of ecosystem condition are relative measures, area weighting is necessary to ensure they remain proportionate across larger spatial scales. The Norwegian Nature Index approach is to apply area weighting is to ensure indicators remain proportionate when aggregated across larger spatial scales (e.g., aggregating all EUs in an EAU). Furthermore, any such approach fails to account for the diversity of ecosystems within the EAU (i.e., gamma biodiversity discussed in Section 2), which is likely to have implications for inter ecosystem service flows. Overall, further testing of aggregation approaches on both pragmatic and theoretical terms is required.

6.3 Ecosystem Capacity Accounts

The Ecosystem Capacity Account organises information on the capacity of ecosystems to deliver a range of ecosystem services. Whilst the concept of a capacity account still requires development, it can offer a useful construct in which to consider ecosystems that provide particular benefits to society and direct more sustainable management of these assets.

The condition of biodiversity within the EU will be significant for the delivery of a number ecosystem services considered in the capacity account. For example, as noted in Section 1.3, biodiversity provides direct benefits to society on the basis of its continued existence and its experiential aspects (e.g., viewing nature). Biodiversity also contributes directly to human well-being by providing a stock of genetic material that provides medicines, crops, livestock and biofuels.

In addition, studies reviewed in Annex 1 identify evidence of a relationship between higher levels of biodiversity and certain ecosystem services. Notably, Tilman et al. (1997) identify a positive relationship between vegetation biodiversity and net primary productivity (and associated provision services). This includes the production of non-timber forest products (NTFP) (Njana et al., 2013) as well as general biomass. In addition there is some evidence to directly link decreases in surface runoff, soil erosion and nutrient loads to plant species richness (Wang et al., 2007). Evidence is also found for a link between species richness and increased pollination and disease control (Hicks et al., 2014).

The Ecosystem Capacity Account adopts the familiar asset account structure, with opening and closing stocks of the ecosystem asset and associated addition and reductions recorded. For cultural and other ecosystem services where biodiversity is valued directly, the capacity of the ecosystem asset would be defined as the carrying capacity of the asset for a given species of conservation importance or, for example, the number of visitors that visit the asset for nature viewing activities. As discussed in SEEA-EEA Tech. Guide 6 (2015), the stocks of some provisioning services (e.g., m^3 timber) may be relatively easy to record. Stocks of regulating services may be somewhat more intangible but could be recorded in terms of a rate (e.g., amount of air pollution removed per unit area of ecosystem per year). For ecosystem service flows whose delivery is related to a measure of biodiversity in a statistically significant manner, discussed above, an ecological production function can be specified. These approaches are described in Banzhaf & Boyd (2012). Nonetheless, capturing indicators of biodiversity that could be useful in informing such production functions and assist in estimating ecosystem asset stocks remains a challenging task. Furthermore, ecosystem production functions will be dependent on particular aspects of biodiversity, relevant to the service. This has clear implications for the selection of indicators for the condition account. Biodiversity will also only be one factor in the production function for a majority of ecosystem services, implying the number of overall indicators in the condition account could become very large where multiple ecosystem services are considered in the capacity account.

Biodiversity also underpins many of the intermediate services that transfer energy and nutrients both within and between ecosystem assets (Balvanera et al., 2006). For ecosystem services associated with specific species (e.g., soil fertility and dung beetle lifecycles), such species will be dependent on these intra- and inter-ecosystem flows to some degree. High levels of biodiversity also provide functional redundancy, which improves ecosystem resilience (Elmquist et al., 2013). This is a very important consideration for the delivery of future ecosystem service flows by ecosystem assets and their capacity to deal with shocks and change. Further research is required to understand the full extent of the relationship between biodiversity and sustaining the delivery of ecosystem service flows. Nonetheless, this remains an important conceptual issue to overcome within ecosystem accounts due to the fundamental importance of biodiversity to ecosystem functioning. Developing approaches to capture the importance of these service flows within is a possible role for the Ecosystem Capacity Account.

Edens & Hein (2013) discuss the challenges of accounting for ecosystem degradation (e.g., due to changes in species composition). Edens & Hein (2013) observe this is particularly complicated due to the existence of non-linear relationships between condition indicators, including biodiversity, and ecosystem services. Dickie et al. (2014) discuss the idea of thresholds for ecosystem condition, beyond which the level of provision of ecosystem services changes and ability of ecosystem assets to recover functionality may be compromised.

The Natural Capital Asset Check (Dickie et al., 2014) developed under the UK National Ecosystem Assessment Follow-on project, provides a structured approach to organise information on whether such a threshold is in danger of being crossed. This approach could be useful in the context of biodiversity in the Ecosystem Capacity Account. This would reveal if the current stock of biodiversity was approaching a level that threatened ecosystem resilience and the delivery of ecosystems services. Accordingly, the stock of inter- and intra-ecosystem flows and also the service flows to the society and the economy, could be recorded in qualitative terms of their threat status.

7. Conclusions

This section identifies key points from exploring the development of Biodiversity Accounts based on the System of Environmental-Economic Accounting Experimental Ecosystem Accounting (SEEA-EEA) framework. It summarises a number of experimental approaches, which vary depending on resource availability, to assist countries starting out on this process. A number of issues to resolve are also discussed. This section concludes with a list of recommendations for testing, refining and validating the approaches mentioned.

7.1 Why account for biodiversity?

To deliver sustainable development, national accounting systems need to fully account for the foundation for sustainable economic growth provided by ecosystems and their services. Biodiversity, which consists of ecosystem, species and genetic diversity, contributes to the 'wealth' of a country by providing benefits that support human well-being and supply inputs into the economy. The SEEA provides an internationally adopted framework for producing national statistics on the environment and its relationship with the economy. Biodiversity accounting, through the SEEA-EEA provides, the methodology to help understand the contribution of biodiversity to the economy by explicitly considering its role in the provision of ecosystem services and as a determinant of ecosystem condition.

Biodiversity accounting allows trends in ecosystem and species diversity (and the benefits they provide) to be compared with economic and social activity in a spatially explicit manner in order to answer key policy questions. Establishing linkages between biodiversity and the economy provides an important opportunity to integrate biodiversity into a wider range of decision making contexts (e.g., sectoral and development policy).

Biodiversity accounting also provides opportunities for the harmonisation of biodiversity information alongside other reporting mechanisms, such as the Sustainable Development Goals (SDGs). There are potential linkages, in terms of the monitoring and use of biodiversity information, with the established suite of headline Biodiversity Indicators under the Convention of Biological Diversity (CBD). Member States to the CBD are also in the process of revising their National Biodiversity Strategies and Action Plans (NBSAPs) and establishing national targets and indicators to track progress towards the Aichi Biodiversity Targets. The NBSAP process presents one mechanism for identifying biodiversity information that could be suitable for accounting purposes.

Biodiversity accounting provides a framework for organising spatial information on biodiversity for aggregation and communication across all scales. The accounting structure allows statistics on biodiversity information to be presented and summarised at the individual ecosystem scale (i.e., at an ecosystem unit), at the scale of ecological features (e.g., watersheds), administrative areas (e.g., counties or departments) and at national scales.

7.2 Getting started

A key starting point for biodiversity accounting is to identify policy priorities and to help determine what information, including on plants, animals and to a lesser extent fungi, should be compiled. This will also establish the resolution of data (both spatial and temporal) necessary to address these priorities.

Establishing an inventory of all existing monitoring data will identify any 'data-gaps' for biodiversity accounting. Countries should consider their reporting obligations to regional processes and biodiversity-related conventions/agreements, such as the CBD and Ramsar. Identifying data gaps could inform a protocol for further data gathering (e.g., via monitoring or modelling approaches).

The first step in the Biodiversity Accounting process is to delineate ecosystem assets spatially on the basis of similarities in ecological and ecosystem characteristics (i.e., Ecosystem Units [EUs]). This can be considered the foundation of a Biodiversity Account in the broadest sense, as it provides information on ecosystem diversity. Changes in the extent of these Ecosystem Units can then be recorded over time (in an Extent Account). Once a spatially explicit system of Ecosystem Units has been determined, Biodiversity Accounts can be constructed for each Ecosystem Unit or for a group of Ecosystem Units (e.g., all Ecosystem Units within a watershed), depending on the policy priorities being addressed.

Due to the complexity of accounting for all components of biodiversity, the Biodiversity Accounts suggested in this document focus mainly on developing information on ecosystem diversity and the construction of 'species accounts'. There is considerable research on species, decades of science on their measurement and many countries have long-term monitoring programs for species. In addition, species are often used as proxies for biodiversity more broadly and can be used as indicators of ecosystem function and condition. Further, developing information within the Extent Account for ecosystem units is considered in order to account for ecosystem diversity. Genetic diversity within species is not considered in this document due to the cost and complexity of assessing genetic diversity.

Measures of species diversity are resource intensive and have methodological challenges. Species richness is a frequently used measure of species diversity but species abundance is a more sensitive measure. These concepts capture the 'quantity' component of biodiversity. The concepts of alpha, beta and gamma diversity capture the 'variation' component of biodiversity. National accounts should attempt to capture gamma diversity and various techniques are being developed for factoring beta diversity into the measurement of biodiversity change across large spatial extents. Therefore, creation of species accounts should be done in consultation with ecologists to ensure meaningful data is collated and collected. Data will likely be a mixture of direct measurement and modelling approaches.

A complete inventory of a country's species is not feasible and instead specific species should be prioritised based on the objective of the Biodiversity Account. The subset of species selected for inclusion should include ones that are important for the delivery of ecosystem services, such as species that play a role in ecosystem functioning (e.g. keystone species). Economically importance species (e.g. viewing game species) is another example. In order to represent biodiversity, the subset of species should sample from across multiple taxonomic groups (e.g. mammals, birds, plants).

More than one Biodiversity Account may be required to answer the full range of biodiversity-relevant policy questions. For instance, information on biodiversity relevant to ecosystem functioning may require a different accounting structure than information on species extinction risk.

A number of experimental approaches to biodiversity accounting that build on the SEEA-EEA are reviewed or suggested in this document. These approaches may be implemented at an Ecosystem Unit scale or across groups of Ecosystem Units on the basis of ecological features, such as watersheds (Ecosystem Accounting Units, [EAUs]), administrative boundaries, such as county borders (Administrative Accounting Units, [AAUs]) or even national scales, according to the availability and necessary resolution of data. These approaches vary in complexity and resource requirements and are presented as a three tiered approach ('Tier 1', 'Tier 2', and 'Tier 3'):

- 'Tier 1' Ecosystem Diversity Account based on the information used to define different Ecosystem Unit classes, the Ecosystem Unit Extent Account and/or additional information on biodiversity important habitat (e.g., Statistics Sweden, 2015).
- 'Tier 1' weighted Ecosystem Diversity Account that applies existing or new species diversity indicators to weight Ecosystem Units in terms of biodiversity quality (e.g., Norwegian Nature Index – Certain et al., 2011; Natural Capital Index – ten Brink & Tekelenburg, 2002; Natural Capital Asset Index – Albon et al., 2014).
- 'Tier 2' Species Richness Account, for example, on the basis of available monitoring data (e.g., Burdekin National Resource Management [NRM] region – Bond et al., 2013). Species richness data can also be estimated for Ecosystem Units using species range data, expert knowledge, distribution models or species-area curves (e.g., Brooks et al., 2002).
- 'Tier 2' Species Extinction Risk Account, for example, on the basis of a compiled National Red List, complemented by the IUCN Red List of Threatened Species (e.g., Burdekin NRM, Bond et al., 2013). This provides important information on the relative importance of the species diversity within Ecosystem Units and an indication of the presence of ecosystem specialists.
- 'Tier 3' Species Abundance Account using existing monitoring data. This can be supported by modelling approaches (e.g., Peru example, Grantham, pers comm.) or expert knowledge (e.g., Biodiversity Intactness Index – Scholes & Biggs, 2005) to generate spatial data to attribute to Ecosystem Units.

Whilst primary monitoring data is the ideal for assigning biodiversity information to ecosystem units this is unlikely to be available at the spatial resolution required for ecosystem accounting. **A number of approaches exist for upscaling or downscaling data on biodiversity, these include habitat modelling, land use modelling, species-area curves and expert judgement approaches.** A portfolio of these approaches may be required to inform biodiversity accounting. Any application of these approaches should be supported by regular updates of primary monitoring data.

7.3 Limitations and issues to resolve

This review has found that the majority of potential global datasets in their present state do not provide the temporal or spatial resolution necessary to inform national biodiversity accounting. One exception is the IUCN Red List on Threatened Species which can complement National Red Lists in assessing species extinction risk. The Red List of Ecosystems, which will complete its first global assessment in 2025, also has the potential to inform national biodiversity accounting.

Whilst a single biodiversity indicator can provide an overall indication of ecosystem condition (potentially for an Ecosystem Capacity Account), it is unlikely to be useful in informing the link to ecosystem service delivery (via an Ecosystem Service Account). This is because there may be many aspects of biodiversity that will be of importance to different ecosystem services. Consequently, a broad suite of biodiversity indicators is likely to be required.

For those aspects of biodiversity considered an ecosystem service in their own right, (e.g. for their existence or aesthetic enjoyment), information in the Biodiversity Account can directly inform the stock of these services recorded in the Ecosystem Capacity Account. Information on the stocks of other ecosystem services recorded in the Ecosystem Condition Account can be linked to biodiversity through the use of ecological production functions, although establishing such a link will be challenging for many services.

The link between biodiversity and ecosystem service delivery is complex. Capturing information on the importance of biodiversity to intermediate services (intra- and inter-ecosystem flows) and ecosystem functional redundancy and resilience is challenging due to non-linear and threshold effects. In addition, there will also often be time lags between changes in biodiversity stocks and resulting changes to the level of ecosystem service provision. Time lags will also exist between social and economic development pressures and their effect on stocks of biodiversity. However, given the importance of biodiversity to ecosystem functioning and sustainable ecosystem service provision, considering ecosystem functional redundancy and resilience is a key issue to be addressed in the ecosystem accounting framework. Further research is required in this regard, however, the Ecosystem Capacity Account provides a useful construct in which to capture and analyse this type of information.

Ultimately the market value of the contribution of biodiversity to ecosystem service provision would be extremely useful to record in the Ecosystem Services Account. Where biodiversity is valued directly as a good (e.g., medicinal plants) or for experiential services (e.g., viewing iconic species) markets may function where such a value can be inferred (e.g., park admission prices). Where market prices are unavailable, revealed preference methods (e.g., the travel cost method) can be used to obtain a welfare estimate for visitors travelling and incurring expense to see iconic species. Stated preference methods can also be employed, which elicit willingness to pay for both use and non-use welfare benefits associated with biodiversity (although communicating such benefits to respondents and capturing meaningful responses in challenging). It should be noted that some adjustment of the welfare estimates derived using revealed and stated preference valuation methods will be required in order to generate a 'market price' suitable for accounting purposes. The SEEA-EEA discusses approaches in this regard. Whilst these approaches are not necessarily straightforward they would generate a lower bound for consideration in the monetary Ecosystem Services Account.

Capturing the entire value of the contribution of biodiversity to ecosystem services is not possible. Ecosystem production functions can be specified to isolate the monetary value of biodiversity to ecosystem service flows into the economy. However, this is likely to only be possible for a subset of very specific and economically important services. Regardless of the final approach, any valuation of biodiversity should be considered an underestimate, as it will be not account for all intermediate services (intra- and inter-ecosystem flows) dependent on biodiversity.

7.4 Recommendations for testing, refining and validating

Further testing of the modelling, expert knowledge and other approaches for generating spatially explicit information on biodiversity described in this document would be useful. For instance, using more modelling instances that are country specific, possibly supported using a smaller subset of relevant biodiversity studies or expert knowledge. Furthermore, adapting these models to report on measures of different taxonomic groups could prove particularly useful. An assessment of protocols for validation or calibration of these approaches to downscaling and upscaling based on existing or new point data for biodiversity would also be useful. This would provide crucial insight into the magnitude of error associated with these approaches.

More testing is required of suitable spatial scales for biodiversity accounting. For instance, none of the example Biodiversity Accounts presented captured information on biodiversity at the Ecosystem Unit scale. As such moving directly to accounting for biodiversity at the Ecosystem Accounting Unit or Administrative Accounting Unit level is recommended as the pragmatic approach in the piloting phase. Whilst some work has been undertaken in the context of rivers in South Africa, more testing is also required around the use of River System Units and Marine Coastal Units and, in particular, linking these to economic statistics.

The issues around appropriate scale also have significant implications for aggregation of biodiversity information. Further research into and testing of methods to aggregate biodiversity information and condition indicators across ecosystem units, is required. This should consider the implications of ecotones between ecosystem units and the diversity between ecosystem units in larger reporting areas, such as Ecosystem Accounting Units (i.e., gamma biodiversity).

The asset accounts for biodiversity recommended in the SEEA-EEA allow for causes of addition and reduction in the stocks of species diversity to be recorded. There are obvious benefits to establishing such a clear causal relationship. However, this will require additional data collection and may often be difficult to complete in a balanced manner. As such the possibilities **for undertaking this would benefit from testing in a specific case study, possibly via linkages with the land ownership or land use.** In the first instance, it is likely to be pragmatic to focus on open and closing stocks only.

Biodiversity is considered as an indicator of condition in the Ecosystem Condition Account. **Improvements and reductions in the indicator levels can also be recorded in the condition account. However, there exist multiple drivers of biodiversity loss and so a supplementary account for drivers of ecosystem condition could be a possibility for testing.** This would also provide a suitable structure for capturing considerations such as habitat fragmentation, invasive species and ecotones.

To conclude, the Biodiversity Accounts presented in this technical guidance document are experimental accounts and are not necessarily prescriptive of a recommended course of action. They require testing, refining, and validation by countries in different contexts and with levels of data availability to determine their practicality to be integrated into national accounting and to inform decision making.

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Annexes

Annex 1 - What is the evidence of the relationships between biodiversity richness, ecological functions, ecological conditions and ecosystem assets and services?

Biodiversity was highlighted by the Millennium Ecosystem Assessment (MA, 2005) as important in ecosystem functions that provided a range of ecosystem services. As a final ecosystem service, biodiversity at both the genetic and species level contributes directly to the provision of goods that provide human welfare (Mace et al., 2012). In addition, biodiversity supports ecosystem functions and therefore indirectly supports important intermediate services (inter- and intra-ecosystem flows).

This Annex provides a summary of the relevant literature around the relationship between biodiversity and:

- ecosystem function and delivery of ecosystem services;
- ecological productivity (ecosystem production function); and
- maintaining resilience of systems;

Relationship between biodiversity and ecosystem function and delivery of ecosystem services

A considerable body of research now exists that examines the relationship between biodiversity and ecosystem functioning (Naeem & Wright, 2003; Balvanera et al., 2006; Luck et al., 2009). Ecosystem function is where biological activity causes the change in energy in matter over time and space[6] (Mori et al., 2013). Functioning ecosystems are essential to the delivery of ecosystem services (Harrison et al., 2014). During their systematic review of this literature, Hooper et al. (2005) identify that species' functional characteristics strongly influence ecosystem properties and 95% of experimental studies support a positive relationship between biodiversity and ecosystem functioning. These functional characteristics of species operate in different contexts, including the effects of keystone species, ecological engineers and the interactions amongst species (Gorman et al., 2011).

Some ecosystem services in particular have been found to be enhanced by higher biodiversity. The relationship between plant species richness and soil erosion processes, shows that surface runoff, soil erosion and total phosphorus loss decreased with increasing plant species richness (Wang et al., 2007). From a social perspective, Njana et al. (2013) found that the well-being of local communities utilising non-timber forest products (NTFP) was dependent on the diversity of tree and shrub species with different functions. There is also evidence of a link between species richness and increased pollination, and also the prevention of human and animal diseases (Hicks et al., 2014).

A number of studies have examined the roles played by particular species groups. For example, Chazdon et al. (2003) and Tabarelli & Peres (2002), have demonstrated the importance of a range of animal groups, including large vertebrates, in dispersing tree seeds to regenerating forest. Dung beetles and ants are particularly important for their contribution to soil fertility (Nielsen, 2007; Folgarait, 1998). But some studies have found a negative relationship between the economic values of ecosystem services with bird species richness (or vascular plant species richness) (Carrasco et al., 2014). Within the scientific community it is recognised that further work on these relationships is required, in particular because ecosystem service delivery is not just related to species richness (only one measure of biodiversity). Some of the most valuable services are associated with particular species (rather than biodiversity in general). There is a high degree of specificity with regard to which plants are useful for medicinal purposes. For example individual plants can become essential for the delivery of benefits such as *Prunus africana* which is highly demanded by pharmaceutical companies in developed countries because it is crucial in treating prostate cancer (Mugula et al., 2010). Individual species also have a value to people in the context of cultural ecosystem services (Mace et al., 2012). Charismatic or iconic species are valued on the basis of aesthetics, their characteristics and behaviour, or because of the cultural status given to them (Kellert, 1997; Martín-López et al., 2007). These species

[6] The changes in energy and matter over time and space occurring through biological activity, such as primary production, nutrient uptake, decomposition, and evapotranspiration. The rates of such functions are often positively associated with higher levels of biodiversity (Balvanera et al., 2006; Cardinale et al., 2007 cited in Mori et al., 2013)

may, or may not, play a significant role in the functioning of the ecosystem. Even if it is the case that they do, then the aspect of value is a different one to their cultural service value (i.e. combined they give the total value (Nunes & van den Bergh, 2001)). It should be recognised that some elements of biodiversity cause dis-services to people including insects acting as disease vectors or where they are disruptive such as crop raiding by wild animals.

While it is the case that individual species can be disproportionately valuable in terms of the ecosystem services that they provide, these species require a functioning ecosystem to support them and as a result it is important to consider biodiversity as a whole as individual species providing services rarely exist in a vacuum.

Ecological productivity and biodiversity (ecosystem production function)

The functioning of ecosystems results in, for example, primary production such as vegetation growth and decomposition of organic matter releasing nutrients (Balvanera et al., 2006).

While it has been found that less than half of species are needed to maintain most ecosystem processes (Schwartz et al., 2000), studies have also shown that ecosystem productivity is also related to biodiversity. The quantity of plant biomass increases with increased biodiversity of the vegetation and therefore biodiversity increases the productivity of ecosystems (Tilman & Downing 1994; Tilman et al., 1997; Flombaum & Sala, 2008). The clearest evidence of this comes from grassland experiments (Thompson et al., 2012) however, Potvin & Gotelli (2008) found that compared to monoculture experimental plots, mixed-species forest plots yielded, on average, 30-58% higher total tree basal area, indicating higher rates of tree growth. Through simulation models they demonstrated that the increased yield of mixed-species plots was due mostly to individual tree growth being enhanced.

Contribution of biodiversity to the resilience of ecosystems

It is generally agreed that biodiversity is key to supporting ecosystem function and the supply of ecosystem although there are still many questions to answer around the mechanisms of how this works (Balvanera et al., 2006; Tilman et al., 2006). An important role, in addition to supporting the supply of services, is in ecosystem resilience and therefore maintaining the flow of services during times of disturbance or stress that the ecosystem experiences. Ecosystem resilience is the capacity of an ecosystem to absorb shocks and disturbance while retaining the same level of fundamental functions (Mori et al., 2013). For example, species richness is generally found to reduce susceptibility of ecosystems to invasions by exotic species (Hooper et al., 2005). Aichi Biodiversity Target 15 specifically relates to ecosystem resilience and the contribution biodiversity makes to carbon stocks enhancement, a key part of climate change mitigation and adaptation (CBD, n.d).

Mechanisms of resilience

Ecosystem resilience is thought to be achieved through *functional redundancy*. Functional redundancy is when an ecosystem contains a number of different species, each with similar functions but which are affected by disturbance in different ways. The variety of reactions to a pressure or disturbance is described as *response diversity* meaning that the same disturbance will not result in decrease in function for all species present. If disturbance causes a species to go extinct in a biodiverse system, one which contains many species with similar functional roles, there are more likely to be species with a similar functional capacity, but different responses to that disturbance, allowing the ecosystem to continue to operate (Elmqvist et al., 2003). Ecosystem changes following loss of species can vary between different ecosystems. Therefore, some ecosystems are less sensitive to disturbance than others. However, intensification of resource extraction and land use change causes environmental disturbance and has been shown to result in a decline in response diversity (Lalibert et al., 2010).

Quantification of functional diversity (the value, range, and relative abundance of traits present in the organisms in an ecological community [MA, 2005c]) can be very difficult. There are a number of reasons for this including the fact that organisms are not clearly divided into distinct functional groups, functional traits and disturbance responses are likely to be different, and species diversity does not perfectly correlate with functional diversity (Hooper et al., 2005). There are non-linear relationships and thresholds for biodiversity and the delivery of ecosystem services (Luck et al., 2009).

Annex 2 – The IUCN Red List of Threatened Species – breath of the Red List and associated training materials

As of May 2015, the Red List has global assessments for 76,199 species, of which 22,413 (29%) are threatened with extinction. All bird species have been assessed six times by BirdLife International (2015), who have produced regular updates to the global list of threatened birds since the 1980s, building on detailed regional Red Data Books. There have been two comprehensive assessments of all mammals (Schipper et al. 2008), amphibians (Stuart et al. 2004), and reef-building corals (Carpenter et al. 2008). Third reassessments are underway for mammals and amphibians, and comprehensive global assessments of reptiles and fishes are far-advanced, with the former including assessment of all seasnakes (Elfes et al. 2013), and the latter all sharks and rays (Dulvy et al. 2014), tarpons, ladyfishes, and bonefishes (Adams et al. 2013), parrotfishes and surgeonfishes (Comeros-Raynal et al. 2012), groupers (Sadovy de Mitcheson et al. 2012), tunas and billfishes (Collette et al. 2011), and hagfishes (Knapp et al. 2011). Other animal groups already comprehensively assessed include freshwater caridean shrimps (De Grave et al. 2015), cone snails (Peters et al. 2013), freshwater crabs (Cumberlidge et al. 2009), and crayfish (Richman et al. 2015). Among plants, comprehensive assessments have been conducted twice for conifers (Farjon & Page 1999) and cycads (Donaldson 2003), and once for mangroves (Polidoro et al. 2010) and seagrasses (Short et al. 2011). In addition, a sampled approach to Red Listing (Baillie et al. 2008) has been implemented for reptiles (Böhm et al. 2013) and dragonflies and damselflies (Clausnitzer et al. 2008), and is being undertaken for various other invertebrate (Lewis & Senior 2011) and plant (Brummitt et al. 2015) taxa. The target is to assess 160,000 species by 2020 (Stuart et al. 2010). Minimum documentation for all assessments includes taxonomic identity, justified application of the IUCN Red List Categories and Criteria, a geographic range map, application of classification schemes for threats and actions needed (Salafsky et al. 2008) plus habitats, and literature cited. All species evaluated for the Red List are required to be re-evaluated every ten years and if not done are marked as “in need of updating” to indicate that the data are old and may no longer be relevant or appropriate.

The Species Information Service is used for the management and storage of the species assessments that underpin the Red List. A spatial database is also used to manage the distribution maps, with longer term plans to integrate the two systems. Web-based open-access discussion forums, such as BirdLife’s Globally Threatened Bird Forums, have emerged as a means for facilitating more efficient and transparent discussion and review, especially of reassessments (Rondinini et al. 2014). The Red List is published online through websites providing search and download functionalities. Web services using Application Programming Interfaces (APIs) are also available for programmatically connecting to the Red List tabular and spatial data. The data are freely available for non-commercial use according to the published terms, and under data licence for commercial use through the Integrated Biodiversity Assessment Tool for Business (IBAT; https://www.ibatforbusiness.org), a global biodiversity decision support platform governed by a partnership among BirdLife International, CI, IUCN and UNEP-WCMC. In addition, IUCN provides tools for assisting with application of the criteria themselves (including calculating generation length, extent of occurrence, and declines under criterion A). Independent tools, including RAMAS Red List, which assists assessors with making assessments under situations of extreme uncertainty (Akçakaya et al. 2000), and GeoCAT, which assists with geospatial analysis especially of point locality data (Bachman et al. 2011), have also been developed.

IUCN receives frequent requests from around the world for advice on red listing and for formal training on how to use the IUCN Red List Categories and Criteria. In response to these requests, IUCN has developed a package of training materials to facilitate understanding and application of IUCN Red List methodology (http://www.iucnredlist.org/technical-documents/red-list-training). This includes comprehensive documents on each aspect of the Red List process; Red List Assessor training workshops that use a mix of short presentations and practical sessions; workshop curriculum to train people with previous Red List experience to become Red List Trainers, and help expand the global network of facilitators for Red List Assessment Training workshops; and a free online training course available in English, French and Spanish. The training and capacity building programme is coordinated by the IUCN Red List Unit and aims to improve the quality and quantity of Red List assessments by supporting individuals in the IUCN network. In addition, IUCN and Partners run assessment workshops around the world, and these provide invaluable opportunities for capacity building and training (Rondinini et al. 2014).