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Biodiversity Accounts and Indices
Preliminary Draft

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Chapter 6. Biodiversity accounts and indices

This is a preliminary draft for Chapter 6 of the UN Statistical Handbook of experimental ecosystem accounting. Lead chapter author is Per Arild Garnåsjordet. Contributing authors will include: Jane McDonald, Peter Cosier, Ben ten Brink, Andrea Saltelli, Bill Magnusson, Signe Nybø, Olav Skarpaas, Iulie Aslaksen and others, including colleagues from Statistics Norway.The draft consists of texts from available relevant sources that – following from discussions with the contributing authors – will be edited and further developed for this chapter. Drawing on the interdisciplinary research on the Nature Index for Norway, an important focus point for the chapter will be to convey the usefulness of this framework for integrated biodiversity measurement for application internationally. The Nature Index has strong similarities with the approach to biodiversity measurement developed in Australia. The Nature Index approach has a comprehensive and flexible structure that represents a large potential for contributing to the core element of an international standard for ecosystem and biodiversity accounting in ecological terms.

1. Introduction

The purpose of this chapter is to explore how biodiversity can be represented and measured in an experimental ecosystem accounting framework. The chapter will review the ecological/biophysical and economic representations of biodiversity with the intention of clarifying how biodiversity can be taken into account in environmental accounting for both economic and ecological audiences, and will highlight where progress can be made towards a comprehensive environmental accounting system. In particular the following issues will be addressed:

- *The concept of biodiversity and why accounting for biodiversity*
- *The relation between SEEA and biodiversity accounting*
- *Compare methods for calculation of biodiversity indices*
- *Propose the structure of a biodiversity diagnostic account*
- *Special methodological problems in biodiversity accounting*
- *Data requirements for compiling biodiversity accounts and how data may be generalized and stored*
- *How biodiversity accounts may be used to set policy targets for biodiversity and be an important element in biodiversity policy.*

Biodiversity is itself a powerful indicator of a healthy and functioning ecosystem. A diagnostic account should provide regular information on all the important parts of an ecosystem, and at the same time present information of how these parts are functioning together (UNSD, EEA, and WB 2011).

2. The concept of biodiversity

The Convention on Biological Diversity (1992) suggests the following definition of biodiversity: "Biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and aquatic ecosystems and the ecological complexes which they are part of; this includes diversity within species, between species and of ecosystems."

Numerous international agreements require nations to reach biodiversity conservation targets including the Convention on Biological Diversity which mandated "to achieve by 2010 a significant reduction of the current rate of biodiversity loss" [3]. A policy goal of preserving biodiversity is part of environmental policy and national legislation of most industrialized countries. These agreements are a reflection of the values society places on biodiversity and the land in which the biodiversity values are contained and provide a clear indication that humanity derives direct and indirect benefits from the protection of biodiversity and moreover, expresses a commitment to preserve existence values of biodiversity. At the 2010 Conference of the Parties to Convention on Biological Diversity (COP 10) in Nagoya, Japan, a target relating to the recording of value of biodiversity was included:

- "Target 2. "By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems." (UNEP/CBD/COP 10 2011)

New methods for developing indicators measuring biodiversity loss, as well as improved and increased biodiversity monitoring giving data to indexes, are needed worldwide. The CBD's scientific body, Subsidiary Body on Scientific, Technical and Technological Advice (SBSSTA), is working on establishing indicators that can be implemented worldwide, on national or regional scale (Subsidary Body on Scientific Technical and Technological Advice 2011).

Biological diversity is intimately linked to ecological sustainability, and closely tied to social and economic sustainability through ecosystem services such as food production, water purification and carbon sequestration (Millenium Ecosystem Assessment 2005, TEEB 2010). The looming biodiversity crisis has urged the international community to set ambitious goals such as the 2010 and 2020 targets of the Convention on Biological diversity (http://www.cbd.int/2010-target/). However, the absence of integrated biodiversity measurement and monitoring tools has constrained the ability of national and international organizations to measure progress and respond to the biodiversity crisis. While data on certain aspects of economically and socially important biodiversity (such as timber, fish stocks and large predators) are collected systematically and regularly, these aspects are measured in very different ways, and the information on many other aspects of biodiversity is limited to expert knowledge (Certain and Skarpaas et al. 2011).

Until the approval of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) in June 2010, the Convention on Biological Diversity, and other international agreements concerned with biodiversity, there was no organized structure for mobilizing the expertise of the large scientific community to inform governments. The effectiveness of the scientific body that advises the Convention on Biological Diversity (CBD) is being undermined by the increasing dominance of politicians and professional negotiators (Brauer 2005, Laikre *et al.* 2008, Ahlroth and Kotiaho 2009) more concerned with the inclusion of trade, economic growth and public opinion in conservation debates than in operational efficiency and scientific verification. What is lacking is a mechanism that is able to bring together the expertise of the scientific community to provide, on a regular basis, validated and independent scientific information relating to biodiversity and ecosystem services, to governments, policymakers, international conventions, non-governmental organizations and the wider public (Loreau *et al.* 2006). The work to establishment of the intergovernmental panel of biodiversity and ecosystem services (IPBES) is recognizing this problem, and is aiming to enhance assessments of biodiversity and ecosystem services worldwide, building on existing methodologies and indicators. In this chapter some of the most relevant work on biodiversity indexes is highlighted.

The world is facing a biodiversity crisis (Hassan et al. 2005). The Convention on Biological Diversity (CBD) has recognized the severity of the biodiversity crisis, and in 2002 the world leaders committed to halt the loss of biodiversity by 2010 (World Summit on Sustainable development in Johannesburg). In spite of the numerous scientific approaches to address biodiversity decline, only a few have been able to synthesize this information to give an overview of the overall trend (Walpole et al. 2009; Butchart et al. 2010). Previous attempts to develop aggregated indexes to give an overview of biodiversity trends have however not resulted in regularly updated national or regional statistics (ten Brink 2000; Scholes & Biggs 2005; Alkemade et al. 2009).

Biodiversity is essentially multi-dimensional and any assessment and accounting standards need to approach it as such. (Office of Technology Assessment 1987). Any approach to accounting for biodiversity needs to take into account the intertwined relationships between biodiversity and the ecosystems it is part of. Protection of biodiversity raises issues of protection of ecosystems, species and genetic diversity. . For establishing accounts for biodiversity and ecosystems, a careful consideration of the concept of biodiversity is recommended, with a step-by-step process to determine what components of biodiversity are being accounted for and why. This will help inform how different aspects and values of biodiversity are accounted for, and what measures are appropriate for different purposes.

3. Why account for biodiversity?

As the increasing recognition of biodiversity loss, reaching scientific and public attention through the decline of particular species and habitats, has led to political initiatives for biodiversity conservation, it has also resulted in closer examination of why biodiversity is important, how it is an essential foundation for life on earth and what role it plays in human civilization. Below is a list of some main findings of the role of biodiversity in recent literature of conservation biology and ecology:

1. An intrinsic part of the natural world (that ought to be protected) (Nash 1989)
2. Responsibility of humanity
3. Critical to achieving sustainability
4. The essential foundations upon when humanity depends [(CBD 2003)
5. Representative of conservation as a whole
6. Aesthetic qualities – (Ehrlich 1981)
7. Biodiversity is a 'good' (Mace 2012)
8. Essential for the functioning of ecosystems that underpin the provisioning of ecosystems that affect human well being (MEA 2005)
9. Insurance against future unknown threats
10. Unknown potential future source of benefits for example pharmaceuticals
11. An indicator of ecosystem condition (Karr 1991)
12. A measure of only species extinctions (ref)
13. A measure of all of biology (Sarkar 2005)
14. A major factor affecting ecosystem stability (Elton 1927,MacArthur 1955. Elton 1958, May1975)
15. Correlated to productivity (more diverse communities are more productive) (Darwin 1872, Di Falco,2009)
16. A input influencing many ecosystem properties (Tilman 1994, MEA 2005, UK NEA 2005)
17. Important element in the functioning of ecosystems
18. Critical to the viability of indigenous communities (WWF 1997)
19. Unique and irreplaceable part of our world
20. Providing incalculable benefits of genetic variability that people everywhere use daily and depend upon (McAfee 1999)
21. Is inherent in all ecosystems and is not an entity that can be separated.
22. Contributes to security, resiliency, social relations, health and freedom of choices and actions (MEA 2005)
23. Biodiversity is synonymous with ecosystem services (TEEB 2010)
24. Supports cultural value
25. Responsibility of humanity

4. Biodiversity accounting, Link to the SEEA

This chapter outlines an approach to biodiversity that aims to take into account the multi-dimensionality suggested by the list above, recognizing the use of biodiversity for human benefit, as well as the intrinsic values of the natural world.

The Experimental Ecosystems Accounts suggested in this Handbook will be closely linked to the System of Environmental-Economic Accounts (SEEA), a multi-purpose conceptual framework that describes the interactions between the economy and the environment and changes in the state of the environment over time (Draft SEEA.manual). The SEEA is based on The UN System of National Accounts (SNA), and its approaches to value accounting, introducing a terminology of environmental asset accounting. The System of National Accounts (SNA) defines assets as: "a store of value representing a benefit or series of benefits accruing to an economic owner ….It is a means of carrying over value from one accounting period to another" (p. 617). Items of value to society are the source of inputs to the economy, to society and also to ecosystems.(UNCEEA 2011). Assets are accounted for as stocks in order to measure their depletion and degradation. Asset accounts seek to measure the quantity, value (and condition in the case of ecosystem accounting) in order to record and explain changes in value over time.

There are many parallels between economic accounts and environmental accounts but there is one important difference. Economic policy is focused on improving living standards by continually *expanding* the value of the flows of good and services, whereas environmental policy is about *maintaining* the stock (condition) of natural capital, including ecosystems, so that they continue to provide services to humanity into the future (Cosier 2011).

The System of Environmental-Economic Accounts (SEEA) Experimental Ecosystem Accounts have been proposed to address elements of the SEEA Central Framework for which there is no current statistical standard. The SEEA defines an environmental asset as "naturally occurring living and non-living components of the Earth, together comprising the bio-physical environment, that may provide benefits to humanity". (UNCEEA 2011). Biodiversity is therefore already in the scope of SEEA. Some biodiversity, for example, farm animals and wild species subject to commercial harvest, are covered by the SEEA Central Framework. However, a large part of biodiversity is not explicitly included in the asset classes described in the SEEA Central Framework or System of National Accounts, because they do not provide precisely defined benefits to humanity that can be converted to economic terms through prices. Hence, biodiversity for which there are no direct economic benefits needs to be addressed in the development of the SEEA Experimental Ecosystem Accounting.

The main objective for introducing Experimental Ecosystem Accounts will be to compare the overall challenges in environmental policies in different countries. The accounts for biodiversity will not be identical, both because of natural variations and because of differences in the use of natural resources and human pressures on ecosystems and biodiversity. Indeed the experimental ecosystem accounts of SEEA may be the start of a trend towards common monitoring schemes. Buckland (2005) states that "such schemes will allow greater power in measuring changes and determining the reasons for them, and will provide economy of scale so that nations that otherwise would have been unable to monitor their own biodiversity adequately can participate." Using the same accounting framework may make it possible to compare how different biodiversity polices are implemented in different countries and how trade-offs between biodiversity priorities and economic priorities are expressed.

From the ecological point of departure, biodiversity measurement generally concentrates on the diversity of species of flora and fauna, as well as the abundance, function, community composition and distribution of these species. There are a number of good reasons for this: a) species are the subject of most international treaties and national policy on biodiversity, b) species are relatively conspicuous, c) there is considerable research on species, with decades of scientific effort on the measurement of species and many long-term monitoring programs for species, d) species are often used as a surrogate for biodiversity in general, e) ecosystems consist of species and the abiotic environment, and f) indirectly the species level reflects the genetic pool of this species.

From the economic point of departure, accounting for biodiversity in terms of physical assets is seen as the most straightforward and intuitive method of accounting, and most likely to appeal to the scientific and wider public audience. Asset accounting is based on an opening stock and closing stock at the end of the accounting period. It can be argued that ecosystems, species, populations and genes in principle can be accounted for as stocks. In scientific terms a stock can be thought of as a measure of state or a state variable.

Sustainable use of an ecosystem may be interpreted as a flow of goods from the environmental stocks involved, maintaining its ecological capacity. Destruction of ecosystems and biodiversity represents depletion and degrading of environmental assets. Measuring depletion and degradation is an important challenge in stock accounting. Net reductions in biodiversity as observed in the change in stock accounts are measured as depletions. With renewable resources the depletion rate depends on both the rate of additions and reductions. Biodiversity decline can be renewable and non-renewable, hence is important to how depletion is measured.

At a local scale biodiversity may in some cases be renewable when inputs can occur from surrounding area or by natural regeneration. For example, in the case of a population, the input would be the number of births, immigrations and translocations. In the case of species diversity, it would be an increase in species from surrounding areas or reintroductions. In the case of large-scale (national or global) measures of species diversity, biodiversity is a non-renewable resource because once a species becomes extinct, it will not be renewed. Therefore, measures to assess the depletion of biodiversity (extinction rate) and policy instruments to counteract biodiversity loss become extremely important to policy makers.

There may be scope for improvements in measures of biodiversity by expressing biodiversity in terms of flows of goods from environmental stocks. For example, the Habitat Hectares metric has been used as a surrogate for biodiversity in Victoria, Australia (Parkes 2003). This is an index of vegetation condition, extent and connectivity relative to an 'undisturbed' benchmark. An Environmental Benefits Index is another method of measuring bundles of services of which habitat for flora and fauna is one

(Eigeraam 2011). The UK Ecosystem Assessment created a new subcategory of 'wild species diversity' (UK NEA 2011).

Valuation of ecosystem services in monetary terms is currently the focus of work by the TEEB, the World Bank, and many others. Flows of environmental goods will interact with the economy where markets are established for their trade. Numerous objections can be raised against relying on monetary valuation of ecosystem services as a primary policy tool (Spash 2008). On a local level, however, one may envision the use of measurement of ecosystem services as tool for environmental management. Eigenraam et al are currently using ecosystem services (bundles of flows as measured as Environmental Benefit Indices) to distribute stewardship funds for the Victorian Department of Sustainability and Environment in Australia (Eigeraam 2011).

Attempts by the Netherlands Environmental Assessment Agency PBL to use the Mean Species Abundance index (formerly known as the Natural Capital Index) to calculate the economic value of biodiversity were however not considered very successful. For example, the policy option of Reduced Impact Logging (RIL) showed almost zero impact in terms of monetary value of biodiversity because RIL does not make a difference in terms of area affected but in terms of quality preserved **–** which was not registered for lack of unit values (Bakkes 2011). The topic of valuation of ecosystem services will be further discussed in another chapter of this Handbook.

5. Generalisation of biodiversity measures, the use of indices and composite indicators.

5.1 The Information pyramid.

The complexity of biodiversity and its relationships to ecosystems probably implies that selecting only a few indicators will not give anywhere near a reasonably complete picture of this multivariate concept, and a "system of indicators" is needed. Different types of indicator systems may then be used in different ways for communication and policy purposes, representing the different perspectives and values for different stakeholders. The information pyramid in Figure 1 (ten Brink 2006) illustrates the process from raw data through calculation procedures to single and then to composite indicators.

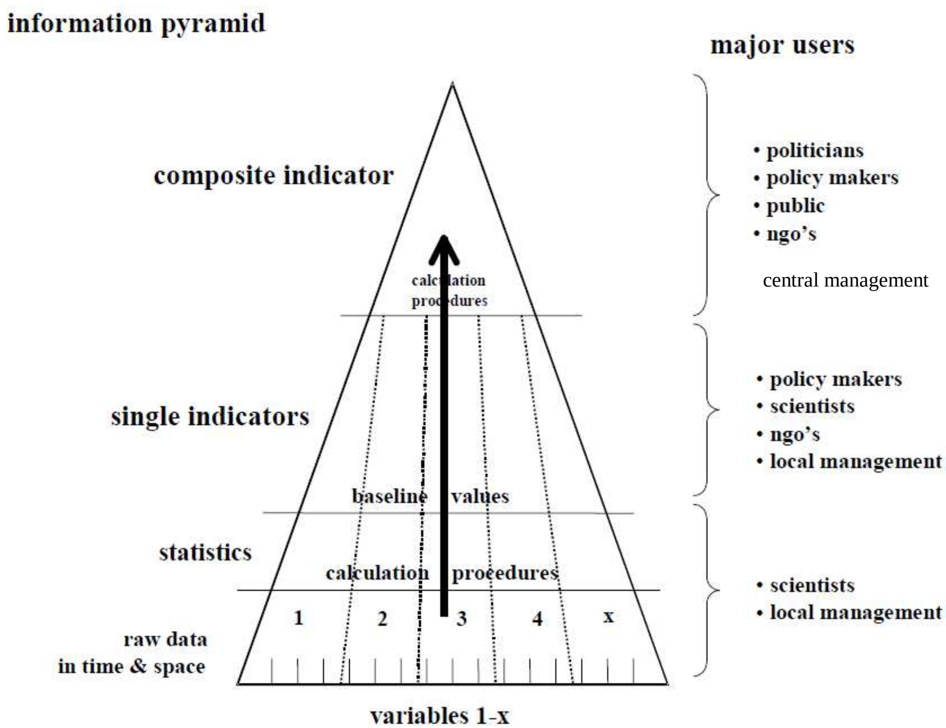

Figure 1 The Information pyramid

As in any other aggregation and compilation of scientific information of high complexity, this process is not "value-neutral", and every step entails normative choices and evaluations. The information pyramid demonstrates that there may of considerable interest to link the different levels to each other through systematic procedures as to observe how changes in a single indicator or sets of indicators may explain changes in a composite indicator. Hence, a composite biodiversity indicator may be seen as a "tip of the iceberg", where the underlying information serves to support the interpretation of the aggregate index.

5.2 Composite indicators.

Composite indicators (CI) have to a large extent been successful as policy instruments. There has been a lot of work in the field of composite indicators and some have got a lot of attention. Notably the Human Development Index (UN), Environmental Sustainability Index (World Economic Forum), Dashboard of Sustainability (EU) and Ecological Footprints (land, water and carbon) are interesting examples, but the list is longer (Moldan et al 2004).

Composite indicator is a manipulation of individual indicators, and possibly weights, to produce an aggregate (ordinal or cardinal) measure. They are data-based narratives – implicitly describing relationships between indicators belonging to different dimensions - and as such a model that represents reality in a certain way (Saltelli xx). Their primary function is to concentrate statistical information in order to present an overall picture, and the aggregation necessarily has to reflect certain assumptions regarding differences, trends and normative values. It may be argued that composite indicators may be interpreted as advocacy of pre-conceived normative assumptions reflected in the choice of indicators and data (Boulanger 2007).

The first step in the construction of a composite indicator is to establish a clear understanding of the multidimensional phenomenon to be measured and make a nested structure in terms of various sub-groups of this phenomenon. Then a criteria list for the selection of individual indicators is required. The rest of the construction process requires careful considerations in terms of imputation of missing data, multivariate analysis of auto-correlations, normalization, weighting and aggregation, robustness and sensitivity analysis (OECD 2008). Even different ways of presenting composite indicators may influence their interpretation.

There is a large literature on composite indicators. There are many arguments of pros and cons As summarized in Figure 2 (Saltelli 2007).

Pros	**Cons**
CI can be used to summarise complex or multidimensional issues.	**CI** may send misleading, non-robust policy messages if they are poorly constructed or misinterpreted.
CI provides the big picture.	The construction of **CI** involves several stages where judgement and selection has to be made,
CI helps attracting public interest	There could be more disagreement about **CI** than on individual indicators
CI can help to reduce the number of indicators	The **CI** increases the quantity of data needed both for completeness and for statistical analysis

Figure 2.Pro and cons for the use of composite indicators

”Composite Indicators are much like mathematical models: As such their construction owes more to the craftmanship of the modeller than to the universally accepted scientific rules for encoding. As for models, the justification for a composite indicator lies in its fitness to the intended purpose and the acceptance of peers.” (Rosen 1991.)

If the composite indicators are to be used in decision making, they have to be scientifically accepted, easy to understand and they should make it possible to discuss what type of trends that are unwanted and may require policy action

5.3 Composite biodiversity indicators .

As discussed above, the protection of biodiversity raises issues of protection of ecosystems, species and genetic diversity. A careful consideration of these aspects of biodiversity is needed, in order to recognize what components of biodiversity are being accounted for and what measures are appropriate for different purposes. There is clearly no simple way of selecting biodiversity indicators for measuring variations in genes, species and ecosystems that could be used for all major ecosystems, as the various nature types or biomes in the world are very different in terms of complexity and diversity.

The purpose of ecosystem and biodiversity accounting is to develop ways to measure the overall loss of biodiversity, considering the interactions between ecosystems and biodiversity. As human activity increases, it becomes crucially important to assess and counteract the degradation and loss of biodiversity. By the selection of species and their populations different trends can be measured. Recently, ecologists have developed many different composite indicators in order to characterize different properties of ecosystems and biodiversity. In general most of the indicators are measures of species variations and changes in the population of the different species.

Table 1 shows an overview of 14 different biodiversity indicators. The table starts with internationally well known indicators of abundance and rareness of species. Then more typical statistical measures of ecological complexity are mentioned. These are the Simpson and Shannon indices. The Natural Capital Index introduced the concept of reference conditions to measure both the quantity and the quality of an ecosystem compared to a natural condition. These ideas have been used in many of the later indices. The next step was the introduction of trophic function and organization. The use of an index for biotic integrity has a long tradition in the analysis of aquatic ecosystems and as a part of watershed management. The Marine Trophic Index is another special index describing “fishing down the food-chain”, and how modern large-scale fisheries have changed the large marine ecosystems.

The European Water Framework systematically used biological indices or proxies for biological conditions (physical and chemical concentrations) to define “good” water quality, and in 2008 these were calibrated among all the European countries. As the purpose was better watershed management, the degree of uncertainty became important as well in the assessment of water quality as basis for management decisions.

The Biodiversity Intactness Index is closely linked to The Natural Capital Index but is more flexible in terms of analysis. The total indexes for the different ecosystems are not the only important measure, but it is also important how different types of changes in groups of species, or within a geographic region, or by human pressure factors may be monitored. Expert judgements were made to calculate how land-use changes may generate changes in biodiversity conditions. This may be useful method in data-poor region. Even in such areas there may exist real data for part of the biosphere, and it would have been useful to supplement model data with empirically observed data-sets.

The Nature Index, developed and implemented in Norway, allows for this possibility of combining different types of data: monitoring data, model based data and expert judgement. It also demonstrates clearly where there are available data, the quality of these data in terms of uncertainty assessments, and indicates for which important parts of the ecosystems there is a lack of knowledge. The experiences achieved in the implantation of the of The Nature Index helps to illustrate both theoretical and practical issues that will have to be addressed in any process of biodiversity accounting, including the final steps of communicating the results and policy formulation.

Table 1: Biodiversity indices. (based on McDonald 2011)

Index	Measure	Characteristics	Baseline	Purpose and scale	Source
Wild Bird Index	Abundance of birds	Group of birds f.ex. Farm birds, Seabirds, Wood-land birds	UK 1970, Eurostat 1990	Indicator of biodiversity, National	Gregory et al (2004). Eurostat 2011
Living Planet,	Abundance of different species	7953 species, interpolations and extrapolation	WWF, UNEP 1970,	Indicator of biodiversity, Global, Regional	Loh (2002) Loh et al (2005)
Species Assemblage Trend Index	Abundance of species	Can be different group of species, taxonomic groups, endemic species or threathened species.	CBS,NGO's Various years	Indicator of biodiversity, Regional	Brink (2006)
Red List Index	Change in rareness status	Species extinction risk by weighting the extinction risk of all species of a particular taxonomic group	IUCN, Now	Indicator of biodiversity, Global, National	Butchart (2004)
Simpson Index	Statistical measure of species richness and relative abundance	The probability that two randomly selected individuals belong to two different species	Now	Indicator of biodiversity, Any scale	Simpson (1949)
Shannon Index	Statistical measure of richness and evenness (relative abundance)	Measuring the order/disorder in a particular system (entropy)	Now	Indicator of biodiversity, Any scale	Shannon (1948)
Natural Capital Index	Area of ecosystem and mean abundance of core set of species	Quantity and quality, both natural and cultural ecosystems	Netherland, Pre-industrial or low impact	Indicator of 'quality' of ecosystem, Regional	Brink (2002)
Mean Species Abundance	Abundance based on modeling	Pressure factors from human activities impacting on different land use and physical characteristics	UNEP,OECD, Pristine or primary vegetation	Indicator of 'quality' of ecosystem, Regional	Alkemade et al (2009)
Index of Biotic Integrity	Species composition and relative abundance of fish	Trophic function and organi-sation, reproductive behaviour. Expert judgements of quality	Natural state	Indicator of ecosystem condition, Regional	Karr (1981).
Sustainable Rivers Index	Functional diversity of macro-invertebrates and nativeness of fish	Functional and structural links between ecosystem components, biophysical condition and human intervention. Sampling and modelling.	Reference condition (undisturbed)	Indicator of ecosystem condition, Regional	Davies et al (2010)
Marine Trophic Index	Position of species in the food chain	Replacement indices used to describe the interactions between fisheries and marine ecosystems	FAO,CBD. Now	Biodiversity composition' Regional	Pauly (1998) Watson et al (2004)
The Water Quality Index	Quality of inland surface waters, transitional waters, coastal waters and groundwater	Indicator species and physico-chemical parameters for ecological classification and how to deal with uncertainty	EU, calibrated 2008, close to undisturbed conditions	Indicator of ecosystem quality, Regional, National	Kallis et al (2001)
Biodiversity Intactness	Abundance of species, constructed for data-poor regions	Calculated from land use and land cover data based on expert judgements. May be disaggregated in terms of taxa, ecosystems and land-uses. Uncertainty measures.	Naturalness as observed in national parks	Indicator of biodiversity, Any scale	Scholes and Biggs (2005) Biggs et al (2006) Hui et al (2008)
Nature Index	Species or proxy for species, cover both terrestrial and marine ecosystems	Based on data, models and expert judgments (125 scientists). Data for 1950, 1990, 2000 and 2010. 308 indicators - representation of all major trophic levels. Uncertainty measures.	Norway, Undisturbed or sustainably managed	Indicator of biodiversity, Any scale	Certain et al (2011) Nybø et al (2012) Skarpaas et al (2012)

5.4 The Nature Index

In general a biodiversity indicator may refer to a population of a single species, a genetic metric, a functional diversity index, a demographic parameter, a community metric, or any other metric fitting the definition.

The Nature Index was presented for Norway in 2010, collecting data for 1990, 2000 and 2010, and where available, for 1950 (Nybø ed. 2012, Certain and Skarpaas et al 2011, Nybø et al 2012). The Nature Index is based on species or proxy measures for species as representative of different parts of ecosystems conditions. The definition of indicator was:

: **"**A natural parameter related to any aspect of biodiversity, supposed to respond to environmental modification and representative for a delimited area. It is a parameter for which a reference value can be estimated. The set of indicators should cover as homogeneously as possible all aspects of biodiversity, and any addition of a new indicator should result in the addition of an amount of independent information" (Certain and Skarpaas et al 2011).

In total 309 biodiversity indicators were used in the Nature Index. The nature Index was established in a comprehensive cooperation between leading research institutions, where 125 scientists participated in defining the criteria, selecting the biodiversity indicators, and entering the data, consisting of monitoring data, model based data and expert judgement.

There are a number of criteria for measures of biodiversity that should be used as a guide to inform choice in existing biodiversity metrics and the design of new ones:

1. Established measures of biodiversity component of interest and relevant pressure factors. The indicator set has also been designed so that individual indicators are sensitive to different environmental pressures, such as land use change and habitat fragmentation, overexploitation, pollution, climate change and invasive alien species.
2. Representative. Patchy data related to incomplete taxonomic and geographic coverage, are generally a problem when aggregating biodiversity data (Walpole et al. 2009). Also, composite indices entirely based on monitoring data are often biased with respect to the overall state of biodiversity
3. Measures change – time series
4. Scalable in term of components and geography
5. Scientifically robust based on data and expert knowledge

The purpose of the Nature Index is to give an overall picture of the state of natural biodiversity, based on resident species. All the major taxonomic groups are represented, both common and rare species should be represented, indicators should be complementary with regard to their response to anthropogenic pressures, keystone species should be included when possible, and a wide variety of ecosystems and habitat should be represented by the indicator set. Furthermore, keystone species have been given extra weight as extra-representative indicators, as these species are important to populations of several hundreds of other species (Certain and Skarpaas et al. 2011).

The formula used to calculate the indicator - values is close to that of Natural Capital and even closer to that of the biodiversity intactness index.

$$NI_t = \sum_{ijk} S_{ijkt} W_{ijkt}$$

Where:

S = State ,W= Weighted at trophic level, t= time, i= species, j= ecosystem, k= geographical unit (municipality)

The weighting system was designed to control for biases arising from the over or under representation of certain taxa and functional groups (Certain and Skarpaas et al. 2011). For example, to ensure that different functional levels of the ecosystems are represented, it is assumed that 8 functional groups (carnivores, herbivores, primary producers etc.) contribute equally and in total 50 % to the final value of the Nature Index. Extra-representative indicators, such as keystone species and indicators representing the status of many species, count the remaining 50% of the overall index within each major ecosystem.

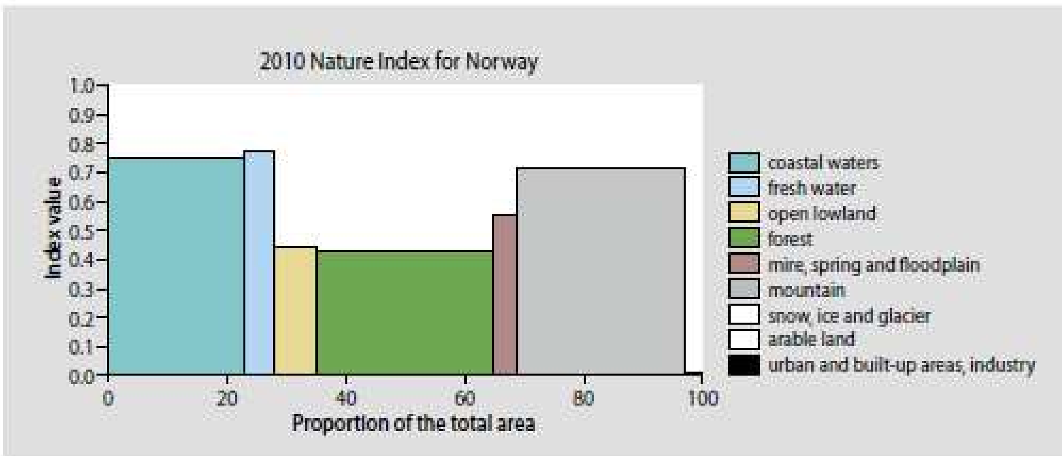

Figure 3. Nature index for Norway 2010.

The Nature Index for Norway showed some difference in the state of biodiversity between the major ecosystems. In 2010 the state of biodiversity was highest in mountains, ocean, coast and freshwater (NI = 0.69-0.80), intermediate for mires and wetlands (NI = 0.55), while open lowlands and forests had the lowest NI values (NI= 0.43-0.44). This reflects that the impacts of forestry and modern agriculture are quite visible, while the marine environment seems to be relatively well managed.

There has been a moderate decline since 1950, although the data are uncertain. There are different trends in the different major ecosystems the last 20 years. The NI increased 8 -10 % in freshwater and the ocean (bottom and pelagic) from 1990 - 2010, but decreased by > 10 % in open lowlands during the same period. The condition in fresh water has improved as a consequence of less acidification and pollution control.

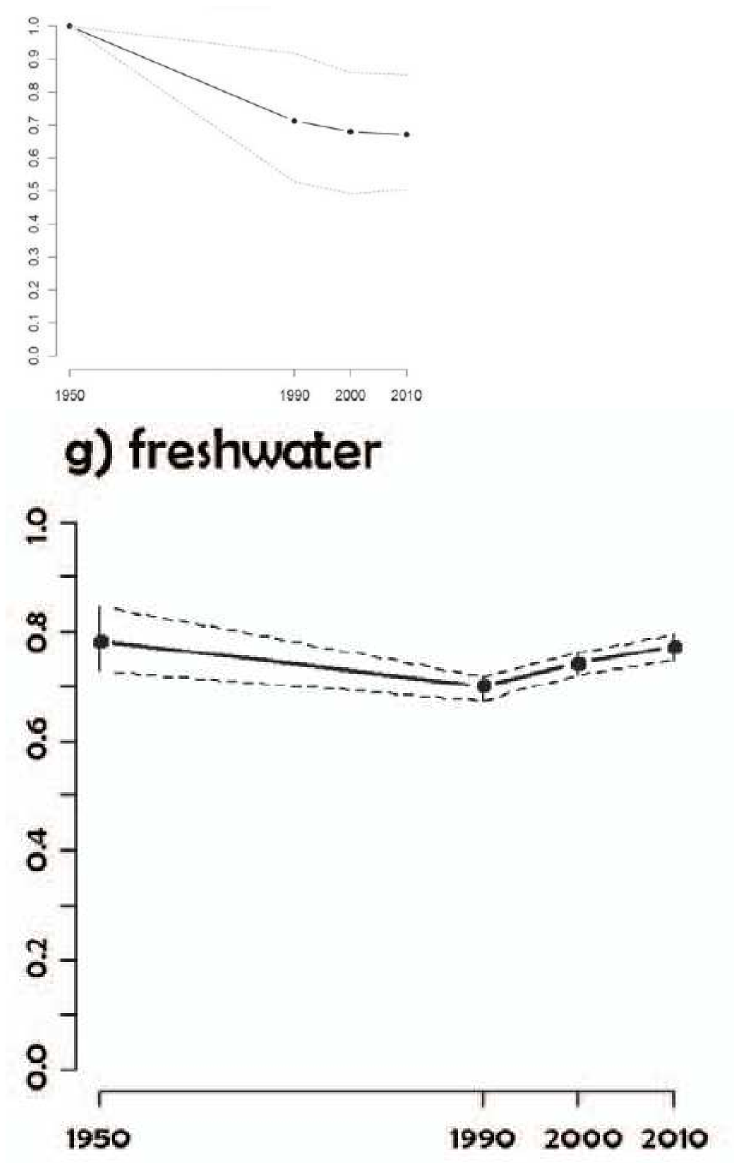

Figure 4 The Nature Index for Norway1950-2010, general and in Freshwater

In the Nature Index a decomposition of the index makes it possible to show what type of changes are behind the aggregate picture. As data were gathered on many species and geographical regions, and experts recorded additional information on pressure factors relevant for the species, it is possible to calculate different thematic indices. A thematic index consists of a sub-set of indicators reflecting different aspects of biodiversity or environmental management issues. Thematic indices can be calculated along the dimensions of species groups, taxonomic groups or major ecosystem (Fig. 5), but also for species being sensitive to defined pressure factors. Thematic indices are regarded as being a particularly relevant tool to illustrate specific management issues, e.g. indices for commercial fish stocks, populations of top carnivores, and impacts of forestry on biodiversity

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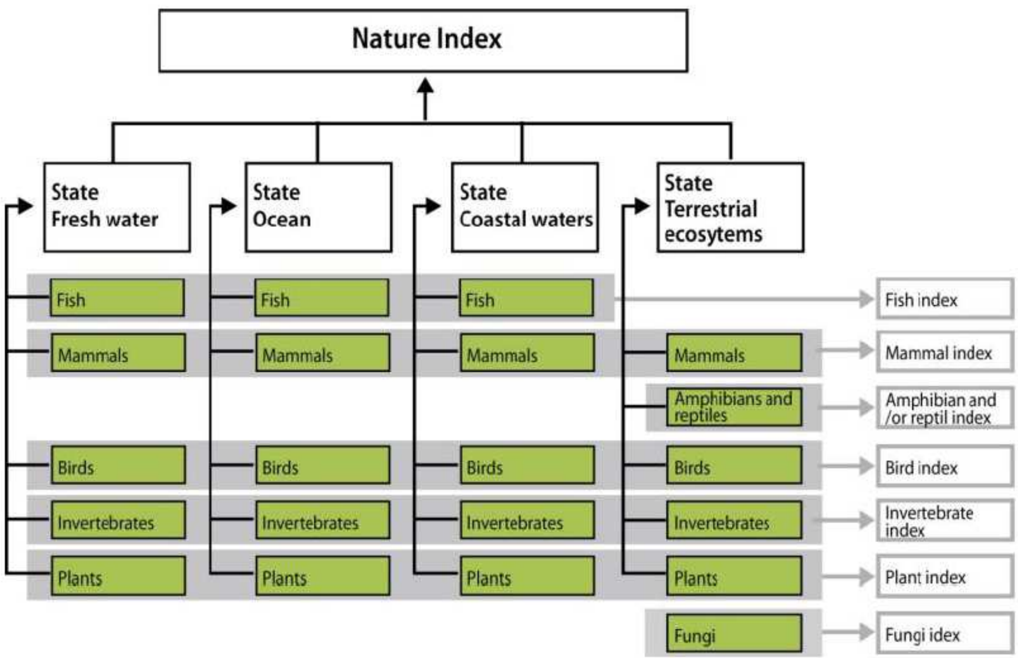

Figure 5. Examples of how a composite indicator can be presented in terms of sub-indicators – each contributing its own narrative.(Nybø et al 2012)

The NI can be aggregated or disaggregated in different ways to produce various management-related thematic indexes at national, regional and local levels. Thematic indexes are formed by taking a subset of the indicators in order to illustrate a particular issue of biological or political concern (Nybø & Skarpaas 2008, Certain et al. 2011). For instance, by combining indicators that are tightly linked through trophic interactions (a food web) in the mountain ecosystem, large-scale geographical variation in the food web (trophic cascade) becomes evident (Pedersen & Eide 2010). These patterns may be related to climate-driven changes in rodent dynamics (Ims et al. 2008), . The Nature index can be aggregated or disaggregated to address specific management themes and how the NI framework provides information not only on the state of biodiversity, but also on lack of knowledge and lack of data, which can be used to inform and optimize research and management policies.

6. How to measure biodiversity some common problems for all composite biodiversity indicators.?

6.1 Reference values or baselines

An appropriate indicator set is an important first step towards a meaningful account of biodiversity. However, to compare and combine indicators in a meaningful way, appropriate reference values or baselines are equally important. In fact, some authors distinguish indicators from statistics by the use of a reference point or baseline, allowing the significance of the statistic as indicating change over time to be gauged(ten Brink 2006).

Comment [OS1]: Only under a particular definition of “indicator”

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Comment [OS2]: Only under a particular definition of "indicator"

Identifying appropriate indicators and establishing benchmarks are an important process [47] and will greatly affect the usefulness of the accounts. The phenomenon of shifting baselines has been undermining measures of biodiversity based on arbitrary baseline[(MEA2005)

The functions of benchmarks (baselines) in a biodiversity index are clearly articulated by ten Brink 2006 (below):

- *Give meaning to raw data*
- *Allow aggregation of different indicators into coherent composite indicators*
- *Make biodiversity indicators comparable within and between countries*
- *Simplify communication with politicians and the public*
- *Provide a fair and common denominator for all countries, being in different stages of economic development.*

Reference values or baselines have been implemented in many different contexts and in different ways. Concurrent with the development of the field of ecosystem health was the development of aquatic bioassessment, primarily in the form of the index of 'biotic integrity' for American rivers Wright, J.F. and al 1984) and the efforts in Great Britain to classify rivers using biological assessments [. Both were responding to a policy need to understand, measure and contribute to the management of the quality of rivers. The great contribution of this work was recognising and establishing a common baseline. That each metric was based on a comparison to a regional reference site with relatively little modification was an innovation in the Index of Biological Integrity (Fausch 1984), but implicit in the Wright et al river assessment methods which are based on identifying unpolluted rivers (Wright, J.F. and al 1984).

The incorporation of biogeographic variation, also identified by Karr et al 1991, inherently changed the capacity of bioassessment methodology to contribute to environmental accounting. The fundamental difference was that sites could now be compared, change observed on the same scale and very simply, provided 'criteria for what is excellent and what is poor (Miller 1988). Using a reference condition benchmark recognised that ecosystems exhibit natural variation in productivity, structure, diversity etc and what might be considered a low condition for one ecosystem, based on these measures, could be healthy for another if that is the natural levels expected for that system (eg rainforests vs deserts) (Davies 2010). Studies that have adopted the Rapport indicators of ecosystem health have applied similar benchmarks (Weber 20007, UN 2003).

While it is possible to express many physical environmental assets as quantities in standard units, there is no established unit for measuring the condition (quality) of biodiversity and ecosystems. However, by comparing indicator levels to a common reference or baseline (as in many of the composite indexes listed in section 5.3, table 1 indicators can be made comparable. In the Nature Index framework this is achieved through the "reference state", defined as follows:

Comment [OS3]: Link to scaling, move to end?

"The reference state, for each biodiversity indicator, is supposed to reflect an ecologically sustainable state for this indicator. The reference value, i.e. the numerical value of the indicator in the reference state, is a value that minimises the probability of extinction of this indicator (or of the species/community to which it is related), maximises the biodiversity of the natural habitat to which it is related, or at least does not threaten biodiversity in this or any other habitat." (Certain and Skarpaas et al. 2011)

The reference value is used to scale the observed value of each indicator, so that all scaled indicator values are directly comparable. Note that there is no need that all indicators share the same reference state. Reference states can be defined specifically for each indicator, according to the current state of knowledge on each indicators and ecosystems. The constraints are that the reference state chosen by the expert does not deviate substantially from the definition above, it corresponds to well formulated hypotheses and assumptions so that it is tractable, and points toward high biological diversity.

There are, in practice, several ways to estimate such a reference value. In addition to pristine or near-pristine conditions (used in e.g. BII and NCI, see overview in Table 1) there are several concepts that correspond to the general definition of a reference state above. For instance, the ecological concept of carrying capacity, precautionary harvesting levels, traditionally managed habitat and best theoretical value of composite indicators. In the Nature Index for Norway 2010, all of these and several others were used to set reference values for different indicators (Certain and Skarpaas et al. 2011). This gives a flexible mosaic of reference states that captures the diversity of natural and semi-natural ecosystems and facilitates the inclusion of many different kinds of data. However, because the definitions were used to differing degrees in different ecosystems, comparisons across ecosystems have been questioned by various stakeholders. For instance, forest ecosystems, with a predominance of pristine reference states, were deemed incomparable to oceans, with a predominance of precautionary harvesting levels as the reference. Similarly, open lowland, i.e. the cultural landscape, was judged different from most other ecosystem because many of its central indicators used traditional agricultural management as the reference. In ongoing work efforts are made to reach a common framework for defining reference states in terms of intact ecosystems (not pristine), in the sense of human influences that maintain biodiversity in the system intact, but without negative pressures.

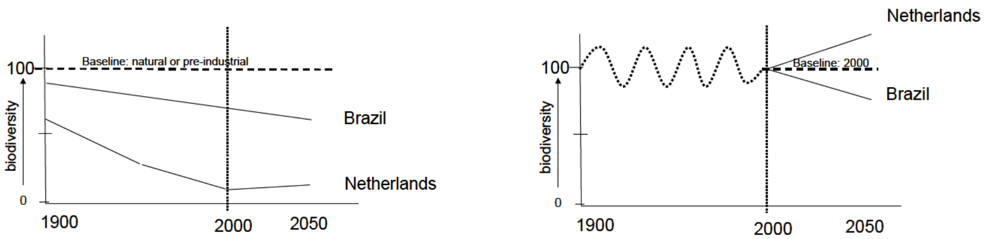


Figure 6. Illustration of the problem of shifting baselines. With a common fixed baseline (left) both the deviation from the baseline and the current trend are visible and comparable among countries or regions; with shifting baselines (right) the only information remaining is the recent trend relative to a state that may be desirable or undesirable, rendering a comparison of states among countries impossible (from ten Brink 2006).

6.2 Normalisation and Scaling

Identifying a reference condition provides a scale to understand the current observations, as well as to simplify, quantify, communicate and most importantly standardise information. This scale is very useful in analysing change over the gradient of human disturbance, as we try to understand the relationships between human activity and ecological disturbance. This is also a significant progression from earlier binary assessment that measured whether a site was degraded or not [12]. It is also an appropriate measure for describing where ecosystems are approaching critical thresholds, which are common in complex ecosystems [60-63].

Scaling is the process of relating observed values to a reference. This can be carried out in different ways. In the BII the indicators are scaled by experts reporting the remaining populations of plants and animals as a fraction of the original population. This simplistic approach may be suitable when all indicators relate to a common reference state in the same manner. However, as discussed above indicators can relate to different reference state in different ways, and so different approaches for scaling may be needed. In the Nature Index framework, these approaches are referred to as ‘scaling models’ (Figure 7). In general the reference state should describe a situation with the highest biodiversity. Ecological thresholds are expressed in terms of the different scaling models chosen for different biodiversity indicators. The Nature Index framework invokes three types of general scaling models with different ecological thresholds (Certain and Skarpaas et al. 2011).

The "optimal" scaling model (figure 7 a) assumes an optimal indicator value (e.g. population size) in the reference state and that any departure from the reference state results in a degradation of the ecosystem to which the indicator is related. This may be useful in the case of indicators related to species such as moose *Alces alces*, that may experience strong decline (e.g. because of hunting), but whose increase in large numbers may also be detrimental to the ecosystem (through increasing grazing pressure). In this model the reference value is a threshold where the interpretation of an increase in the unscaled indicator shifts from positive to negative, and two times the reference value is a threshold between the increasingly negative intrepretation and a range of values to which the scaled indicator is insensitive (the situation is as bad as can be; for all of these values the scaled indicator is zero).

In many cases the decline of an indicator from the reference state reflects a negative development, as for example in marine management of small pelagic fishes, where the reference value refers to a low, precautionary level. In such cases, the "minimal" scaling model (figure 7 b) is appropriate. The model says that a deteriorated state for the indicator only corresponds to a decrease below the reference level, and that any value above this reference level corresponds to an optimal situation. In this model, the reference value is the threshold between a positive interpretation of the unscaled indicator and a region of insensitivity (the situation is as good as can be; the scaled indicator is always one).

When the reference state refers to a "maximal" value, for example a maximal limit for the density of a proliferating species or community (e.g. phytoplankton, jellyfish) above which detrimental effects on ecosystems are observed, the "maximal" scaling model is appropriate. Again the reference value is the lower threshold for a sensitive region with a negative interpretation, and two times the reference value is the upper threshold for this region.

Comment [frb4]: Er vel bedre med for eksempel her. e.g.?

The scaling models in the Nature Index were chosen for their simplicity. This facilitates interpretation and communication. However, there are also some known problems with these functions. The combination of non-linearities in the scaling functions around the reference states and uncertainty in indicator values have complex effects on the estimates of the aggregated index calculated from the indicators (Pedersen & Skarpaas 2012). This may cause difficulties in the interpretation of the index near the reference states, because the index value may change when the uncertainty increases and the index becomes less sensitive to changes near the reference state. This means that reporting on improvements becomes increasingly more difficult. However, the problem is less for the LOW and MAX models than for the OPT model, and there are ways to quantify and corrected the problem (Pedersen & Skarpaas 2012).

Comment [OS5]: Pedersen, B & Skarpaas, O 2012. Statistiske egenskaper til Naturindeks for Norge: Usikkerhet i datagrunnlaget og sensitivitet. NINA Rapport 797

These issues with the scaling models reflect general tradeoffs between the need for simplicity in communication and proper representation of natural phenomena, such as thresholds and tipping points. On the one hand, introducing thresholds where there are none in nature may give undesired results and unnecessary difficulties in interpretation, as discussed above. In such cases, natural processes may perhaps be better represented with continuous scaling models, such as logistic functions, at the cost of introducing at least one more parameter, and thereby another aspect to estimate, communicate and discuss. On the other hand, for indicators that do exhibit thresholds and tipping points in nature, such as species extinctions or shifts between equilibrium states, it is important to take these into account when designing scaling models. A precautionary approach should be used when setting thresholds to keep the planet within a "safe operating space" (Mace, Rockström et al. 2009). The simple general scaling models in the Nature Index framework may serve as starting points for considering such thresholds.

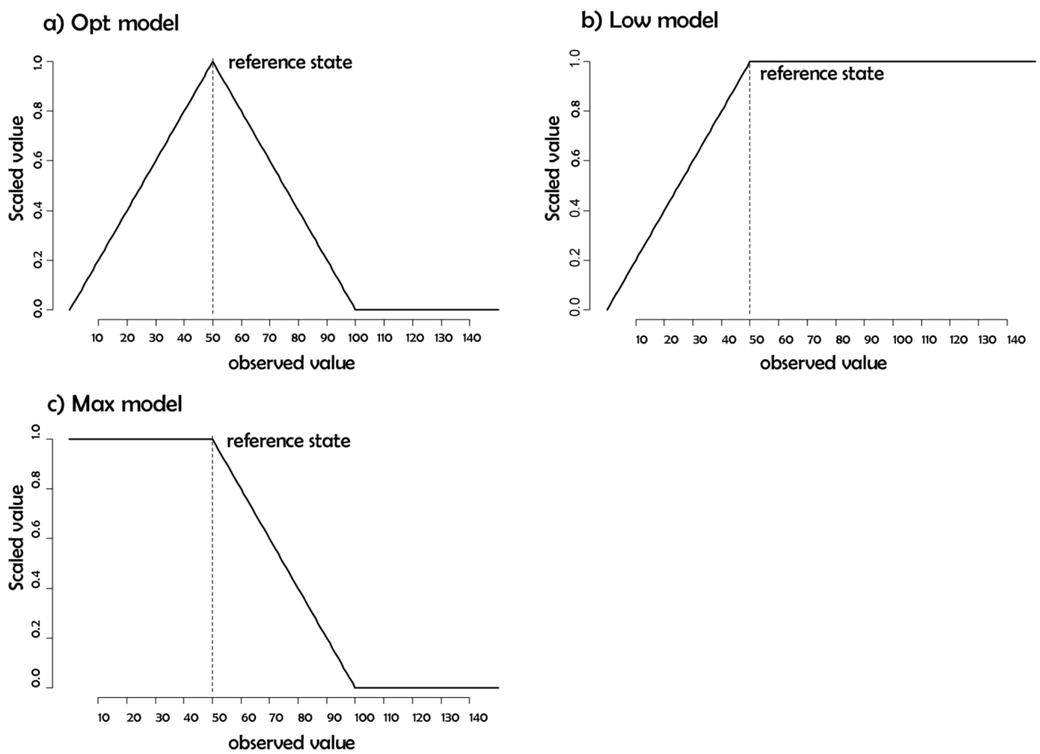


Figure 7. Different models of thresholds and reference states in biodiversity assessment – the nature index for Norway. Source: Certain and Skarpaas et al. (2011).

6.3 Geographical scales.

Nature types or administrative boundaries or both?

In several contexts there is a discussion on how one should deal with the geography in calculating biodiversity indices. The use of remote sensing data and other cartographic material makes it possible to delineate natural spatial units. These may be based on watersheds, elevation, primary production properties etc. The use of Catchment as a basic unit is quite common: In the UAS the National Science Foundation and NGOs are cooperating to create a National Ecological Observatory Network (NEON) that use catchment areas as the sampling unit to monitor land use change, biodiversity and invasive species (Magnusson 2011). The problem is that catchment area may not be uniform, they vary in sizes and human activity as well as pressures may vary substantially. The result is that biodiversity can not easily be correlated with these kind of units – it is far more complex.

Another possibility may thus be to start out with some kind of administrative units, delineate major ecosystems within this unit and describe the biodiversity of each of the major ecosystems. These administrative units may be properties or they may be local communities or municipalities.

To illustrate this difference in reasoning a simple map with to administrative units and with borders delineating major ecosystems or "biomes" may be analyzed. In figure 8 there are 4 ecosystems and 2 administrative units. The ecosystems may be more or less homogenous in terms of their biodiversity indices, but by having ecosystem x administrative unit as basic observation unit we may be able to differentiate the values for the same ecosystem between the administrative units. If the size of the ecosystems (both terrestrial and marine) are considerably larger than the administrative units the possibility to differentiate in terms of geography can be used to show differences in biodiversity conditions.

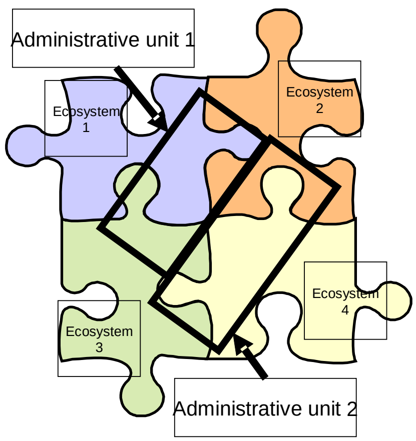


Figure 8 Geographical boundaries

In some countries the administrative boundaries will more or less follow some sort of natural barriers so the fragmentation due to the mismatch of boundaries may not be that serious. The alternative to this approach would be to use smaller and more detailed ecosystems as units, but even these would not be homogeneous, and they would be difficult to add up to an administrative unit.

Geography and decision making.
Another important issue is that since biodiversity varies geographically, so does biodiversity loss and the range of required biodiversity policies and the consequences of these policies for different interest groups. Biodiversity policy objectives and instruments may be defined and decided nationally, but any implementation will have to be done locally. Even if the national government has the authority to make laws and national guidelines, the implementation will have to be done by a regional or local government.

Hence to collect information on a local level, and establish a system where provinces, regions and local government may understand how they may contribute in the implementation process of a biodiversity policy is important. This is even more important because changes in land use represent perhaps the largest pressure factor and cause of biodiversity reduction, and land use change is a policy area very much influenced by regional and local politics in most countries.

The definition of major ecosystems. Some examples
Normally composite biodiversity indicators refer to the condition in major ecosystem in a certain region. The UK National Ecosystem Assessment used 8 major ecosystems to quantify the status of ecosystem processes, assets and the final ecosystem they generate across the country, including: Mountains, moors and heath lands; Semi-natural grasslands; Enclosed farmland including arable and

improved grasslands; Woodlands; Freshwater, wetlands and floodplains; Urban, Coastal margins; and the Marine Environment. In Norway the Nature Index used 9 major ecosystems: Marine water pelagic; Marine water bottom; Coastal water pelagic; Coastal water bottom; Freshwater; Open lowland; Forest; Mire-wetlands; Mountains (excluding Agricultural land and Urban areas).

Normally a limited number of major ecosystems will be selected for this kind of analysis. It may be based on more general land use classification systems, and the delineation may be based on maps or remote sensing. Anyhow the final selection will depend on the policy issues that are framed as the reason for the analysis. What are the questions to be addressed? Indeed there may be several, expressed as a series of narratives and formulated as a multipurpose grammar describing the various reasons for biodiversity conservation and the consequences as seen from the perspectives of different stakeholders (Giampietro, Mayumi, Sorman 2011, Garnåsjordet et al 2012).

7.Data acquisition. Data formats and web-solutions.

7.1 Basic data.

Basic ecological data if collected by scientists will nearly always be based on some sort of systematic sampling. These data are often monitoring data based on systematic sampling- either in a regular network like the one implemented by Alberta Biodiversity Monitoring Institute (AMBI) with different protocols for different ecological systems or it may be more general grids like the European Lucas ***(ref)*** which is close to the forestry sampling systems used in Western Europe, or it might be more loosely define long terms strategies for data collection like the RAPELD system in Brasil (Rapid Assessment Projecto Ecologico de Longa Duracao). (Magnusson 2011).

Even in data rich countries however, systematic inventories of the environment are highly limited. General investigations and assessments are most concentrated to sites with high human activity or which are most problematic in terms of pollution. Water quality measurement for instance will systematically be in the most problematic parts of lakes, rivers and coastal areas.

The quality of the data is crucial. Lack of robust methodology, lack of peer reviewed data acquisition processes and poor tracebility of secondary data have been reported (xx 2010)

8.2 Data storage (based on Magnusson 2011)
It is important that data are stored in a way that the original data formats are evident.

Data management is about five questions: Who will store it? What to store? Where to store it? How to store and retrieve it? To whom should it be available? Those questions are not independent, but they represent one way of segmenting a complex problem. There are complex layers of hardware and software behind each question, but here we will deal with the problem from the point of view of a biologist or science administrator rather than a computer scientist.

What to store?

Place-based research results in many types of data, such as research plans, maps, measurements, scientific publications, photographs and information on biological collections, each with different storage and access requirements (Billick 2010). Space is no longer the major limitation.

The most important thing to make available is the metadata (data that describe primary data). “Without information on what the data represent and how they were collected, it is typically impossible for individuals not involved in the actual study to use the data” (Billick 2010). Conversely, even without the data, it is often very useful to know what has been done, where, and by whom. Many so-called data-management systems, especially the national ones, only record metadata (see below). When original data have been manipulated, or results presented, it is very important that processing and analysis details are documented as workflow metadata. This can be done by using the scripts for analyses used in statistical programs, or with specific workflow systems, such as (Reichman et al. 2011). Many standard statistical analyses of today will be considered quaint or inappropriate in the future (e.g. Warton & Hui 2011).

Where to store it?

There are now many commercial sites that allow you to store and make available information, such as photographs, films and publications, on the internet. These have the advantage of ease of use and generally have no direct costs. However, they have limitations in terms of links between different types of information, and the security of the data may be insufficient. Although they may be useful in some cases to mirror the data, they should not be the primary repository.

Stable systems require redundancy, and that is achieved in data-management systems by data mirrors that allow the same data to be stored in a number of different sites. There are a number of ways that can be done, but the Knowledge Network for Biocomplexity (KNB), which is the system used by ILTER and several other major ecological programs seems to be a promising alternative. Basically, data is mirrored to several different locations on different continents and can be accessed through any of them, but data uploading is the responsibility of the primary node.

How to store and retrieve data?

Data are usually provided by researchers in commercial spreadsheets (e.g. Excel) or commercial data bases (e.g. Dbase), non-proprietary databases (e.g. MySQL) or files associated with statistical programs (e.g. SPSS or R). The first requirement is therefore to get them out of those formats and into simple text tables. The Knowledge Network for Biocomplexity has developed the METACAT system to store data and the associated metadata (http://knb.ecoinformatics.org/index.jsp). The METACAT can make use of the system to search the data and integrate it with data from other international LTER sites. KNB has developed MORPHO (http://knb.ecoinformatics.org/morphoportal.jsp) to allow production and upload of data and metadata to METACAT.

When the Nature Index was developed and implemented in Norway, an Ecological Research Network (ERN) was established as a permanent forum of experienced ecologists and conservation biologists, in order to provide biodiversity data for the general, integrated Nature Index framework, designed to collect and synthesise information from all available sources, as a tractable, calibrated and scientifically-based knowledge base to environmental management and policy makers on the current knowledge of the state of ecosystems.

7.3 The Generalisation from basic data to standard data models and expert judgements

With some few exceptions the translation of the basic scientific data to a more general assessment for specific species may not be straightforward. Biodiversity data are found in numerous diverse data sources, not only in peer-reviewed publications. In some cases the monitoring data are based on comprehensive statistical sampling and the data may directly be entered into a common database.

Modelling based on land use changes and pressure factors are one of the most used techniques in the generalization process. The problem is however that biodiversity is extremely difficult to model. The result is at best a very rough assessment. Modelling may however be part of the assessment process.

Ecologists and conservation biologists usually study specific ecosystems for long periods of time, often most of their life. The combination of basic data and expert assessments regarding the individual species seems to be the most promising strategy for biodiversity measurement. The knowledge integration performed by the human mind seems still to be the most powerful possibility for generalisations. Explicit use of expert judgment is useful because this information, which constitutes a large and important part of knowledge on biodiversity in (Norway between 70 and 80% ref) , would otherwise be neglected or only implicitly used. In cases where only strongly biased data are available (e.g. from a highly polluted river in an area with otherwise healthy water bodies), expert judgment may be an important corrective to the data. However, in other cases, individual expert judgment may, for several reasons, be biased compared to a more classical, empirical approach. Using a high number of experts is one way to control for these biases. Even if calibration experiments attempted on similar expert-estimate collection process showed a reasonable accuracy of expert performance (Scholes & Biggs 2005), it is likely that expert-based judgments result in increased uncertainty (Johnson &

Gillingham 2004). It is therefore recommended to replace expert judgments with well-conducted monitoring whenever possible. However, given that expert judgment is likely to remain an important source of information in many areas in the foreseeable future, calibration should be used (for example simultaneous collection of expert estimates and field data, see Garthwaite et al. 2005) to assess the relevance, precision and bias of expert-based judgments. Even if monitoring data are obtained there might still be a need for expert judgments. The objective is to have a simultaneous generalization process both in space and time. The purpose is not only to collect a snap shot of the situation today but to say something about trends and how they have developed in different regions.

The flexibility of the Nature Index framework allows for applying a combination of monitoring data, model based data and expert judgement, hence utilizing all available data sources and realizing the potential of the knowledge integration by experienced scientists. This procedure can be implemented in data-rich and data-poor areas alike, it contains information on both the state of biodiversity and the state of the scientific experts' knowledge, and it can be aggregated or disaggregated to address specific management themes, which gives this framework the potential to become an efficient management tool (Certain and Skarpaas et al 2011).

7.4 Data quality and uncertainty

In the Nature Index framework, each of the experts enter data for the biodiversity indicators into a common database, giving their best assessments of the (median) indicator values for 1950, 1990, 2000 and 2010, in addition to the reference value for the indicator, as well as the lower and upper quartiles. This information is recorded by the experts for each spatial unit, which is the local administrative unit of municipality. It is important to register what type of data this is. Hence, each expert indicate the type of information, whether it is based on statistical sampling, field observations, monitoring, models or expert judgement. Recording the name of the expert allows for future meta-data inquiries. The data should be given in physical units and not truncated, scaled or normalized. The database will most efficiently be put on a web-server, making online maps from the data-entering process, assisting the researchers to secure the overall quality of the procedure. Figure 9 shows a part of the data-entry interface for the Nature Index. The experts were also asked to provide additional qualitative information, including impact factors for biodiversity loss specified for each indicator.

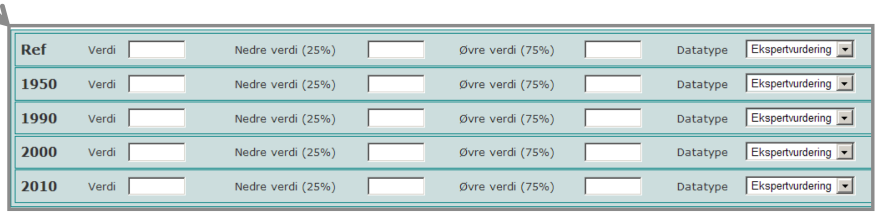


Figure 9. Website interface for entering data into the Nature Index database

The experts were asked to provide an estimate of data uncertainty of each indicator value in the form of quartiles, and they were asked to indicate where information was missing or insufficient, in order to provide an estimate of the uncertainty of the indicator value. The information on uncertainty is crucial in at least two respects. First, the uncertainty in each indicator allows estimation of uncertainty in the aggregated index and in thematic indexes (Certain and Skarpaas 2010; Certain et al. 2011). Second, the experience from implementing the Nature Index in Norway showed that acknowledging and displaying the uncertainty increased the trust and willingness of experts to participate and contribute their expertise and admittedly uncertain data and judgments. Altogether, the information on the type of data (expert judgment, monitoring data or models), numerical uncertainty and missing values for each indicator gives a detailed picture of the state of current knowledge of biodiversity. In the implementation of the Nature Index framework in Norway data uncertainty and missing data are analyzed and actively used in several ways. Numerical uncertainty in the indicator estimates is aggregated to the index level using Monte Carlo methods (Certain et al. 2011). This uncertainty can readily be displayed in plots showing trends over time.

Although there may not be a high awareness of scientific uncertainty in the general public (cf. the recent IPCC debate), addressing uncertainty is an important part of scientific practice, and it is likely that the science-policy communication on management of biodiversity will benefit from an open discussion of uncertainty. In the case of the Nature Index for Norway, the number of biodiversity indicators available for forest was high whereas the uncertainty of recent estimates was low (Certain et al. 2011, Nybø et al. 2012), suggesting that improved management and conservation actions are more important than increased monitoring in this ecosystem. In the ecosystems of ocean, coast, and mountains, the confidence intervals were wider and trends unclear (Nybø 2010b, Certain et al. 2011, Nybø et al. 2012), indicating that increased research and monitoring efforts in these ecosystems would be beneficial.

7.5 Organizing the data acquisition process

The data collection needs to be organized such as to use all the expertise available. In data-rich countries this may be facilitated by involving specialized research institutes for the different major ecosystems. In data-poor countries much of the knowledge may be abroad in international collections, museums or universities. Hence, the organization of the data acquisition process should be properly documented in terms of funding and delivery contracts. The participants will have to be peer-reviewed or the quality will have to be guaranteed by their institutions. For developing countries this process may be a way to get control of their data. The data acquisition process will normally include a large number of scientists - in the Nature Index implementation in Norway 125 scientists participated, in the UK National Ecosystem Assessment there were 16. The scientists have to interact in order to agree on a common framework, calibrate reference values, and develop criteria for indicator selections. Indeed the meetings in themselves may contribute to generate new approaches and contribute to a common understanding of the methodology (Figari 2012).

8. Communication and policy formulation.

Biodiversity accounting is complex issue and is certainly not easily communicated and understood by policy makers and the general public. A study of biodiversity policy in Norway showed that youth politicians have difficulties in communicating biodiversity as policy issue (Seippel 2012).

Some of the main challenges include (based on Cosier 2011):

1. **Information and communication.** Data about the quality of ecosystems and how it changes over time may be described by the biodiversity indicators. The communication itself is however a process that may require changes in the indicator set and the “ narratives” that are formulated. This is a process which does not only involve scientists alone, but also the public, interest groups, and policy makers.
2. **Informing policy.** Biodiversity indicators provide numerical measures for environmental management objectives or targets. They may be used for describing policy trade-offs between biodiversity changes and economic development.

3. **Guiding investment decisions.** Active adaptation management may be formulated based on cost-benefit analyses or multi-criteria analyses that make it possible to evaluate the cost-effectiveness in environmental management and repair.

8.1 Information and communication.

The science-policy communication in the area of biodiversity is challenging. Composite indicators can be interpreted as data-based "narratives". To present trends or development over time in terms of changes in biodiversity indicators it is necessary to explain what the specific causes for these changes are. In the Nature Index framework, the experts providing data are asked to specify impact factors for biodiversity loss for each indicator. When the Nature Index for Norway was reported, the experts were specifically asked to apply thematic indexes and "narratives" about biodiversity changes in the communication to readers.

There are numerous ways to present comprehensive sets of single indicators, displayed in different types of tables and diagrams. To be comparable they have to refer to some common reference value or base line. Diagrams may show how different indicators have changed over time for a specific region, series of maps may display regional variation, and tables may show what indicators show the most serious development by categorization and use of colours. An interesting technique is the amoeba-approach that refers to different functions in an ecosystem (ten Brink et al. 1988)

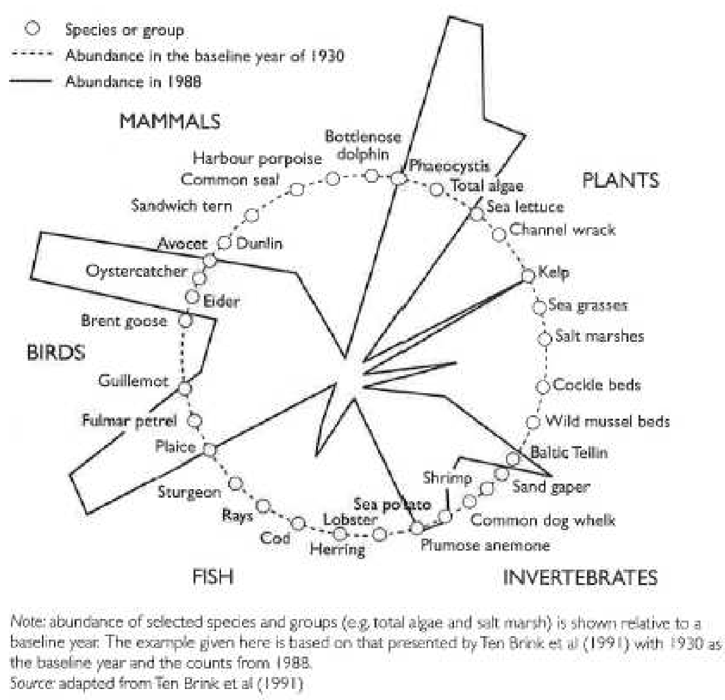


Figure 10. The Amoeba approach for displaying and communicating biodiversity indicators

There are several interesting features with the Amoeba approach. The reference point is the condition in 1930, probably close to what might have been pristine conditions. 1988 figure are probably based on real data, but the reference condition must be based on some modelling efforts or expert judgements. The other interesting feature is that of the functional representation of the ecosystem by clearly showing the development in different trophic levels. The last observation is that this is clearly a diagram meant for reasoning about policy and decision- making in a way that is easily understandable.

The communication process itself represents a double loop of learning (Garnåsjordet et al 20121). The indicator set as well as the monitoring and reporting will be changed over time by adopting a participatory approach including both the general public and national and local stakeholders.

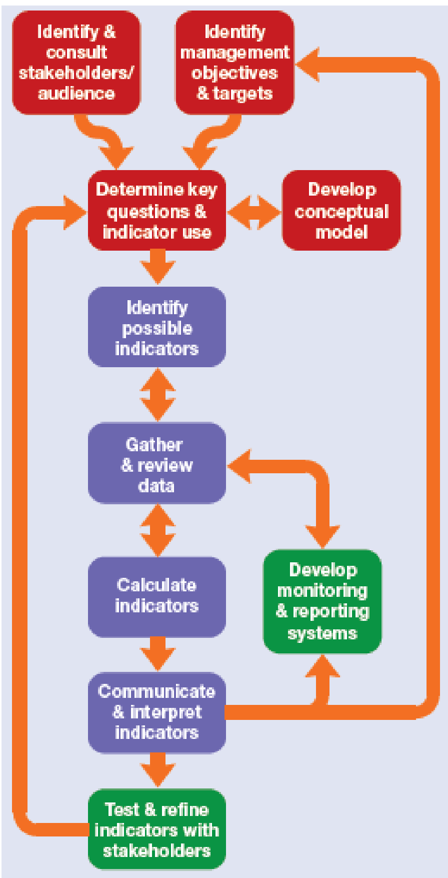


Figure 11. Framework for national indicator development and use

Source: Biodiversity Indicators Partnership (2010)

The extensive cooperation involved in establishing the Nature Index, challenging a large number of experts from different research institutions to provide biodiversity information and "forecasts", extrapolate beyond the normal statistical basis for their data, evaluate ecosystem uncertainties, and consider the need for active biodiversity management policies, raise the question of whether the process of establishing the Nature Index framework and its potential application to biodiversity policy can be understood in terms of post-normal science (Funtowicz and Ravetz 1990, Ravetz 2003). Post-normal science can be defined as the extension of scientific practice into situations when scientists take into account the intertwined relationships between facts and values, the possibility of catastrophic decision-stakes, the legitimate plurality of conflicting interests and ethical complexities, beyond what is usual in normal scientific practice. Moreover, the idea of post-normal science is that involvement and participation of stakeholders and citizens may contribute to improve the quality of the policy deliberations (Funtowicz and Strand 2011). It has been argued by Francis and Goodman (2010) that biodiversity policy can be understood as an example of post-normal science as it "represents a range of urgent problems that require immediate attention but cannot be adequately addressed by current scientific knowledge or methods, relies heavily on practitioners who are not scientific experts, (an extended 'peer community'), where decisions made may have substantial repercussions regarding human lives and livelihoods, and in which laypersons from a range of backgrounds have a stake". The rationale for this position may be found in the importance of biodiversity as the very basis for life supporting ecosystems, providing an ethical imperative to give priority to protection of biodiversity (Heywood and Iriondo 2003). As a tool for biodiversity policy, the Nature Index can be applied for different purposes, serving to express the political objectives or serving as input for different types of deliberations or communication in society.

Biodiversity and healthy ecosystem, securing ecosystem functioning and services, are of vital importance to human society. Suitable communication of biodiversity loss and its consequences is urgently needed on all policy levels. Policy making needs to be aware of the importance of "narratives" of trade-offs between biodiversity and economic development. The United Nations Environmental Program suggests the following advice for communication about biodiversity indicators (UNEP 2011):

- Be clear about what the indicators are telling,
- Be transparent about uncertainty
- Use maps
- Avoid oversimplification
- Economics metric is useful, but do not ignore non-monetary values.

8.2 Policy targets

The Convention on Biological Diversity (CBD) has suggested that policymakers address the following questions when considering biodiversity: What is changing, why is it changing, why is it important and what can be done? (ten Brink 2006). These key questions relate directly to the policy cycle and feed back principles. Cost-effective biodiversity management is only possible if following conditions are met (Wiener):

1. There are verifiable policy targets for biodiversity
2. Timely and sufficient knowledge of the current and projected states is established as well as of progress made towards the targets.
3. It is possible to make corrections.

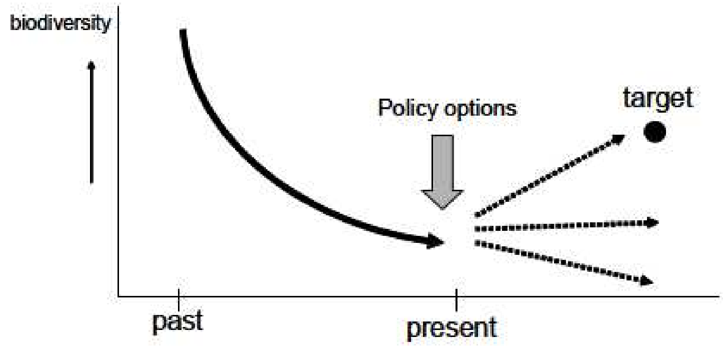
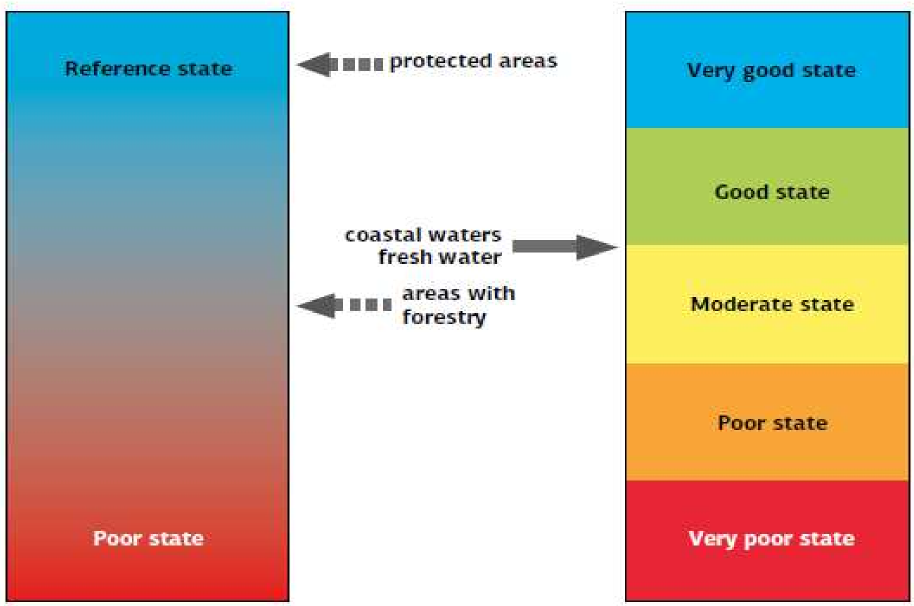


Figure 12. The basic elements of effective biodiversity management. (ten Brink 2006)

In the European Water Framework Directive the ambition is that all water resources in Europe shall have what is characterized as good state in terms of physical and biological indicators by 2015 (calibrated among the European countries in 2008).



Figur 13. Example of possible policy targets for forest and water

The European Water Framework Directive clearly illustrates the difference between reference value (base line) and policy target. While the reference value is the “Very good state” indicated in Figure 13, it is recognized by policy makers that this water quality may be unattainable many places, and that the policy target should be to aim for a water quality at least reaching the “Good state”, corresponding to a policy target of at least 0.6 on a scale from zero to one. In the Nature Index, a similar procedure is suggested to distinguish between reference values and policy targets. For a major ecosystem like forest, including protected areas as well as areas with intensive forestry, society may accept different policy targets for biodiversity in different forest areas. In Norway, the average national Nature Index for forest is 0.43. A forest biodiversity target close to 0.5 may be acceptable for forest areas where forestry has reduced biodiversity dramatically, while a much higher policy target, say 0.9, may be appropriate for protected forest areas.

Accounting with a measure of critical values for biodiversity facilitates the setting of limits for a 'safe operating distance' for biodiversity critical values and identifies thresholds for policy targets. One of the major policy failings for biodiversity is that the 'costs of changes in biodiversity have historically not been factored into decision making' [6]. Whilst there is continuing discussion on economic valuation of biodiversity, these 'costs' can just as equally well be quantified by changes to biodiversity itself.

Aligning economic and biodiversity accounts will enable policy makers to consider the interaction between biodiversity condition and economic indicators, beyond monetary valuation. With this spatially explicit information we can analyse how to improve or maintain biodiversity at the least cost to the economy (or perhaps even vice versa). Accounting for biodiversity will provide a unit to measure 'biodiversity return' for our investment and hence inform overall investment and intervention.

The costs of declining biodiversity is largely borne by society, and not taken into account by individual decision makers (MEA 2005).[Until these costs are distributed for payment by individual decision makers, costs will continue to be borne by society, as management, repair and mitigation. In this case, society is paying for biodiversity return, not an economic return. Stock accounts can reveal these transactions as consumption or depletion.

8.3 Biodiversity targets as part of adaptive planning and policy processes

Establishing policy targets will have to be based on a participatory process. These processes may however be passive or active in their use of indicators. In a passive process the use of indicators is restricted to establish trends and then discuss policy action. In an active process forecasting changes is a necessary part of the policy process.

Passive use of the biodiversity index:

Active use of the biodiversity index:

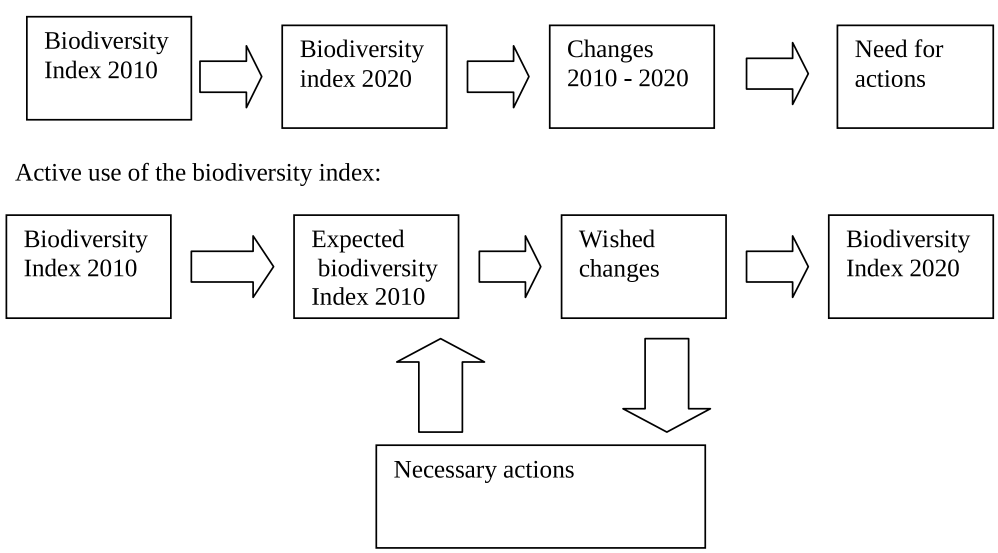


Figure 14. Passive and active use of the biodiversity indicators.

Improving biodiversity measurement as basis for ecosystem and biodiversity accounting may be attained in several ways. The knowledge basis for enhanced biodiversity policy needs to take into account biodiversity measurement combined with trends in economic and social development, in order to facilitate a mix of policies that may achieve the results wanted. The UK National Ecosystem Assessment is an example of a scenario-analysis of future biodiversity policies. In the Nature Index for Norway, experts were asked about their assessments of future biodiversity trends, consequences of biodiversity loss, and the need for urgent management action (Aslaksen et al. 2012).

International comparison is an important goal for biodiversity measurement, in accordance with the Convention on Biological Diversity and other international agreements. It is therefore of great importance to coordinate various national initiatives, based on underlying methods that are not totally identical, in order to develop an internationally applicable framework for integrated ecosystem and biodiversity accounting. Based on several of the most widely accepted methods for integrated biodiversity measurement, The Nature Index has a comprehensive and flexible structure that represents a large potential for contributing to the core element of an international standard for ecosystem and biodiversity accounting in ecological terms.

The biodiversity accounting framework outlined here has the potential to serve as a diagnostic account in order to improve the knowledge basis for biodiversity policy.

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Appendix 1.
Other multidimensional composite indicators, resilience and ecosystem conditions

Based on McDonald (2011):

Indicators of ecosystem condition measure the outputs of a system in order to assess the overall function of that system. Biodiversity is one of those outputs that can be measured to gauge the condition of the ecosystem of which it is part.

This aspect of biodiversity accounting is the focus of the work of Weber [36] and outlined in the experimental framework for ecosystem capital accounting in Europe, and methods are being developed as part of the Australian Environmental Accounts model based on ecosystem condition [8, 9, 36]. However, it might be helpful to review the science from a biodiversity perspective.

Since it was recognised that measuring and describing the condition of ecosystems is critical to quantifying the state of an ecosystem, to understanding the extent of human impact, evaluating the effectiveness of conservation actions [37] and estimating the quantum of services provided by ecosystems, there have been several useful developments in the definition of ecosystem condition and methods for measurement.

Evolving from the concept of stress ecology, Rapport et al [38]identified ecosystem 'health' as a measure of 'system organisation, resilience and vigour as the absence of signs of ecosystem distress'[38]. To operationalise these concepts they recommend the following list of broad indicators to measure ecosystem health:

- vigour, which refers to the level of activity, metabolism or primary productivity;
- organisation, which refers to the structure or number of interactions within an ecosystem; and
- resilience [39], which refers to an ecosystem's ability to recover following disturbance.

It is well know that it is difficult to measure all the components of an environmental asset. The ecosystem or biological system that make them up are far too complex. For these reasons, an integrated approach involving several indicators may be best [41]. Presently several types of frameworks are under development (UNEP 2011).

Biodiversity indicators
Components of biodiversity
Trends in extent of selected biomes, ecosystems, habitats
Trends in abundance of selected species
Coverage of protected areas
Changes in status of threatened species
Trends in genetic diversity
Sustainable use
Area under sustainable management
Proportion of products from sustainable sources
Ecological footprint and related concepts
Threats to biodiversity
Nitrogen deposition
Trends in invasive alien species
Ecosystem integrity, goods and services
Marine Trophic Index
Water quality of freshwater ecosystems
Trophic integrity of other ecosystems
Connectivity/fragmentation of ecosystems
Human-induced ecosystem failure
Health and well-being of communities
Biodiversity for food and medicine
Status of knowledge, innovations, and practices
Linguistic diversity
Indigenous and traditional knowledge
Status of access and benefits sharing
Access and benefits sharing
Status of resource transfers
Official development assistance
Technology transfer

Table 1. Current development of the headline biodiversity indicators within the CBD framework. ■ Fully developed with well-established methodologies and global time-series data, ■ under development, and ■ not being developed. Multiple labels indicate multiple measures under each headline. See also SOM and 2010 Biodiversity Indicators Partnership, www.twentyten.net.

Ecosystem health. (based on Cosier 2011)

An interesting approach that synthesise several components of and concentrate them into one composite indicator might be the solution. An index measuring the overall ecosystem condition. Measuring how the present condition deviates from the natural or potential of an ecosystem in the absence of significant human alteration (Cosier 2011):

"Applying a reference condition benchmark performs the essential function of allowing different landscapes to be measured with indicators that are specifically suited to a particular location. This avoids having to use one set of indicators for distinctly different landscapes.

The advantages of such a benchmark metric are that:

- it creates a common environmental currency that allows us to evaluate the relative environmental improvement of one action over another from investments we are making; and
- they drive cost efficiencies in data collection, because they allow areas under intense environmental pressures to be measured with greater precision than areas under less pressure, without diminishing the ability to compare one asset or region with another.

- a sand dune with a river; an estuary with a rainforest, or one river system with another.

- An environmental health index can be generated by selecting a range of indicators that, when combined, best describe the condition of that environmental asset at a particular location.

- These environmental health indices can be used to create the common measure of condition for each environmental asset. This allows any asset to be compared relative to a similar asset at any location; it allows us to compare the rate of change between different assets, and it enables this information to be aggregated to produce environmental accounts at a range of spatial scales.

- To avoid confusion with the condition score of an individual indicator, each environmental health index could be referred to as an ***ECOND***.

- An ***ECOND*** is a scientifically accredited measure, metric or model which reflects the health of an environmental asset, and is created by combining (where appropriate) condition scores of environmental indicators based on a reference condition benchmark.

- The ***Econd*** describes the common environmental currency, in the same way a dollar ($) describes a financial currency."

The Econd will be differently calculated for different biomes. For terrestrial ecosystem Cosier proposes a combination of biodiversity, water conditions, soil quality and land use. No weighting procedures are adviced

The problem of such complex composite indicators is that one does not know how different parts of the index dominate other parts. The trade offs become very difficult to examine. The reasons for differences in Econd- values between one region an another may become difficult to explain.

Bundles of composite indicators.

Complex composite indicators are of course interesting- also in the discussion of biodiversity. But mostly they are interesting for promoting understanding and discussions. They focus on the political agenda. The diagrams below is showing three interesting composite indicators, one for biodiversity, one for pressure and one showing responses (Butchart et al 2010) (***discussion of data)***

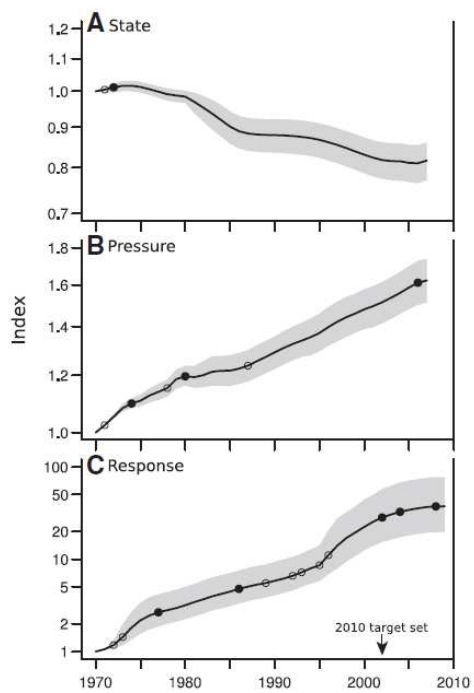
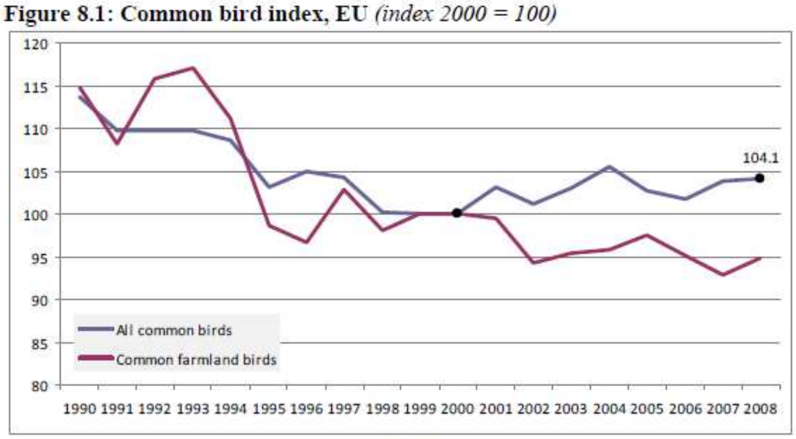


Fig. 2. Aggregated indices of (**A**) the state of biodiversity based on nine indicators of species' population trends, habitat extent and condition, and community composition; (**B**) pressures on biodiversity based on five indicators of ecological footprint, nitrogen deposition, numbers of alien species, overexploitation, and climatic impacts; and (**C**) responses for biodiversity based on six indicators of protected area extent and biodiversity coverage, policy responses to invasive alien species, sustainable forest management, and biodiversity-related aid. Values in 1970 set to 1. Shading shows 95% confidence intervals derived from 1000 bootstraps. Significant positive/upward (open circles) and negative/downward (filled circles) inflections are indicated.

Appendix 2. Biodiversity indices

The Bird-index is one of the traditional biodiversity indicators used first in GB and now in all Europe. The indicator has been included in the standard list of Eurostat's sustainability indicators. By now it gives a picture for 19 EU-member countries and show the development both for "common" and "farmland" birds.

Figure 8.1: Common bird index, EU *(index 2000 = 100)*



Period evaluated: 2000-2008 (all common birds)

Average annual rates of change:

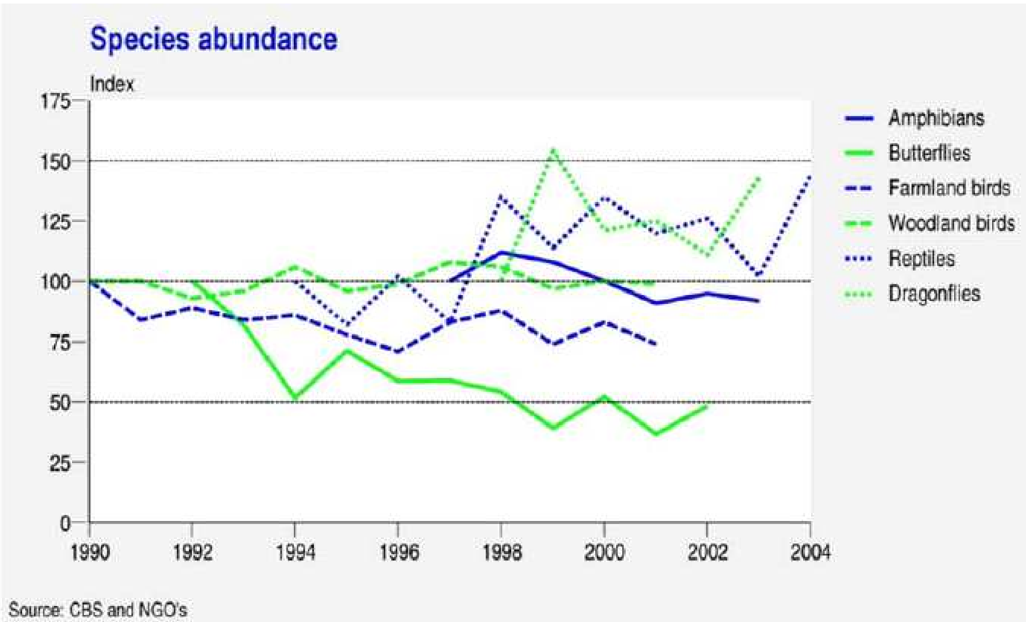
All common birds: +0.5 %

Common farmland birds: -0.7 %

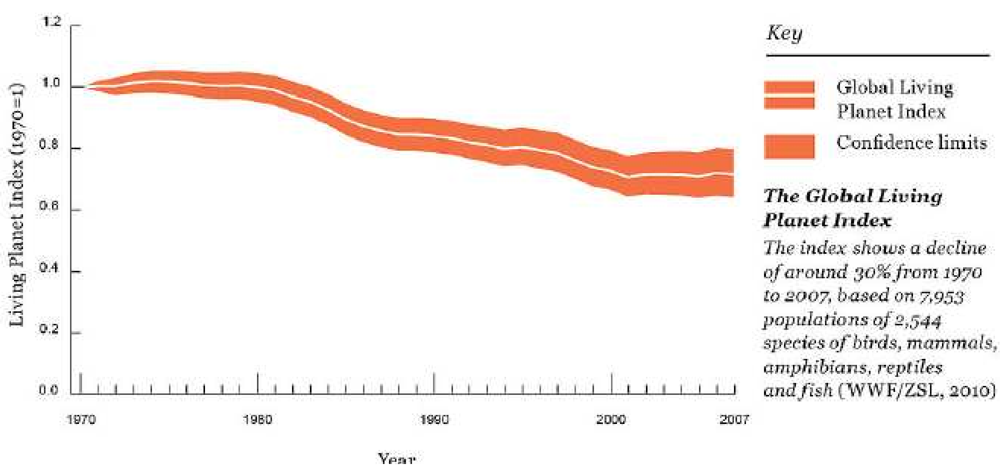
Source: Eurostat (online data code: tsdnr100)

NB: The EU aggregate is an estimate based on 19 Member States: BE, BG, CZ, DK, DE, EE, IE, ES, FR, IT, LV, HU, NL, AT, PL, PT, FI, SE and UK.

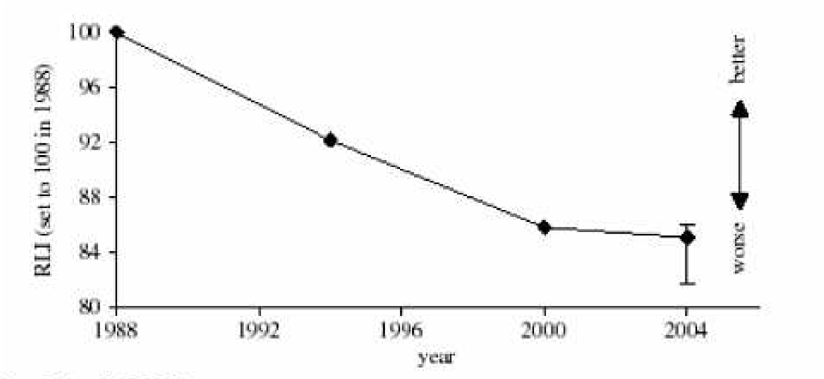
Species Assemblage Trend Index is the mean species abundance group compared to a reference year. Trhese could be different group of species, taxonomic groups, endemic species or threathened species.The asssessment principle is more individuals the better.



The living planet index is based on time series that if necessary are extrapolated to the base year 1970. The principle is more individuals the better are conditions .All species contribute equally in the calculations of the index. The Living Planet Index was originally developed by WWF in collaboration with UNEP-WCMC. Between 1970 and 2007, the index fell by 28%. In the later years the trend has levelled off.



The Red List Index measures species extinction risk by weighting the extinction risk of all species of a particular taxonomic group. In the figure below RLI is calculated for all birds, for the period 1988-2004. The RLI probabilities are made by expert judgement, as species moves from one class to another in terms of extinction threat.



Source: Butchard et. al. (2005).

Statistical measures

Species richness and evenness are often used to describe an ecosystem. Richness is the total number found in an environment, and evenness describes how like the population figures are for the different species.

Simpson's index (D) is the probability that two randomly selected individuals belong to two different species/categories

There are two versions of the formula for calculating **D**. Either is acceptable, but be consistent.

$D = \sum (n / N)^2$	$D = \frac{\sum n(n-1)}{N(N-1)}$
n = the total number of organisms of a particular species **N = the total number of organisms of all species**	

The value of **D** ranges between 0 and 1

With this index, 0 represents infinite diversity and 1, no diversity. That is, the bigger the value of D, the lower the diversity. This is neither intuitive nor logical, so to get over this problem, D is often subtracted from 1 to give:

Simpson's Index of Diversity 1 - **D**

Shannon-Wiener index (H) is measuring the order/disorder in a particular system. This order is characterized by the number of individuals found for each species/category in the sample. A high species diversity may indicate a healthy environment. Actually this is close to other measures of entropy.

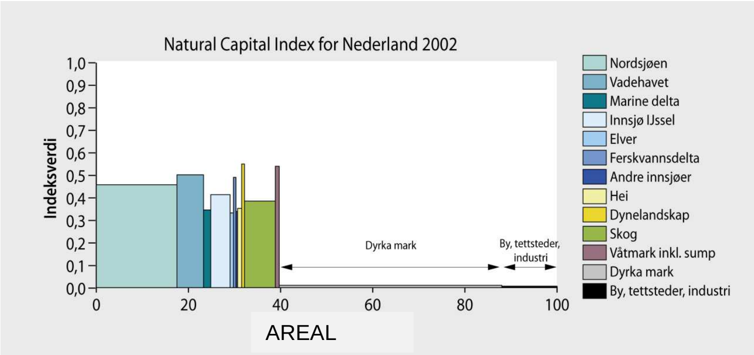
H = sum over i =1 to s of   Pi log(Pi)

Where s is the number of species and Pi is the proportion of population in I compared to the sum of all the populations.

Natural Capital Index and the Mean Species Abundance.

Both the NCI and MSA measures species abundance relative to low-impacted or pre-industrial state. NCI have both been calculated both based on sample of species and by models based on land use. Bothe indices are calculated as the product of the remaining ecosystem area (quantity) and the ecosystem quality (mean species abundance in the remaining ecosystem). In NCI there is a distinction between natural (self generating) and cultural ecosystems. Traditional landscapes may have their own specific biodiversity values which are higher that the natural systems.

MSA is model based designed to calculate past, present and future for different scenarios. The model is based on pressure factors from human activities impacting on different land use and physical characteristics. The model has been used to present a number of international studies Global Biodiversity Outlook, regional assessments of UNEP, OECD’s Environment Outlook etc)



The Index of Biotic Integrity reflects fish species richness, number and abundance of indicator species, trophic function and organisation, reproductive behaviour and some other fish information. In total the metric contained about 12 types of Information (Karr 1981). This metric was used to classify river part in different quality classes based on expert judgements.

The sustainable Rivers Index “is designed to represent functional and structural links between ecosystem components, biophysical condition and human intervention.” Through sampling and/or modelling indicators in the areas of hydrology, fish, macro vertebrates, vegetation and physical form were calculated. Reference conditions were no significant human intervention in the landscape.

Both these indexes are typical regional and closely linked to regional watershed management.

The Marine Trophic index have really been one of the most used indicators in the marine environment used both by FAO and CBD **.**MTI is the mean trophic level of fish landings. Trophic levels measures the position of species in the food chain, starting out with the primary producers and ending up with 2-5 top-predators. It is mathematically a calculation of replacement indices and may be used to describe the interactions between fisheries and marine ecosystems. Lower MTI-values over time are a signal of "fishing " down the food chain.

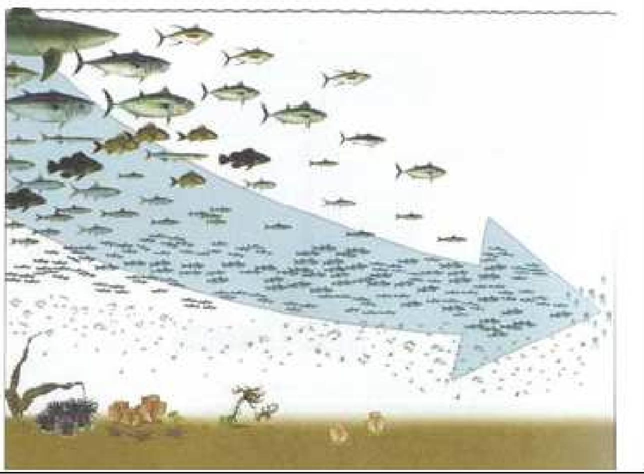


Figure xx. Fishing dow the food-chain (Pauly 1988).

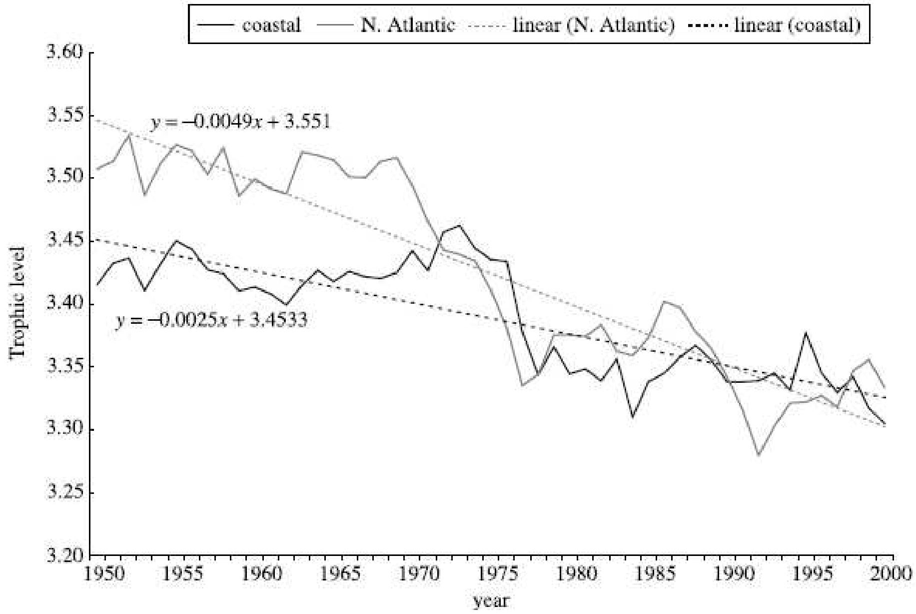


Figure xx. Trends in meantrophic levels of fisheries landings, 1950-2000. Watson et al (2004)

The Water Quality Index (The Water Framework directive EU)
The WFD has been in force since 22 December 2000. Its purpose is to establish a framework to protect all waters (inland surface waters, transitional waters, coastal waters and groundwater). The Water Framework Directive deals with three central elements of the aquatic ecosystem: the quality of water, the quantity of water and the aquatic habitat. Reference conditions are described as close to undisturbed conditions. Typology, reference conditions and classification systems were developed for the three ecosystem types Classification rules and guidance documentation were established explaining for instance to how to use physico-chemical parameters for ecological classification and how to deal with uncertainty. In addition to the selection of indicator species the water quality index system was calibrated among the European countries in 2008.

The Biodiversity Intactness Index (BII) was constructed for data-poor regions like that of southern Africa. It is calculated from land use and land cover data. Reference is naturalness as observed in national parks. Impact assessment were made by structured interviews of 16 taxon specialists. BII is an aggregated index that may be disaggregated in terms of taxa, ecosystems and land-uses. Past trends may be calculated as well as future projections and it may be shown with an error bar indicator the uncertainty. Biggs (2005); Kirton (2008)

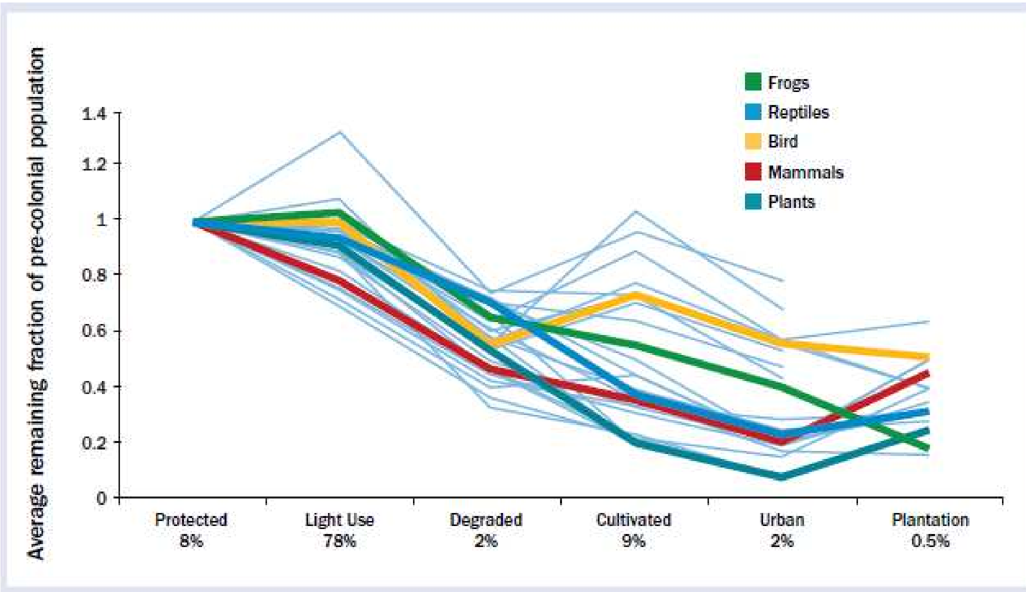


Figure xx The effect of increasing land use intensity on the inferred original population for land-use categories in southern Africa. Scholes and Biggs (2004).