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Dit rapport geeft een overzicht van de aanpakken en methodes waarmee je de impact van menselijk handelen op de biodiversiteit kunt meten. Daarbij zijn nog veel uitdagingen te overwinnen. Ten eerste is biodiversiteit een begrip dat moeilijk te vangen is in één indicator, omdat het een begrip met meerdere dimensies is. Ten tweede is het monitoren van biodiversiteit vaak kostbaar en een uitdaging. Ten derde zijn de causale relaties tussen menselijk handelen en biodiversiteit ook onbekend. Dat heeft invloed op de resultaten van de impactmodellen voor biodiversiteit. Ten vierde is er geen consensus over hoe (de waarde) van biodiversiteit meegenomen zou moeten worden in de besluitvorming van bedrijven of overheden. Met biodiversiteitsaccounting wordt geëxperimenteerd, maar de methodes moeten nog verder ontwikkeld worden om er een gangbare methode van te maken. Dit rapport besluit met een aantal aanbevelingen voor toekomstig onderzoek en beleid.

This report provides an overview of the approaches and methods of measuring the impact of human activities on biodiversity. Measuring these impact still has numerous challenges. First, biodiversity is hard to capture with one single indicator, because it has many dimensions. Second, monitoring biodiversity is often expensive and remains a challenge. Third, the causal relationships between human activities and biodiversity are often unknown. Thus, the result of current impact models for biodiversity are based on many assumptions. Fourth, there is no consensus on how biodiversity should be included in the decision-making of companies and nations. There are experiments with biodiversity accounting at company and nation level but the methods need to be developed further. This report concludes with a number of recommendations for future research and policy.

Key words: biodiversity, biodiversity monitoring, impact of human activities on biodiversity, biodiversity accounting, human activities.

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Preface

In ‘The methodological assessment report on the diverse values and valuation of nature’, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) advocates a shared vision of prosperity for people and the planes including biodiversity. However, there are many challenges to make people, companies and governments more aware about the consequences of human activities on biodiversity in the local, national and global context. First, biodiversity is hard to capture with one single indicator, because it has many dimensions. Second, monitoring biodiversity is often expensive and remains a challenge. Third, the causal relationships between human activities and biodiversity are often unknown. Thus, the result of current impact models for human activities on biodiversity are based on many assumptions. Fourth, there is no consensus on how biodiversity should be included in the decision-making of companies and nations. There are experiments with biodiversity accounting at company and nation level but the methods need to be developed further. This report provides an up-to-date overview of the definition, monitoring, and assessments of biodiversity. Moreover, it gives an overview of the ways that biodiversity can be valuated and incorporated into decision making by governments and companies. Therefore, this report can serve as a guide for researchers and policy makers to get an update of the status of the challenges surrounding biodiversity monitoring and valuation and provides recommendations for future research and policy.

This study has been executed by a multidisciplinary team of Wageningen University & Research including Geert Woltjer, Rolf Michels, Monica van Alphen, Vincent Linderhof (Wageningen Economic Research), Eric Arets, Sven van Baren (Wageningen Environmental Research), Ruud Jongbloed, Gerjan Piet (Wageningen Marine Research) and Iris Vural Gursel (Wageningen Food & Biobased Research). We would like to thank Geert Woltjer, who retired in September 2023 from Wageningen University & Research. This report has been one of his final publications as colleague of Wageningen University & Research. Geert has made valuable contributions to the economic research at Wageningen Economic Research and the international literature in the field of valuing biodiversity.

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Summary

S.1 Main conclusions

1. The inclusion of monetary values for biodiversity in the decentralised decision-making of a market economy requires consistent accounting of biodiversity analogue to financial accounting systems.
2. Although there a large number of biodiversity indicators in use, there is no framework that integrates all relevant aspects of biodiversity consistently. Biodiversity accounting systems used so far are mainly based on ecosystem quality indicators such as the Potentially Disappeared Fraction of Species (PDF), or the Mean Species Abundance (MSA). These indicators do not provide a full picture of what is really important in biodiversity, such as the multi-dimensionality of biodiversity.
3. The assemblance of biodiversity information is usually expensive. However, the use of information technology, remote sensing and citizen science in combination with improved ecological insights might lead to better and cheaper monitoring of biodiversity.
4. Although the logic of the accounting systems is the same, the approaches of terrestrial ecosystem accounting and marine ecosystem accounting differ widely. While terrestrial ecosystem accounting currently has a focus on quantifying a limited set of indicators such as MSA or PDF, marine ecosystem accounting focuses more on describing policy-related indicators and assembling indicators for pressure factors. The challenge is to set up a consistent approach which would fit both.
5. Life cycle accounting (LCA) for products, accounting systems for companies and the national accounting systems relating economic activities with monetary biodiversity values use their own type of causal logic and type of location specific information. There is a need for consistency for all types of analyses of the causal chain from activities to emissions to (midpoint) impact indicators and to (endpoint) indicators that are directly related with human welfare.
6. The biodiversity accounting systems related to the System of Environmental Accounts (SEEA) of the United Nations may potentially provide the local information required for biodiversity accounting. Ecosystem Service accounting from SEEA uses market prices for valuation, while for business application in many cases other valuation approaches are used. In cost-benefit analysis, company biodiversity accounts or biodiversity offset systems, explicit monetary valuation of biodiversity is often challenging. In some examples, such as the Nature Points in the Netherlands, the Biodiversity Metric in the UK or and the Healthy Ecosystem Metric in the UK, biodiversity quality indicators such as PDF or MSA are weighted with the location-specific areas values based upon criteria such as extinction risk and policy importance.
7. The monetary valuation of biodiversity is in many cases mainly based on ecosystem services, although there is a limited relation between ecosystem services and biodiversity indicators. Therefore, the challenge is to integrate the different biodiversity indicators in an expansion of ecosystem service systems definitions.
8. The way in which offsets are implemented is inconsistent, because these offsets have their own rules and criteria. Moreover, many offset schemes do not provide sufficient guarantee for biodiversity compensation.

S.2 Executive summary

Introduction
Biological diversity (biodiversity), defined as variety of living organisms, considers all types of terrestrial, aquatic and marine ecosystems. It includes their genetic diversity, their species diversity and their ecosystem diversity. The protection of biodiversity is increasingly receiving global attention, as it is rapidly declining which threatens the survival of humankind. The 2019 UN IPBES report shows that biodiversity is affected by a wide range of human-related pressures and that many actors need to change their behaviour. To keep track of the status of biodiversity and its interaction with human-related pressures, adequate and consistent accounting for effects of socioeconomic activities on biodiversity is crucial. However, financial and

legal decision-makers lack market incentives to take biodiversity values (monetary or non-monetary) into account in their already very complicated decision-making.

To include biodiversity in market decisions, it would be ideal if prices would be set for biodiversity loss or improvement, so that all decision-makers adjust their behaviour consistent with a sustainable level of biodiversity. However, there is no system yet that adequately prices biodiversity loss or improvement. Therefore, the second-best solution would be to develop accounting systems that make the hidden costs for biodiversity loss visible.

This report discusses the main approaches of analysing the impacts of socioeconomic activities on biodiversity. In particular, it emphasises how those approaches relate to integrating monetary values for biodiversity in the market system. The challenge is to develop a framework of biodiversity measurement and analysis that is suitable to analyse impacts of socioeconomic activities on biodiversity, and that is able to identify opportunities to improve biodiversity by changing production and consumption decisions. This is relevant both on the scale of governments, business and consumers.

Biodiversity indicators
The management of biodiversity in an efficient way requires having good indicators of biodiversity, so that biodiversity can be monitored. However, there have been suggested a wide range of indicators depending on the type of biodiversity, the (geographical) context of biodiversity, the scale of biodiversity from local to global biodiversity and the pressures considered, amongst others. From an analytical point of view, this is unsatisfactory, because these indicators refer to different components in the causal chain from drivers to changes in biodiversity till policy responses. Changes in the state of biodiversity are not represented in a way that they can be directly related with welfare changes. Biodiversity impact methods such as LCA or global biodiversity models use one state indicator that represents the number of lost standardised hectares. In practice, indicators that compare the current species richness to that in the pristine reference of natural biodiversity, such as PDF and MSA, are most widely used in impact assessments. They make it possible to relate drivers and pressures to the selected biodiversity indicator, but it is impossible to relate the biodiversity indicator with its impacts on welfare. Meanwhile, the marine approach still tries to come up with a coherent framework for biodiversity indicators, which are sufficiently related with policy objectives, and a useful assessment of the state of marine ecosystems. However, the framework in the marine approach is only qualitative and is wrestling with defining relevant aspects of biodiversity and relating them to welfare.

Biodiversity monitoring
If biodiversity indicators are defined, the development of those indicators can be monitored, which can be costly. The exact definition of the biodiversity indicator to be monitored depends on the context. It could be either aspects of species and their environment, such as a population, the density of a population, the spatial distribution of species; the ecological dose-response function that helps to understand how a pressure influences biodiversity; or the effectiveness of different types of management actions on biodiversity conservation. However, with the rapid developments in information and communications technology, more novel opportunities to monitor biodiversity have been developed, such as for example remote sensing, remote cameras, drones, Light Detection And Ranging (LIDAR), automated acoustic sampling, and environmental DNA (eDNA). Another development over the years is the involvement of volunteers in biodiversity monitoring such as citizen science, which increases the capacity of collecting biodiversity data. Most likely, the data of these groups of volunteers will be biased towards for example more interesting or less known species, because only few citizens register and assemble biodiversity information systematically.

Assessing biodiversity impacts caused by socioeconomic activities
Biodiversity impact assessments of specific products, specific companies or specific policies require causal analyses. A Life Cycle Analysis (LCA) uses such a causal analysis to allocate environmental effects of activities in the whole value chain to products or activities of companies. The global biodiversity impact assessment model GLOBIO and the LCA method ReCiPe apply dose-response relations between pressure factors and changes in biodiversity. For both models it is hard to trace the sources of empirical information on which the causal relations are based. GLOBIO and LCA both use an indicator that describes biodiversity loss as a change in standardised hectares, where the indicator describes biodiversity quality in the current system as share of that of the original ecosystem. LC-IMPACT is weighting the lost hectares by the

importance of these hectares for the protection of species on the IUCN Red List of Threatened Species and therefore puts local species loss in the context of global species loss. In the marine environment, the so-called cumulative impact assessment (CIA) is used. It describes all potential impacts on policy goals for biodiversity in the marine environment. CIA tries to capture all pressures and the multidimensionality of biodiversity by using many indicators but is not yet able to quantify the relations and indicators.

All approaches are still searching for opportunities to make the relations more spatially explicit. To improve the approaches, the empirical foundation of the dose-response functions needs to become explicit. Finally, while it is essential that all efforts focus on explaining the consequences for welfare in the broad definition of it, the current methods are still far from this goal.

UN system of ecosystem and biodiversity accounts
The United Nations biodiversity and ecosystem accounting systems has been developed in addition to the System of National Accounts (SNA) and the System of Environmental Economic Accounts (SEEA) for understanding the economic system and considering the location-specific aspects of biodiversity as well. Biodiversity accounts report changes in biodiversity in a systematic way. This can be at the level of specific species, the share of species surviving compared with a reference point for a specific ecosystem, or the abundance of specific species such as the species on the IUCN Red List of Threatened Species, which may be aggregated to one indicator for biodiversity. Some biodiversity indicators, such as the MSA, have the property to be additive, but others are not, such as the number of species in specific categories of the Red List of Threatened Species. At this moment, the biodiversity accounts are not directly related with the SEEA or SNA, which implies that they can neither be directly related with the functioning of ecosystems nor with the impacts on human welfare through ecosystem services. Furthermore, the information assembled by the UN SEEA may potentially be relevant as input for an LCA type of analysis and global impact evaluations.

Biodiversity accounting for business
One of the drivers for reversing biodiversity loss would be that companies take either monetary or non-monetary biodiversity values into account in their decision-making. A first step would be that they include monetary values in their accounting systems. In current approaches, biodiversity indicators related with the quality of ecosystems, such as PDF and MSA, are most used, but they do not measure all relevant aspects of biodiversity, such as the diversity in ecosystem types and extinction of species. Many approaches in business and policy therefore weigh ecosystem quality with the importance of the ecosystem for specific goals. However, these goals are different for different approaches, which implies that the evaluations are not consistent with each other and therefore will not result in optimal decision-making from a societal welfare point of view.

Companies conduct biodiversity accounting for various reasons, such as legal requirements, reputation management or anticipating future regulations. These motivations do not guarantee that these accounts provide sufficient information to tackle the current biodiversity challenges.

For financial institutions, biodiversity accounting is not fundamentally different than for other companies. However, financial institutions have two complementary aspects, which are key here. First, financial institutions may be interested more in the financial risks of changes in natural capital than other companies, because this determines their risk portfolio. Second, the influence of financial institutions on decisions that are relevant for biodiversity is more indirect than for most other companies. This complicates allocating the responsibility for the biodiversity consequences of their investment portfolios to financial institutions.

Biodiversity valuation
To integrate biodiversity in financial decision-making, it is useful to monetise biodiversity changes. This is very difficult, though. In the Handbook of Environmental Prices (De Bruyn et al. 2018b), the PDF indicator is priced based on a meta-analysis that relates PDF in plots of land to the value of ecosystem services provided. However, the relationship between the PDF indicator and ecosystem services is unclear and ambiguous, which makes it unlikely that the estimation of the price per unit PDF is very accurate and reliable. To improve accuracy, more knowledge on causal relationships of pressures and biodiversity is required.

An alternative approach is to use information from the costs needed to compensate biodiversity loss resulting from economic activities at a specific place with biodiversity improvements elsewhere, the so-called biodiversity offsets. However, these offsets are implemented in varies ways and under varies conditions. Many offset examples are criticised for giving insufficient guarantee to compensate the biodiversity damage accurately. Therefore, for accurate offsets programmes, more information on monetary values of the offsets and bio-banking is required.

Conclusions

There is still a long way to go before either monetary and non-monetary biodiversity values will be integrated in decision-making on all levels. Including biodiversity values in the decentralised decision-making of a market economy requires consistent accounting of biodiversity, which is analogue to financial accounting systems. There are a large number of biodiversity indicators, out of which the ecosystem quality indicators MSA and PDF are the most frequently used ones, but the relationship among these indicators and between the biodiversity indicators and the value of ecosystems provided is not sufficiently developed yet. This observation or conclusion is relevant for ecosystem accounting related with national income accounts and input-output accounts, and for calculating and evaluating biodiversity footprints for companies and products. All these systems need analyses of the causal chain from activities to emissions to (midpoint) impact indicators and to (endpoint) indicators that are directly related with human welfare.

1 Introduction

1.1 Context

Biological diversity (biodiversity) is defined as the variety of living organisms. It considers all types of terrestrial and aquatic ecosystems and includes their genetic diversity, their species diversity and their ecosystem diversity. The protection of biodiversity is increasingly receiving global attention. The 2019 UN IPBES report (IPBES 2019) argues that biodiversity is rapidly declining and this threatens the survival of humankind, see Figure 1.1. It shows that biodiversity is affected by a wide range of pressures such as land use change, soil degradation, climate change, water and air pollution, water stress, overexploitation by hunting or fishing and invasive alien species. The conclusions of this UN IBPES report were endorsed by 132 countries, showing the common recognition and interest in tackling this issue. Research shows that the loss of biodiversity constitutes a high risk to humanity because of loss of ecosystem services essential for human life and the risk of reaching turning points (SRC 2023). Examples of ecosystem services are climate regulation, cleaning air, food production, freshwater provision, protection from floods and soil erosion, generation of cultural and educational values, and recreation.

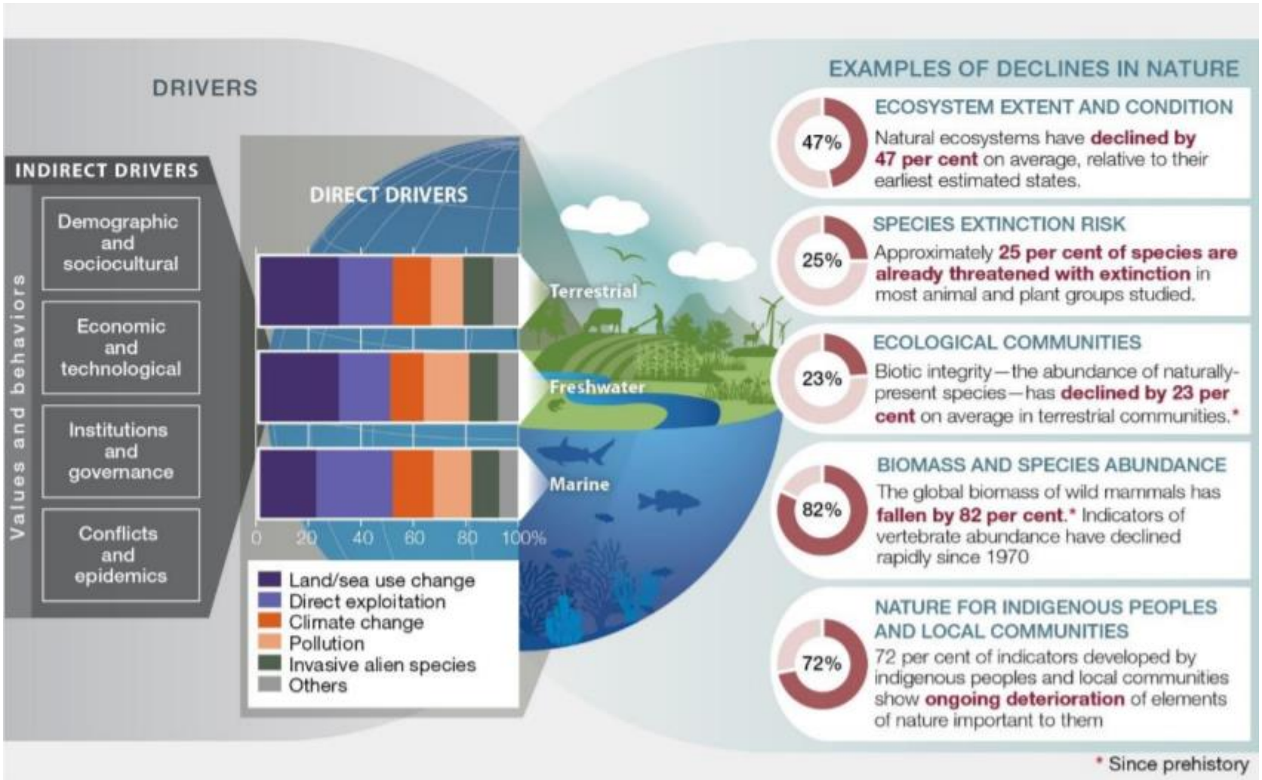

Figure 1.1 *Examples of global declines in nature, emphasising declines in biodiversity, that have been and are being caused by direct and indirect drivers of change*
Source: IPBES (2019).

The current global economy is based on capitalistic decision-making, where decisions of production and consumption of commodities (goods and services) are driven and motivated by market prices. However, many of those market decisions cause pressures on biodiversity which are not included in market prices; so-called externalities. As a consequence, impacts on biodiversity are not or insufficiently considered in market decisions. Even if consumers or companies making decisions have good intentions, the market system does not generate sufficient information on the externalities. However, even if sufficient information on externalities would be available, consumer and companies will try to consume at the lowest price, and to produce to lowest cost, unless there are incentives to take into account the externalities, see Textbox for further explanation. Companies that make extra costs for biodiversity protection will not be able to survive in the highly competitive market system, while many consumers cannot afford to buy more sustainable but higher priced products. Moreover, governments tend to ignore biodiversity in their decisions on regulation and production, because the costs of biodiversity protection are not visible in the prices. In summary, financial and legal decision-makers lack incentives to take biodiversity values into account in their already very complicated decision-making.

The current market system is still far away from integrating the monetary value of biodiversity loss or improvement in market prices. To inform policies that help to protect biodiversity values, an accounting system will be helpful. IPBES (2019) states that

> ‘Alternative models and measures of economic welfare (such as inclusive wealth accounting, natural capital accounting and degrowth models) are increasingly considered as possible approaches to balancing economic growth and the conservation of nature and its contributions and to recognising trade-offs, the pluralism of values, and long-term goals.’ (p. 33).

On a national level, accounting systems are set up to inform government policies, but more and more companies start to include biodiversity values into their accounts. The financial sector in the Netherlands, for instance, has developed a methodology to assess the biodiversity impacts of their portfolio of investments (CREM and Pré Sustainability 2019). However, the number of accounting systems and implicit relative biodiversity values is very large (Lammerant et al. 2019), where one may doubt to what extent the accounting systems really measure what is relevant to protect biodiversity (Woltjer and Michels 2021).

Textbox: Biodiversity incorporated in a neo-classical economy

From the perspective of the neo-classical theory (companies maximising profits and consumers maximising utility), it would be optimal to set prices for biodiversity loss or improvement to include biodiversity in market decisions. In this way, all decision-makers will adjust their behaviour in correspondence to a sustainable level of biodiversity. For optimal efficiency, all decision-makers should face the same prices for the same effect on biodiversity, as this leads to the fact that actors with the lowest costs to reduce biodiversity loss or improve biodiversity, will do so. The result is the highest level of biodiversity at the lowest cost. By integrating a consistent price for biodiversity into the capitalistic market system, sustainable alternatives with low biodiversity loss become relatively cheaper, and therefore people will buy more of these products, stimulating companies to become more sustainable.

However, by making sustainable production methods and products more profitable, R&D and learning by doing will be focused on more sustainable technologies, which makes sustainability cheaper in the future. For example, prices of wind turbines and solar panels for producing electricity that declined very fast in price just because there was a rapidly increasing demand for these technologies. Finally, many decisions by both organisations and individuals are guided by routines, social norms and values, and just by making sustainable alternatives cheaper, there will be changes of these routines, social norms and values will change too.

1.2 Research question

Given the fact that we live in a capitalistic, market-based society, many decisions are determined by comparing private costs and benefits. Since there is no market for biodiversity, it does not have a price, so that biodiversity is often ignored in the current economy. There is a need for finding approaches to include these biodiversity values – either monetary or non-monetary - into decision-making. For the implementation of biodiversity into economic decision making, there are a number of steps required:
1. Determine the (monetary and non-monetary) values of biodiversity
2. Create an overview of the causal impact chains from socioeconomic activities and decisions through pressures to biodiversity
3. To integrate biodiversity impact into decision-making in our current economic system, there is a need for consistent approaches to estimate the monetary values of biodiversity accurately.

The goal of this report is to provide insights into the main approaches to analyse the impact of socioeconomic activities on biodiversity, how these different approaches are related to each other, and to assess opportunities to assign monetary values to biodiversity impacts for inclusion in decision-making. It presents an in-depth overview of the different approaches and relates them to each other and to the issue of monetary valuation of biodiversity. The ultimate step is to integrate all these approaches into a consistent framework with institutions and prices consistent with the protection of biodiversity values. The current report provides building blocks for this integration steps:

The first building block of the analysis examines the elements of biodiversity and the indicators used to measure them, distinguishing between state indicators and indicators used for the causal impact of economic activities on biodiversity. To use these indicators for biodiversity valuation or in an impact assessment, the indicators must be measured. Ways of monitoring state indicators of biodiversity are considered, taking into account how this can be done in a cost-effective way.

The second building block of the analysis focuses on biodiversity impact assessment methods. These methods assess how decisions such as the purchase and production of products cause changes in biodiversity. To understand the empirical foundation of the parameters used in biodiversity impact assessment methods, one has to dig into the background literature. In particular, to relate different parts of the impact chain, in-depth research of these methods is needed, and the results of this in-depth research are documented in this report. Therefore, in this report each approach is described carefully, focusing on the empirical information and logic that feeds the method. This implies that after describing to describe the method, the literature behind it is described in combination with literature that extends or deepens the insights.

The third building block of the analysis is about national biodiversity accounting as part of decision-making. Biodiversity accounting systems are under development on a national scale to be related to economic national accounts, environmental accounts and the recently developed ecosystem accounts. It is not automatically the case that causalities and impact relationships are completely embedded in these accounts, but for example in the environmental accounts attempts are made to attribute pollution, land use, water use to economic activities.

The accounting systems are primarily meant as a source of information for government policies, but the information contained in them can potentially be used for biodiversity impact assessments. On the other hand, information on causal relationships in impact assessments could potentially be used to improve the national accounting systems. Given the interrelationship between the three accounting systems, the analysis goes beyond only the biodiversity accounts; the logic of all three systems together has been analysed in this report.

The fourth building block deals with the application of biodiversity accounting measures in business and finance. In these applications, adjustments and extensions are made guided by practical issues that arise when the methods actually need to be used. It is interesting to discuss these refinements and adjustments because they may have consequences for the development of methods that are broadly and consistently applicable.

The last building block is about allocating monetary values to the assessment of biodiversity. A method to use outputs from biodiversity impact methods for pricing biodiversity loss is useful to investigate. Another approach is to search for market information on biodiversity. Biodiversity offset markets seem to be obvious candidates to get information on the value of biodiversity, but in the current situation, these markets are insufficiently related to real biodiversity values to be suitable for this purpose.

1.3 Report outline

This report is structured according to the building blocks as defined in the previous section. Chapter 2 presents the aspects of defining biodiversity, discussing the indicators developed to measure biodiversity and the approaches for monitoring biodiversity. Biodiversity is a complex and not very well-defined concept and many indicators are used to describe biodiversity decline. Two recent science-based reports that ring the alarm bell about biodiversity decline are investigated on the indicators that are used to underpin the alarming trends. The more than fifty state indicators used in this type of report are in sharp contrast with the single indicator approaches used in biodiversity models that relate human activities to biodiversity. Further information see Chapter 2.

Chapter 3 provides two examples illustrating the application of using indicators and biodiversity monitoring in policy. The first example is the definition of biodiversity indicators for the Marine Strategy Framework Directive (MSFD). The second example is the description of the monitoring of nature policy in the Netherlands with a main focus on the terrestrial nature. Further information see Chapter 3.

The focus of Chapter 4 is on biodiversity impact assessments, and this chapter discusses in detail the inclusion of biodiversity in Life Cycle Analyses (LCA) and biodiversity models such as GLOBIO that have been developed for international policy evaluation but can be applied within the context of LCA. These analyses are frequently used in business to investigate the footprints of products, and biodiversity is an important aspect of this environmental footprint. However, biodiversity is very complex, and it requires many steps across the value chain of production to determine the impact on biodiversity. As a consequence, biodiversity in current LCAs is either excluded or analysed in an over-simplified manner. In the main approach, ReCiPe, biodiversity is measured with one indicator, PDF, which is explained by a number of pressure factors such as land use, acidification and climate change. It is, however, not easy to trace from which data the parameters for these equations have been derived, and therefore these causal pathways in ReCiPe are described in depth. The same is done as far as possible for the GLOBIO 4 biodiversity model designed for international policy evaluation, but sometimes integrated in LCA analysis. Further information see Chapter 4.

Chapter 5 dives into national biodiversity accounting. National biodiversity accounting tries to relate changes in biodiversity to economic activities on the one hand and changes in ecosystem services to economic activities on the other hand. To understand the role of biodiversity accounting, one needs to understand the System of National Accounts, which is an accounting system for economic activities, the environmental extension of the system in the core framework of SEEA, the specific problem of ecosystem accounting in this context, and the role of biodiversity accounting within this system. All these components are described in this chapter. Furthermore, the chapter tries to relate LCA, national accounting and monitoring of biodiversity. Since both the national biodiversity accounting and LCA attempt to relate economic activities to biodiversity, the fundamental issues are the same. However, in general the two branches of analysis are not well-connected. For further information, see Chapter 5.

Chapter 6 dives deeper into the biodiversity accounting approaches in practice, with an emphasis on business and the financial sector. In some cases, only the ReCiPe method of LCA is used, but in many cases, methods are used that assign weights to different ecosystems based on a number of ad hoc criteria that differ between the different approaches. For further information, see Chapter 6.

Chapter 7 addresses the economic valuation of biodiversity. Many of the environmental pressures calculated in LCA are based on a price for biodiversity, and therefore this price is of fundamental importance for the prices in the handbook. However, this price is based on the value of ecosystem services instead of the direct effects of the biodiversity indicator, and the tension between these two is discussed in Chapter 7. As an alternative approach, the extent to which payments in compensation, offset schemes and biodiversity banking can be used to value biodiversity is being explored. For further information, see Chapter 7.

Chapter 8 concludes and suggests recommendations to continue the development of monitoring and assessing of biodiversity and its monetary and non-monetary values for the inclusion in economic decision making. For further information, see Chapter 8.

In Appendix 1, there is a list of acronyms used in this report. Appendix 2 present two lists of databases and models mentioned in the report but not further explained in the report.

2 Biodiversity indicators and monitoring

2.1 Introduction

Decision-makers need adequate knowledge of the status of and trends in biodiversity, of the impact of the main drivers and pressures that determine biodiversity loss, and of the success, or lack of success, of policies and practices designed to conserve or restore biodiversity. The reasons to assess and monitor biodiversity loss include (Puerta-Piñero et al. 2014):

a. To understand patterns of biodiversity at different spatial scales, ranging from individual sites to across entire countries
b. To understand how natural and anthropogenic stressors such as climate change, land use change, and industrial development influence biodiversity
c. To understand the effectiveness of different types of management actions taken to enhance or protect biodiversity.

Such biodiversity assessments can only be carried out using indicators, since biodiversity is too complex to be fully quantified and understandable to policymakers. This complexity relates to the fact that biodiversity has many aspects (Pereira et al. 2013) such as:

- the genetic composition of populations
- the viability of particular populations to the structure
- species richness of communities
- the structure of their habitats and
- the functioning of ecosystems.

It is impossible to define indicators that would cover all these aspects simultaneously. Consequently, the choice of indicators will depend on the specific goals and context of the assessment, and a combination of different indicators may be most effective for providing a comprehensive and nuanced picture of biodiversity (EASAC, 2005).

In this chapter, we discuss the various indicators for biodiversity and their monitoring. We will start with the DPSIR framework in Section 2.2, which is a promising framework for assessing biodiversity change. Then proceed with an inventory of indicators used by the Global Biodiversity Outlook (GBO) in Section 2.3, and the state indicators that are used within the 2019 IPBES report (Section 2.4). Section 2.5 discusses the use of single indicator measures in biodiversity assessments. Section 2.6 is about monitoring and monitoring techniques. Section 2.7 concludes.

2.2 DPSIR framework to assess biodiversity

2.2.1 Introduction

To assess changes in biodiversity, two approaches are relevant, firstly assessing the state of biodiversity (using state indicators) and secondly assessing the causal chain of activities and related pressures on the state of biodiversity. To this end, the causal framework DPSIR was developed. The DPSIR framework is a useful way of organising different indicators, with a focus on cause-and-effect relationships between indicators.

2.2.2 Elements of DPSIR framework

DPSIR is a framework for assessment of change and policies, and it is suggested as an assessment framework for changes in biodiversity risks as well, see Maxim et al. (2009). It is a causal framework that describes the interactions between driving forces, pressures, state, impacts, and responses. DPSIR was first adopted by the European Environment Agency (EEA, see: DPSIR - European Environment Agency | eea.europa.eu), and is used by for example the Environmental Policy Agency of the United States, see Figure 2.1.

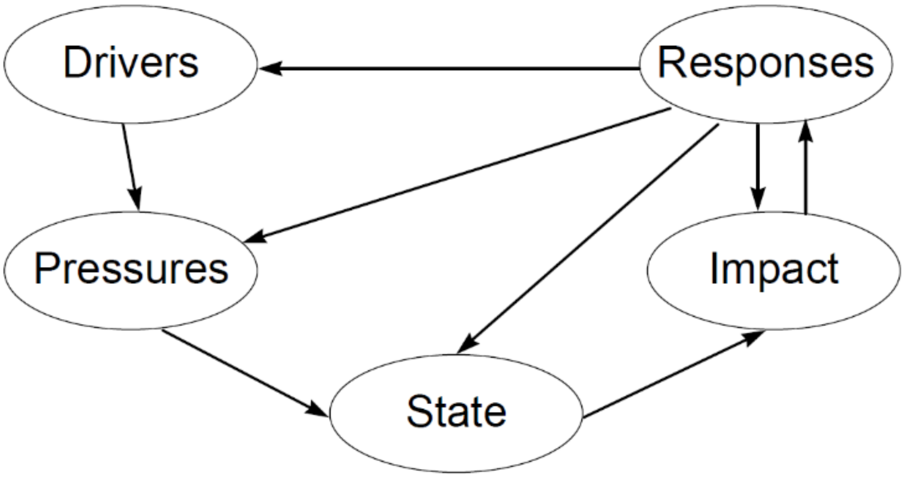

Figure 2.1 *DPSIR framework*
Source: Smeets and Weterings (1999).

The notion of *Impact* can focus on completely different target points. Within socioeconomic science, impacts usually focus on the effect on human systems associated with changes in environmental functions (Maxim et al. 2009). In many policy documents, such as the IPBES report on biodiversity (IPBES 2019), the main reason to protect biodiversity is that it is essential for human survival and the provision of so-called ecosystem services. Besides this direct human importance, biodiversity is seen as a goal in itself, for example from the perspective of intrinsic value. These two are in fact the assumed impact targets of biodiversity.

In contrast, the development of indicators on the *State* of biodiversity has mainly been developed from a biological perspective, describing species richness, completeness of ecosystems and the amount of nature available. Because complete information on biodiversity is not available for each area, many global models and assessments use the relationships between pressures and changes in the state of biodiversity.

For declaring that a factor is a pressure there is need for causal evidence of the relationship between the pressure (e.g., climate change, use of natural resources, pollution and urbanisation) and the state of biodiversity (Maxim et al. 2009). Then indicators of *Pressures* will be used as indicators for biodiversity. Pressures are influenced by human activities, which include both drivers for economic activities and restrictions on these activities. Some of the indicators on *Driving forces* focus on influencing economic activities through policies, for example restrictions set by policy, such as the size of protected areas and certification schemes. In the context of policies, indicators for *Responses* are used.

2.2.3 Conclusions

The strength of the DPSIR framework is that to understand and evaluate the state of biodiversity -in the context of its assumed impacts-, indicators of driving forces, pressures and responses are used to explain the state of biodiversity. In contrast, most biodiversity impact assessments focus primarily on the state of biodiversity. For example, in the frequently used LCA method ReCiPe (see Section 4.3), a biodiversity indicator on the extent and quality of the species composition of ecosystems is used for impact assessment in the value chains. In the biodiversity model GLOBIO (see Section 4.4), a comparable index is used for global biodiversity assessment. Another advantage of the DPSIR framework for assessment is the determination of its assumed impacts. This gives the assessment a clear focus on finding a set of biodiversity indicators that are relevant for evaluating the impact of biodiversity on wellbeing, in the broadest sense of the word.

2.3 Aichi Biodiversity Targets

2.3.1 Introduction

The fifth report of the GBO (Secretariat of the CBD 2020) provides the measurement of progress on the goals of the Convention on Biological Diversity (CBD), the so-called Aichi Biodiversity Targets. In this section, we discuss the indicators used to assess these Aichi goals, and then derive the main insights from this exercise.

2.3.2 Indicators of the Aichi Biodiversity Targets

A number of indicators is based on the drivers. The first Aichi goal is the increase of awareness of biodiversity. This is measured by surveys where questions are posed on having heard of the concept, understanding the concept, and being aware of the steps needed for biodiversity conservation. The second target is the integration of biodiversity in national strategies, planning, accounting and reporting. This is implemented by for example counting the number of countries that implement the System of Environmental Economic Accounting (SEEA) of the UN.

A second number of indicators is focused on policies. The third Aichi goal is the elimination of harmful subsidies and other perverse incentives and the implementation of positive incentives. One indicator is the amount of potentially harmful government support to agriculture in categories of harmfulness of this support. Another is the number of countries that implemented biodiversity-relevant economic instruments, such as taxes, subsidies, and tradable permit systems. The fourth Aichi target is on sustainable production and consumption. Sustainability can be measured by the ecological footprint, which is a pressure indicator. The development of Red List species that are internationally traded is an indicator that is being used. Furthermore, some indicators focus on availability of legislation and certificates.

The fifth Aichi biodiversity goal is that habitat loss is halved. This is measured by changes in the extent of different ecosystems such as wetlands and forests, and added to this may be the quality of these ecosystems. For quality loss in forests, the loss of tree cover is used as an alternative indicator, while the proportion of degraded land is used as an indicator. Alternatively, changes in the Red List Index (RLI) can be used as an indicator of habitat loss. The Biodiversity Habitat Index (BHI), based on land cover change and a database on species richness (PREDICTS database), provide an indicator related with habitat loss.

The sixth Aichi target is on sustainable management of aquatic resources, so this target is more policy focused. This is measured by the percentage of fish stocks that are overfished (pressure) and the percentage of fish stocks that is in safe biological limits (state), while fish catch under certification by the Marine Stewardship Council (MSC) (driver) is used as an indicator.

The seventh target is on sustainable agriculture, aquaculture and forestry. Here, indicators about the percentage of increasing number of species are used as an indicator (state), but indicators on pesticide and fertiliser use (pressures) are implemented. However, this is part of Aichi target 8, the reduction of pollution.

Measurement of Aichi target 9, the prevention and control of invasive alien species, is obviously a pressure target, and is directly measured by the trend of invasive alien species worldwide as recorded by the IUCN Global Register of Introduced and Invasive Species (GRIIS) database.

A very specific target, number 10, is the minimisation of the pressure on ecosystems vulnerable to climate change, with an emphasis on coral reefs. Here, policy indicators are combined with the state indicator percentage of coral reefs showing quality issues (bleaching). Also, a general ocean health index may be used (OECD 2019).

Target 11, goals for the size of protected areas, is obviously a policy/driver indicator, and is measured by the size of the areas that have a protected status, which reduces the risk of destruction. OECD (2019) refers to a Global Database for Protected Area Management Effectiveness (GD-PAME), that is however mainly based on self-assessment.

Aichi target 12, the reduction of the risk of extinction, is measured by state indicators, i.e., an index of the survival of species survival, where a value of 1 indicates that all species in a group are of least concern, and 0 that all species are extinct. Regional indexes weight these indices based on the importance of the area for survival of the species of concern. Furthermore, the Living Planet Index is a state indicator based on the development of species for which sufficient information on trends is available. The Biodiversity Intactness Index (BII) is an indicator of the state of biodiversity, but not directly related OECD (2019). The number of extinctions prevented through conservation policy is a type of state index but related with policy and drivers.

Aichi target 13, safeguarding genetic diversity, distinguished between cultivated plants, farmed and domesticated animals, wild relatives, valuable species maintained, and strategies implemented to reduce genetic erosion. Indicators informing about conservation activities are combined with indicators about the development of for example trends in the number of local breeds that become extinct or are at risk, so a Red List index of different breeds.

Aichi target 14 is about the role of ecosystem services. It is measured by the trends in the capacity of ecosystems to provide the 18 categories of ecosystem services or natures contributions to people. Only qualitative trends are distinguished.

Aichi target 15 is focused on ecosystem restoration and resilience, with an emphasis on carbon stocks. It is measured by a policy/driver variable, the reported projects of restoration, and a Bioclimatic Ecosystem Resilience Index (BERI).

Aichi target 16 is access and sharing benefits from genetic resources, and is therefore not directly relevant for the topic of this report. Target 17 is about the formulation of biodiversity strategies and action plans, and is therefore an indicator for policy intentions measured as the country’s progress towards implementing a national biodiversity strategy and action plan, and target 18 is about respect for and integration of traditional knowledge and practices. Measured as the country’s progress in implementation of traditional knowledge and practices with effective participation of indigenous and local communities at all relevant levels. Target 19 is about the sharing of innovation and knowledge, which is an indicator about potentials for improving policies. The generic indicator is the number of maintained species inventories being used. Finally, target 20 is about the mobilisation of finance for biodiversity, both domestically and in development assistance for developing countries. With as underlying indicator for example official development assistance and public expenditure on conservation and sustainable use of biodiversity and ecosystems.

2.3.3 Conclusions

In summary, the Aichi Biodiversity Targets are a mix of different targets focused on state, policy, drivers, and pressures without a clear analytical structure. The targets themselves are sometimes measured by mixes of elements of the DPSIR framework. Although the purpose of the Aichi Biodiversity Targets is to avoid deterioration of the state of biodiversity, the indicators for the state of biodiversity are only loosely related with the biodiversity concept. Moreover, many of the indicators provide information on the drivers and pressures of biodiversity without a clear systematic guidance. All these measures are useful to provide an indication of how biodiversity is developing, but are not sufficient to create a framework to evaluate the state of biodiversity and biodiversity change. The indicators are certainly not suitable to attach monetary values to it.

2.4 State indicators of biodiversity

2.4.1 Introduction

The IPBES (2019) report provides a broad insight into the state of biodiversity. The report uses over fifty quantitative global indicators covering many aspects of nature, because nature is too complex for trends and status to be captured in one or a few indicators. In this section, we discuss many of these indicators to find out to what extent these indicators may provide a starting point for a list of indicators that may be relevant for biodiversity assessments.

Global indicators of biodiversity decline such as the IUCN Red List Index and the Living Planet Index (Loh et al. 2005) are single dimensional and have relatively long assessment intervals (up to 10 years or longer). In this period, species can go from being relatively abundant to being on the verge of extinction. As a result, biodiversity change is often determined when effective responses are no longer feasible and ecosystem damage is considerable or even irreversible. This basically means that they function as late-warning indicators. While these indicators are instrumental in providing information for biodiversity conservation, the detection of early-warning signs of critical biodiversity change is necessary to be able to proactively respond (Schmeller et al. 2018). Thus, there is an urgent need for a standardised and harmonised observation or data exchange system for delivering regular, timely, and readily comparable information on the different dimensions of biodiversity change (Navarro et al. 2017).

To this end, the Group of Earth Observations Biodiversity Observation Network (GEO BON) proposed the concept of Essential Biodiversity Variables (EBV) in 2013 (Pereira et al. 2013), which is used in for example the IPBES global assessment report on biodiversity and ecosystem services (IPBES 2019). EBVs and indicators derived from them can be used for assessing biodiversity change over time and to measure achievement of policy goals such as the Aichi Biodiversity Targets set by the CBD, or the United Nations Sustainable Development Goals (SDGs) (Hardisty et al. 2019). The EBVs are still in development due to the complexity of the task, the limited resources available, and a lack of long-term commitment to maintain EBV data sets (Schmeller et al. 2018). The EBVs generally cover genetic composition, species populations, species traits, community composition, ecosystem function, and ecosystem structure, all representing the dimensions of biodiversity (Pereira et al. 2013).

Jaureguiberry et al. (2022) systematically reviewed natural science studies that compare the impacts that multiple direct drivers have on any of a large set of indicators of the state of biodiversity. This was done as part of the global assessment report by the IPBES (IPBES 2019). They found numerous large-scale indicators reflecting temporal changes in different dimensions of biodiversity, which are endorsed by global biodiversity-related initiatives, such as the CBD and the IPBES. Jaureguiberry et al. (2022) classified these indicators into the six classes of EBVs (see Table 2.1).

The selection of state indicators of biodiversity presented by IPBES is incomplete. Key indicators are still missing in this list, such as Potentially Disappeared Fraction of Species (PDF), Biodiversity Intactness Index (BII), and Marine Trophic Index (MTI), among others. Therefore, we added these indicators to the table. All indicators are being discussed by EBV class in the subsections below. Even though the table gives a comprehensive overview of relevant biodiversity indicators, the list biodiversity indicators is by no means exhaustive.

Table 2.1 *Biodiversity indicators associated with EBV classes*

EBV class	Indicators	Suggested by IPBES (2019)
Ecosystem structure	Area of mangrove forest cover	Yes
	Extent of intact forest landscapes	Yes
	Extent of remaining primary vegetation	Yes
	Extent of remaining wilderness	Yes
	Leaf area index (LAI)	Yes
	Percentage of live coral cover	Yes
Ecosystem function	Net primary productivity (NPP)	Yes
Community composition	Global bird species richness	Yes
	Local species richness	Yes
	Local species turnover	Yes
	Mean Species Abundance (MSA)	Yes
	Potentially Disappeared Fraction of Species (PDF)	No
	Biodiversity Intactness Index (BII)	No
	Species Threat Abatement and Recovery (STAR)	No
Species populations	IUCN Red List of threatened species	Yes
	Red List index	Yes
	Living Planet Index (LPI)	Yes
	Local species abundance	Yes
	Predatory fish biomass	Yes
	Prey fish biomass	Yes
	Proportion of fish stocks within biologically sustainable levels	Yes
	Wild Bird Index (WBI)	Yes
	Biodiversity Habitat Index (BHI)	No
	Species Habitat Index (SHI)	No
Species traits	Mammalian body mass (MBM)	No
	Marine Trophic Index (MTI)	No
Genetic composition	Proportion of local breeds, classified as being at risk, not-at-risk or unknown level of risk of extinction	Yes
	Phylogenetic diversity	No

Source: based on Jaureguiberry et al. (2022) and IPBES (2019) with additions from the authors.

2.4.2 Extinctions

Extinction of species is the most intuitive measure of biodiversity loss that seems directly related with the concept. Therefore, the evaluation of the state of biodiversity by IPBES starts with mapping out the rate of extinction during history. It shows that due to human drivers the extinction rate, i.e., the fraction of species that becomes extinct per year, increases exponentially. These extinction rates can be split over different classes of animals and plants.

2.4.3 Ecosystem structure

Ecosystem structure deals with 'the state and dynamics of ecologically relevant ecosystem traits' (Jaureguiberry et al. 2022). Most of these indicators deal with the extent of ecosystems, which basically is the area covered by something (usually expressed in hectares or square kilometres). Take for example the extent of mangrove forest or intact forest landscapes. Or the extent of remaining primary vegetation, the extent of remaining wilderness, or the percentage of live coral cover. All these indicators tell something about how widespread a species or habitat is, now and in the past. Leaf area index (LAI) is somewhat different in character, as it is defined as the one-sided green leaf area (canopy) per unit ground surface area. This index is commonly used as an indicator of the growth rate of a plant.

2.4.4 Ecosystem function

Ecosystem function entails 'the dynamics of small- to large-scale ecosystem processes and resulting ecosystem functions' (Jaureguiberry et al. 2022).

Net Primary Productivity
Net Primary Productivity (NPP) is equal to all of the carbon taken up by the vegetation through photosynthesis minus the carbon that is lost to respiration.

2.4.5 Community composition

This is a broad range of indicators, that cover 'measures of community diversity and ecosystem level interactions' (Jaureguiberry et al. 2022).

Species richness
Species richness is a commonly used indicator that measures the number of different species present in a particular ecosystem or geographic area (local, national or global), which provides a snapshot of overall biodiversity in a particular area. It is a relatively simple measure of biodiversity that is understandable for non-experts. It can be used to track changes in biodiversity over time and it can help identify areas that may be particularly important for conservation efforts.

Species turnover
Species turnover is an indicator for shifts in species composition along predefined gradients, either in space or through time. It makes use of metrics of similarity to track changes in species identities or their abundance (Vellend 2001; Dornelas et al. 2014).

Mean Species Abundance (MSA)
Calculating mean species abundance involves determining the number of individuals per species in a particular area. The study area could be a specific habitat type (e.g., a forest or a wetland), a geographic region (e.g., a country), or any other relevant spatial unit. To identify and count all individuals of species present in the study area, field observations can be used. Consequently, for each species identified in the survey, the total number of individuals observed could be recorded in a spreadsheet or database.

Figure 2.2 presents an example of the calculation of MSA. In the undisturbed reference situation, there would be four different species in this area (two deer, three trees, two frogs and an owl). Now, in the disturbed situation, three species have decreased in abundance (tree, deer and owl), while two show an increase (frog and rodent). As in MSA only species present in the undisturbed situation are included, increases in individual species abundance from the reference to the impacted situation are ignored. This means that the MSA is calculated as the mean of the abundance ratios of the four species in the reference situation, and the increase in frog abundance is being ignored. Otherwise, the indicator might be inflated by opportunistic or generalist species that benefit from habitat disturbance.

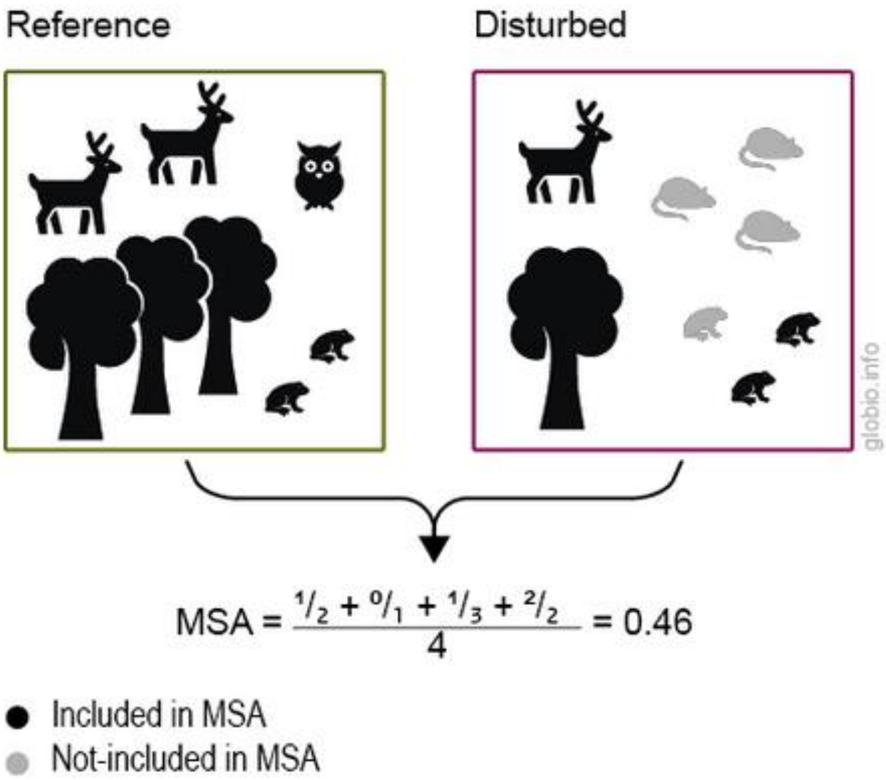

Figure 2.2 *Calculation of Mean Species Abundance (MSA)*
Source: GLOBIO (2023).

Potential Disappeared Fraction of Species (PDF)
Potential Disappeared Fraction of Species or PDF accounts for a fraction of species richness that may be potentially lost due to an environmental pressure (land use, ecotoxicity, climate change, or eutrophication). Take for example the situation in Figure 2.3 below. In the natural state, the potential number of species is four (fox, bird, tree and ant). The potential number of species in the modified state has dropped from four to two, since only one tree and one ant are still there. New species that were not there in the natural state, such as the (abundant) wheat, are not taken into account. In this example, the potential disappeared fraction of species is calculated as follows: PDF = 2/4 = 0.5.

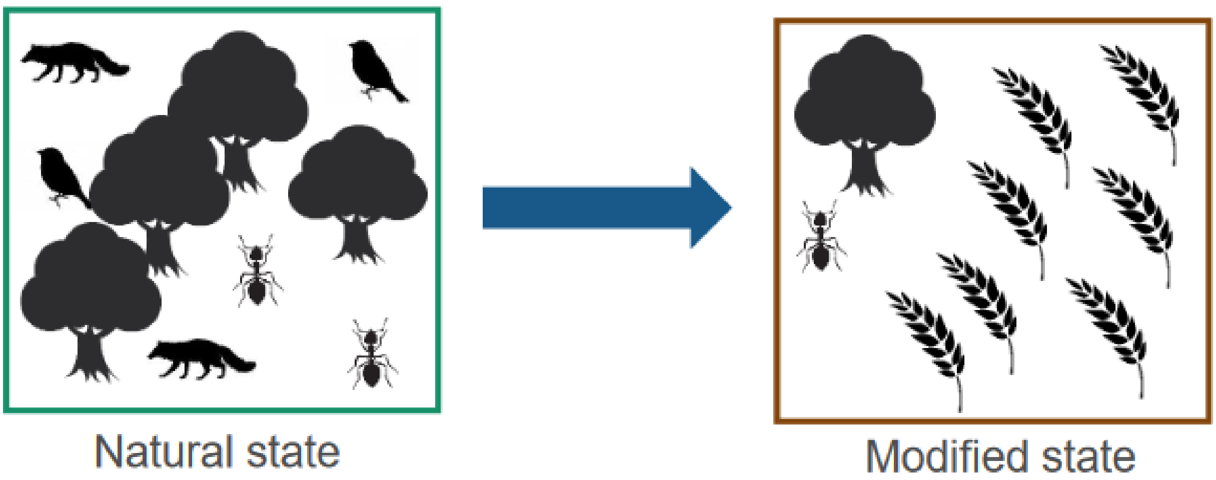

Figure 2.3 *Potential Disappeared Fraction of Species (PDF)*
Source: Sala (2020).

Biodiversity Intactness Index (BII)
The BII measures the degree to which a particular area's biodiversity remains intact, based on the presence or absence of certain species. The index is calculated by comparing the number of species observed in a particular area to the number of species that would be expected to occur in that area based on the area's characteristics, such as climate, topography, and land use. The BII is designed to be used at a relatively small scale, such as a specific geographic region, ecosystem, or protected area. It provides a more localised and detailed understanding of biodiversity loss compared to the global focus of the Red List.

The BII ranges from 0 to 1, with 1 representing a completely intact biodiversity as compared to the reference state, and 0 representing complete loss of biodiversity in the study area. The BII has several advantages as a biodiversity indicator, including its ability to integrate information from different taxonomic groups, its sensitivity to changes in both species' richness and abundance, and its ability to reflect changes in ecological function.

See: Biodiversity Intactness Index | Natural History Museum (nhm.ac.uk)

Species Threat Abatement and Restoration (STAR)
The Species Threat Abatement and Restoration (STAR) is a metric that evaluates the potential contributions that species threat abatement and restoration activities offer towards reducing extinction risk across the world. STAR is calculated from data on the distribution, threats, and extinction risk of threatened species derived from the IUCN Red List of Threatened Species. STAR assumes for most species that complete alleviation of threats would reduce extinction risk through halting the decline and/or permitting sufficient recovery in population and distribution. Species are weighted according to their Red List category, which means that endangered species have a higher weight than vulnerable species, and critically endangered species are weighted more than endangered species. STAR assesses the potential for reducing extinction risk by abating the threats to the species present at the area of interest, as well as the potential for reducing species extinction risk by restoring habitat for species that have been lost from that area (Mair et al. 2021).

2.4.6 Species population

Species population is described as 'the distribution and abundance of species as well as the specific structure of their populations' by Jaureguiberry et al. (2022, Table S4).

IUCN Red List of threatened species / Red List Index
The Red List is a database maintained by the International Union for Conservation of Nature (IUCN), that provides information on the global conservation status of species (BIP 2023). It classifies species into different categories based on their risk of extinction, with categories ranging from 'least concern' to 'extinct.' The information in the Red List can be used to inform conservation decisions and prioritise conservation efforts, such as identifying areas where conservation efforts should be focused or determining which species are in need of urgent protection measures.

The Red List is frequently used by governments, conservation organisations, and researchers to inform policy and guide conservation efforts, However, it is good to realise that the list relies fully on available data, which at times is incomplete or outdated, particularly for less well-known species. This can result in underestimating or overestimating the risk of extinction for certain species. Moreover, the Red List focuses primarily on individual species rather than broader ecosystems or habitats, which are relevant for biodiversity conservation.

The Red List Index (RLI) shows trends in the extinction risk of various species. It requires data from repeated assessments of species using the Red List categories and criteria, which are far more commonly available than detailed reliable time-series of population abundance data.

See: Red List Index | Biodiversity Indicators Partnership (bipindicators.net)

Living Planet Index (LPI)
The LPI is a measure of the global population trends of vertebrate species, calculated by the World Wildlife Fund (WWF) in collaboration with the Zoological Society of London (ZSL). It provides a snapshot of the health of the world's ecosystems by tracking changes in population sizes of thousands of species over time. The LPI is calculated using a statistical model that combines data from a range of sources, including population surveys, research studies, and citizen science programs. The model estimates the average annual rate of change in population size for each species, and then combines these values to calculate an overall index of biodiversity. This index is expressed as a percentage change relative to a baseline year (usually 1970), with a value of 100 representing no change in population size over time. A value less than 100 indicates a decline in population size, while a value greater than 100 indicates an increase.

The LPI is widely used as an indicator of global biodiversity trends and has been incorporated into numerous conservation and policy initiatives around the world. It should be noted that the LPI only captures changes in vertebrate populations and does not provide a complete picture of biodiversity loss, as it does not account for changes in species richness, genetic diversity, or ecosystem function.

Local species abundance
Local species abundance is the relative representation of a species in a particular ecosystem. It is usually measured as the number of individuals found per sample. The ratio of abundance of one species to one or multiple other species living in an ecosystem is referred to as relative species abundances.

Predatory and prey fish biomass
Scaling body mass by body size gives an indication of the condition and energy reserves of animals, such as predatory and prey fish. Often, the body mass is compared with the body mass in pre-industrial times.

Proportion of fish stocks within biologically sustainable levels
FAO states that this indicator 'measures the sustainability of the world's marine capture fisheries by their abundance. A fish stock of which abundance is at or greater than the level, which can produce the maximum sustainable yield (MSY) is classified as biologically sustainable. In contrast, when abundance falls below the MSY level, the stock is considered biologically unsustainable. This indicator is used to measure progress towards SDG Target 14.4' (FAO 2023).

Wild bird index
According to the Biodiversity Indicators Partnership, the Wild Bird Index (WBI) is the 'average trend in relative abundance of a group of bird species during the breeding season, often grouped by their association and dependence on a particular habitat. Birds are recognised as good indicators of environmental change and as useful proxies of wider changes in nature. The Wild Bird Index measures average population trends of a suite of representative wild birds as an indicator of the general health of the environment'.

See: Wild Bird Index | Biodiversity Indicators Partnership (bipindicators.net)

Biodiversity Habitat Index
The Biodiversity Habitat Index (BHI) is a tool used to assess the biodiversity value of habitat patches or sites based on the habitat characteristics and the species that are present. It has been used in various conservation and land-use planning efforts. The BHI considers a range of habitat features, including the size, shape, and connectivity of habitat patches, the presence of different habitat types (e.g., forest, wetland, grassland), and the degree of disturbance or fragmentation in the surrounding landscape. It takes into account the species richness and rarity of different taxa (e.g., birds, butterflies, mammals) found in the area. By means of a scoring system, the BHI gives a value to each habitat patch based on these factors, with higher scores indicating greater biodiversity value.

See: Biodiversity Habitat Index | Biodiversity Indicators Partnership (bipindicators.net) & Biodiversity Habitat Index | GEO BON (geobon.org)

Species Habitat Index
The Biodiversity Indicators Partnership describes the Species Habitat Index (SHI) as a measure for change in suitable habitat and populations of a country's species and the resulting change in the ecological integrity of ecosystems. The SHI is calculated and validated using species occurrence data combined with environmental change data informed by remote sensing. A SHI decrease of 0.01 means that species have, on average, experienced a 1% contraction in the quality of their habitat-suitable range compared to the baseline, and thus losses in total population size. The index is able to account for the effects of connectivity (spatial arrangement of suitable patches) and habitat restoration.

See: Species Habitat Index | Biodiversity Indicators Partnership (bipindicators.net) & Species Habitat Index | Map Of Life (mol.org)

2.4.7 Species traits

Species traits concern 'within-species changes in traits over time' (Jaureguiberry et al. 2022, Table S4).

Mammalian body mass
Body mass information for mammals gives an indication of the condition and energy reserves of these animals. This information can be used to investigate the patterns of body mass seen across geographic and taxonomic space and evolutionary time. Take for example the heritability of body size across taxonomic groups, which is a measure of how well differences in genes account for differences in traits, such as height, eye colour, and intelligence. Alternatively, the body mass can be assessed across continents or countries, and over evolutionary time (Smith et al. 2004; Smith and Lyons 2011; Zheng et al. 2023).

Marine Trophic Index
The Marine Trophic Index (MTI) measures the mean trophic level for all large marine ecosystems. A trophic level refers to a level or a position in a food chain or a food web. The index provides a measure of whether fish stocks, especially of large, bodied fish, are being overexploited and whether fisheries are being sustainably managed.

See: Marine Trophic Index | Biodiversity Indicators Partnership (bipindicators.net)

2.4.8 Genetic composition

Genetic composition refers to 'genetic diversity of populations, structure and inbreeding based on the number and frequency of alleles measured across time and species' (Jaureguiberry et al. 2022, Table S4).

Proportion of local breeds, classified as being at risk, not-at-risk or unknown level of risk of extinction.
Biodiversity Indicators Partnership states that more than 35 species of birds and mammals have been domesticated for use in agriculture and food production, and that there are more than 8,800 recognised breeds. This indicator aims to show whether genetic diversity of farmed and domesticated animals is being maintained using the proportion of local breeds classified as at risk, not at risk and unknown risk of extinction at a certain moment in time, as well as the trends for those proportions.

See: Proportion of local breeds | Biodiversity Indicators Partnership (bipindicators.net)

Phylogenetic diversity
Phylogenetic diversity measures the branch lengths in a phylogenetic tree (or phylogeny), which is basically a diagram that depicts the lines of evolutionary descent of different species, organisms, or genes from a common ancestor. Phylogenetic diversity is a measure of genetic diversity among species, that is most commonly used to compare diversity between geographic areas, based on the amount of evolutionary history (Miller et al. 2018).

2.4.9 Conclusions

The IPBES () report provides a broad insight in the state of biodiversity, using over fifty quantitative state indicators covering many aspects of nature. In the previous sections, we discussed many of these indicators, to find out to what extent these indicators may be relevant for impact assessments.

Community composition indicators MSA, PDF and BII have more or less the same logic in measuring how the current species composition is compared with a reference state, the natural state. Species richness is related with this, but instead the absolute number of species is counted, and it includes all species, including species that were not in the area in its natural state. Therefore, species richness will be higher for areas with a large number of species compared to the natural state.

Functional biodiversity is in the IPBES report mainly net primary production and carbon sequestration. Basically, it suggests that it is about the importance for the resilience of ecosystems and the importance for ecosystem services. The indicators ecosystem phenology and ecosystem disturbances of GEO BON may be related with this idea. However, another approach to functional biodiversity would be to investigate which species composition is needed for which ecosystem functions, including the resilience of the ecosystem.

Indicators about species population provide insights into the development of different species in specific areas or the world as a whole. The focus can be on species about which sufficient information is available such as in the Living Planet Index, the development of threatened species, or can focus on specific areas. However, surprisingly in this category are also included indices that provide information on the availability of suitable habitats for species, which is more a type of pressure indicator in the DPSIR framework.

Not only species are relevant, but also characteristics of these species. One is genetic diversity, which is especially important for long-term survival of the species, but also for protecting opportunities for improvement, substitution and resilience of current domesticated animals and plants. In case of disturbances, for example because of new diseases and climate change, it is important that enough genetic variation within species is available to adjust to these changes.

The last group of indicators is on organismal traits. This includes the size of the individuals, the place in the food web, and other aspects that are relevant in understanding the characteristics of the ecosystem.

In summary, there are various indicators used in the IPBES report, but a clear analytical frame (including pressure-effect relationships) is lacking. It is challenging to develop a more consistent approach, which has indicators that describe the different dimensions of biodiversity, that can potentially be valued monetarily and include a pressure-effect framework. However, this is outside the scope of the current report.

2.5 Use of single indicator measures of biodiversity in impact assessment methods

The previous Section 2.3 and Section 2.4 show that there is a large number of biodiversity indicators, that all assess specific aspects of biodiversity. Obviously, when conducting a biodiversity impact analysis, it is very complicated and unpractical to have a multiplicity of indicators. Therefore, in practice most impact assessment methods focus on just one biodiversity indicator that is assumed to grasp the most important aspects of biodiversity.

We discuss three indicators that are broadly used in biodiversity impact studies with a global focus. The first is the *Mean Species Abundance (MSA)* as an indicator for intactness of ecosystems compared with a reference state before the industrial expansion. This approach is used in the biodiversity model GLOBIO (see Section 4.4), that is for example used for global biodiversity evaluations. For each pressure factor, meta-studies using data on areas for which both MSA and the pressure factor are available a general relation is derived, and this is used to model the effects of changes in the pressure factor and the MSA indicator.

A similar approach is used in standard life cycle analysis according to the ReCiPe methodology (see Section 4.3), but then with the indicator *Potentially Disappeared Fraction of species (PDF)*. In contrast with the MSA, this indicator does not provide information on the development of the abundance of species, but only on the number of species that are lost. Therefore, the MSA usually gives earlier information on biodiversity decline than the PDF-indicator. In new developments in LCA, such as LC-IMPACT, the local PDF indicator is integrated with a global indicator, *Global PDF*. In this indicator, the local PDF of lost quality of ecosystems is weighted with the importance of these ecosystems for species. In that way it directly focuses on a target of biodiversity policies: to reduce the risk of extinction of species globally.

The use of one indicator makes it easier to estimate equations for impact chains from drivers through pressures to changes in the state of biodiversity. The basic assumption is that those single indicators are highly correlated with other dimensions of biodiversity, and that this simplification is necessary to get a grip on the problem. However, the disadvantage of using single indicators is that it does not take into account all relevant aspects of biodiversity.

2.6 Monitoring methods

2.6.1 Introduction

Monitoring results allow to analyse and draw conclusions about biodiversity changes, through the periodic assessment of these attributes on the same locations and across several time periods (Puerta-Piñero et al. 2014). Monitoring the changes in biodiversity is telling what has to be explained in the causal analysis from activity to biodiversity.

Biodiversity monitoring would require measuring almost everything, and this is very costly. Therefore, selections have to be made of what has to be monitored (Section 2.6.2), especially if conventional monitoring methods are used (Section 2.6.3). However, new developments in technology provide new opportunities for more intensive monitoring. New technologies such as remote sensing and satellite data may directly generate opportunities for cheaper and more extensive monitoring (Section 2.6.5) but may create opportunities for citizen volunteers to assemble information with for example mobile phones helping them to recognise species and to register their observations in a central database (Section 2.6.6). For both approaches, the methods may improve through the application of artificial intelligence (AI) (Section 2.6.7). In recent studies, there is more and more emphasis on the use of molecular techniques which are used for more detailed monitoring of biodiversity, which are beyond the scope of this study, see Kerry et al. (2022).

2.6.2 What should be monitored?

To understand patterns of biodiversity, various aspects of species and their environment can be monitored. This includes the population of animals and plants; the density of said populations; the spatial distribution of species, from local to nation-wide scale; the diversity among and within species; the presence and absence of particular species; and the behaviour of animals, for example seasonal migration, spawning and breeding, and habitat use.

Taking into account that there are over 45,000 species in the Netherlands, it is impossible to follow the development of all the species. In practice, a selection of species and groups of species whose changes reflect the trend in biodiversity, will suffice (CLO 2020; 2022). To this end, often indicator species are used. An indicator species is an organism whose presence, absence or abundance reflects a specific environmental condition. Indicator species can signal a change in the biological condition of a particular ecosystem, and thus may be used as a proxy to diagnose the health of an ecosystem (McDonough et al. 2012).

To better understand how pressures influence biodiversity, ecological pressure-effect functions (also called dose-response, stimulus-response functions) come into play. Similar to dose-response that are used in toxicology to measure the effects of a chemical on individual organisms, ecological pressure-effect functions assess how ecosystems change as drivers change because of human or natural causes. These dose-response relations can be estimated when on the one hand changes in the species richness of a taxonomic group, the relative abundance of a species group, or the share of a population is known and on the other information about the driving forces for the same locations is measured. These values should be compared to what would be expected in a similar situation without human influence (Karr 2006). In this light, it can be useful to monitor for example the presence of poachers based on the behaviour of animals, or the illegal logging of trees.

The need to include pressure-effect relationships in the marine environment (otherwise referred to as driver-response, dose-effect, dose-response (Halpern and Fujita 2013) or driver-response relationships (Hunsicker et al. 2016)) is stressed by many authors. However, a clear view is not available for ‘all effects (impacts) of all pressures of all activities’. In general, the default assumption is a linear relation as recommended by Halpern and Fujita (2013) and Judd et al. (2015) if the type of pressure-effect relationship was not specified. Hunsicker et al. (2016) conducted an extensive literature review and robust statistical analysis on single driver–response relationships in marine pelagic systems and found that non-linear driver-response relationships are most common, comprising at least 52% of all driver–response relationships. They found many non-linear relationships where the assumption of linearity could result in an under- or overestimation of impact (Hunsicker et al. 2016).

2.6.3 Conventional monitoring methods

Conventional monitoring methods include point counting and transect counting. Point counting is a method where animals, for example birds, are counted while the observer is stationary. Transect counting means that the observer slowly walks along a line or trail while visually searching for animals, taking care not to count the same individual twice. The observer should try to identify the animal immediately, possibly by capturing it (for example in the case of amphibians and reptiles) (Ottburg et al. 2018; Puerta-Piñero et al. 2014).

Point counting
For birds and other flying species, for instance, mist nets are a useful method when the purpose of monitoring is to create a list of the bird species known to inhabit a location. Observers can visually survey all appropriate habitats and microhabitats for amphibians and reptiles. This method aims to find the maximum number of species at the site as efficiently as possible and to obtain data on the abundance of individuals of a species. Yet another approach is performing the survey inside permanent quadrats, for example quadrats of 8x8 meter. Observers visually search for amphibians and reptiles and count individuals before they escape from the quadrat (Puerta-Piñero et al. 2014).

Transect counting
An approach used to survey primates and mammals in general, distance sampling along line transects requires that the observer accurately estimates the distance to each animal at the time of observation and the angle of the observation from the transect line. This approach can be used to survey animals and calculate estimates of absolute density. Naturally, the distance away from the observer at the time of identification is recorded. Moreover, camera traps are widely used for surveying communities of large terrestrial mammals. Under certain conditions and assumptions, camera trapping can be used to estimate population densities or can serve as an indicator for species abundance (Puerta-Piñero et al. 2014). To effectively sample fish, one needs to select a combination of, for example, direct current backpack electrofishing with dip nets in all habitats and seining in deep areas (Puerta-Piñero et al. 2014).

Bal (2017) notes that the

> ‘monitoring of biodiversity is expensive and can detract resources for managing biodiversity. Given limited resources for conservation, there is a need to decide how to manage biodiversity as well as how to monitor biodiversity. This entails considering the benefits and costs of alternative monitoring strategies, and selecting the ones that best inform and improve management decisions.’

Targetti et al. (2011) state that a

> 'comprehensive quantification of biodiversity in farming systems would require a very significant amount of work (and funds) even for a small area. Therefore, biodiversity indicators are needed to solve the problem of the measurement feasibility.'

2.6.4 Monitoring methods with technology

With the rapid developments in information and communications technology, more opportunities to monitor biodiversity have arisen. Brawata et al. (2017) state that the so-called Conservation Effectiveness Monitoring Program (CEMP) of the Australian Capital Territory government

> 'requires an ability to adapt as our knowledge base increases and new technologies become available (e.g., LIDAR, remote sensing, Collector app.).'

Moreover, Maina et al. (2016) advocate that there is an increasing need to effectively monitor a growing number of ecosystems of interest due to risks posed to these ecosystems by human activity and climate change, novel approaches to biodiversity monitoring are needed. Kerry et al. (2022) provided an overview of the latest techniques for measuring biodiversity including AI and ML techniques, see Section 3.5.

LIDAR is an acronym for Light Detection And Ranging. LIDAR technology uses light sensors to measure the distance between the sensor and the target object. From an aircraft this includes objects such as the ground, buildings and vegetation. For ground-based LIDAR, it measures building fronts and street furniture in extreme detail. With the latest technologies, it is possible to obtain colour values of the scanned surface to create an automatically textured model. It can provide the user with very high accuracy measurements and is very cost-effective for the amount of data generated (NOAA 2023).

The Collector App for ArcGIS enables organisations to use maps to gather data in the field and to synchronise the results with their enterprise GIS data. With this app, observers can update data in the field, log their location, and put the data captured back into a central GIS database directly from their phone or mobile device. This increases accuracy and helps eliminate recording errors. Fieldworkers are much more efficient and accurate, reducing error and time (Esri Insider 2016).

Remote sensing is the process of detecting and monitoring the physical characteristics of an area by measuring its reflected and emitted radiation at a distance from the targeted area. Special cameras collect remotely sensed images of the earth, which help researchers 'sense' things about this planet. Remote sensing can be applied to take for example images of large areas on the globe's surface or to make images of temperature changes in oceans. This may be used to map large forest fires from space; to track clouds to help predict the weather or watch erupting volcanos, and help watch for dust storms; to track the growth of a city and changes in farmland or forests over several years or even decades; and to map the ocean bottom (USGS 2022).

Remote sensing can be used to calculate the share of semi-natural vegetation (Desjeux et al. 2015). Alternatively, this can be done 'through the computation of the difference between the total area occupied by farms (excluding tracks, building and housing areas) and the cropped utilised agricultural area: the larger that difference, the larger the area under semi-natural vegetation is likely to be' (Desjeux et al. 2015). Moreover, the so-called crop diversity index (CDI) can be used. This index is based on the assumption that the richer the crop diversity and the more equal the shares, the better for biodiversity. In line with Paracchini and Britz (2010), crop diversity is assessed using the Shannon index, applied to measure both the diversity of crops and the evenness in their distribution. The index is computed from the shares of 21 crops from Farm Structure Survey (FSS) data (Desjeux et al. 2015).

With automated acoustic sampling, microphones and digital recording devices are placed at selected locations and left to record sounds of birds or frogs, while observers are away working in other locations Puerta-Piñero et al. (2014). Maina et al. (2016) have shown that the application of low-cost acoustic recorders based on the Raspberry Pi microprocessor 'are capable of capturing audio recordings from which

[they] can compute acoustic indices of biodiversity and identify bird species of interest'. Obviously, this methodology can be used to monitor other animals, such as frogs.

Methods that may seem expensive at first sight, for example because large investments are needed or because the methods only become cheaper after getting experience with it and fine tuning them, they may be cost-effective in the long term. In this regard, Bal (2017) suggests that 'monitoring of certain indicators, such as cat abundance using remote cameras, has large upfront costs (e.g. of buying camera traps), but is cheap once cameras are installed (as cameras can be left for long periods without personnel being required to man them).' As a result, the relative costs of monitoring are perceived as 'intermediate' (Bal 2017).

Furthermore, the use of drones helps to better monitor biodiversity. Take for example drones that have thermal cameras attached to them that register infrared. These drones can be used to determine the presence of animal species, but to identify forest fires or people or animals that are drowning.

Environmental DNA (eDNA) is another novel research method, in which the presence of a species can be determined on the basis of DNA in (water) samples. The method is based on the fact that all organisms living in the water leave DNA behind through skin cells, faeces and urine. By collecting water samples and analysing them in a laboratory for DNA of the target species, the presence of the target species can be determined. Research has shown that DNA freely dissolved in water breaks down within two to three weeks. Thus, the detection of DNA of a species in a water sample indicates the recent presence of this species (NDFF 2023a).

2.6.5 Monitoring by volunteers (citizen science)

Another relevant development over the years is the growing importance of the involvement of volunteers (citizen science). Involving citizens in biodiversity monitoring means an increasing capacity to collect data. These volunteers generally need support from professionals, but the costs of this will be rather limited. The support must ensure that for example all volunteers count everything they observe; that they accurately describe the location where the observation was made; that they make accurate estimations of the number of animals; and that they have sufficient knowledge to determine what they observe. Obviously, these factors have a great effect on the quality of the data. The use of automatic image recognition applications, such as ObsIdentify, may contribute to better observations. At the moment, these apps are not able to identify species accurately all the time, but this will probably improve in due time, taking into account the rapid technological advancements.

Even more opportunities arise when citizen science and technological developments are combined. In 2016 for example, satellite imagery was used to count the seal populations in Antarctica, see Figure 2.4. This was carried out with Tomnod, a satellite imagery crowdsourcing platform that in their own words was 'the world's first' such platform (Maxar Technologies 2019). It started as the quest of a few graduate school students at the University of California San Diego to find the ancient hidden tomb of Genghis Khan in northern Mongolia. These students asked the public to help them tag features in satellite images of Mongolia, creating a map of the area and allowing them to focus their search.

Tomnod was used in 2014 to search for the vanished plane of Malaysia Airlines (flight number MH370). The plane disappeared from the radar while flying from Kuala Lumpur to Beijing. More than 1 million km^2 of satellite images were loaded into Tomnod and volunteers were asked to tag oil slicks, wreckage, rafts and other objects of interest. More than 10 million people contributed, placing more than 14 million tags during 775 million map views. However, the plane of Malaysia Airlines Flight with flight number MH370 and its 239 passengers have not been found and the mystery of MH370's location endures today.

As for the seal census,

> 'the Tomnod community helped researchers survey the Weddell seals population in Antarctica to help them understand where seals like to live. In less than two months, about 5,000 volunteers covered the same amount of area that a ground-based research team covered in a decade' (Maxar Technologies 2019).

Research on Weddell seals is particularly interesting because these mammals

> 'are one of Antarctica's icons. These seals are the southern-most mammal in the world, can live up to thirty years, and they are perfectly adapted to living in some of the harshest conditions on the planet.'

Moreover, Weddell seals are an *indicator species*, 'because they prey on Antarctic toothfish (known as Chilean seabass), a high-end delicacy in many upscale restaurants around the world.' There is little information about Antarctic toothfish, but understanding how the Weddell seals populations are doing gives insight into the health of the Ross Sea (LaRue et al. 2021).

However, information from the Dutch practice suggests that observations that are registered by citizens are rather skewed towards two species groups, birds and plants, and that there are not many observations for other species, such as moths, butterflies, dragonflies, reptiles and amphibians, mammals and fungi.

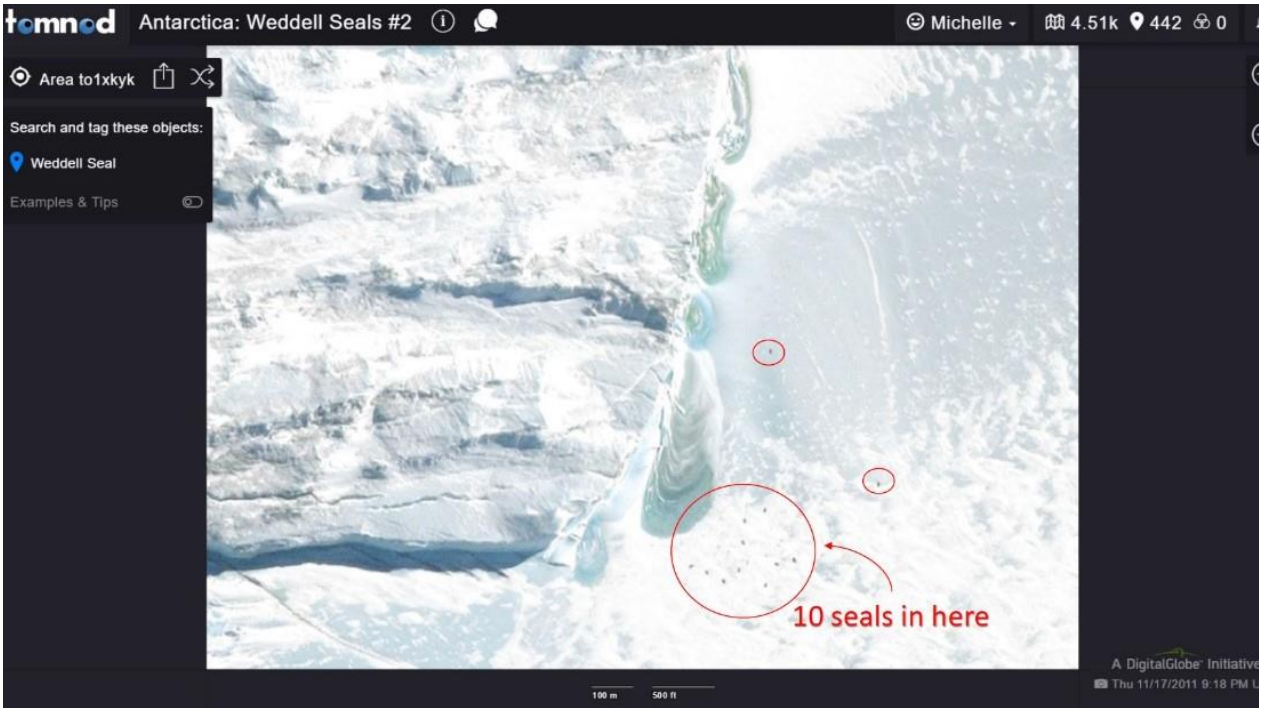

Figure 2.4 *Tomnod satellite imagery with Weddell seals marked in red*
Source: Maxar Technologies (2019).

2.6.6 Using artificial intelligence (AI) and machine learning (ML)

In recent years, the geospatial industry has evolved rapidly. Nowadays, according to Michelle LaRue, 'the most customer demand for crowd-sourced image tagging comes from those looking to train AI and ML algorithms, which need thousands of tags to learn what they are looking for' (Maxar Technologies (2019). Moreover, current 'customers want to know who is creating their training data. This has fundamentally changed the business of image tagging by the crowd and we are evolving with this change' (Maxar Technologies (2019). Due to this shift, the tech behind Tomnod was winded down, and a related platform called Maxar's GeoHive was created that allows customers to select their crowd.

This development illustrates that the training of AI and ML, deep learning or explainable learning gains importance. In short, machine learning can be defined as

> 'an application of AI that provides systems the ability to automatically learn and improve from experience without being explicitly programmed. Machine learning focuses on the development of computer programs that can access data and use it learn for themselves' (Selig 2022).

Little by little, computers will be better capable of automatically recognising images, which in time will make analysing high numbers of data much less time-consuming and less expensive.

2.6.7 Summary

Given the limited resources for nature conservation and taking into account that biodiversity monitoring is generally time-consuming and expensive, the most cost-effective ways of monitoring should be considered. Therefore, indicator species are often used, which are species whose presence, absence or abundance reflects a specific environmental condition. To better understand how pressures influence biodiversity, ecological pressure-effect functions (also called dose-response or stimulus-response or driver-response functions) are used.

Various monitoring methods exist, ranging from conventional to technical methods using artificial intelligence. A relevant development in monitoring is the involvement of volunteers in biodiversity monitoring. This provides an increasing capacity to collect data, however, observations suggest that the data registered by citizens are rather skewed towards two species groups, birds and plants.

2.7 Conclusions

To manage biodiversity effectively, adequate indicators of biodiversity are required. Over the years, numerous indicators have been developed. To get hold of the various types of indicators for biodiversity, the DPSIR approach can be used, which is a framework that is used to describe the interactions and causalities between society and the environment. DPSIR distinguishes drivers, pressures, state, impact and response indicators. The framework is useful to explore what type of indicators can be used to assess both the state of biodiversity and the impact of socioeconomic activities on biodiversity.

The analysis of recent global publications shows that the report that evaluates the goals of the Convention on Biological Diversity contains a mix of all types of indicators. This is consistent with the heterogeneity of the goals of the convention, the Aichi biodiversity targets. From an analytical point of view this is challenging, because these indicators represent different phases in the causal chain. In the global biodiversity report by IPBES (2019) more than fifty state indicators are used to measure the state of biodiversity.

Given the different elements of biodiversity, an assessment of the state of biodiversity using multiple state indicators is desirable. However, biodiversity impact methods such as LCA or GLOBIO, use just one state indicator. The disadvantage of such single indicator approaches is that they do not catch the multidimensionality of biodiversity. The use of one indicator makes it easier to estimate equations for impact chains from drivers through pressures to changes in the state of biodiversity. The idea here is that those single indicators are highly correlated with other dimensions of biodiversity and that this simplification is needed to get a handle on the problem.

To understand patterns of biodiversity, the selected indicators need to be monitored through the periodic assessment of these attributes on the same locations and across several time periods. Within conventional biodiversity monitoring, the use of biodiversity indicators can help with this, as well as using indicator species. The focus must be on measuring those aspects that give the best indication of the value of the ecosystems. With the rapid developments in information and communications technology, novel opportunities to monitor biodiversity have arisen, such as for example remote sensing, remote cameras, drones, Light Detection And Ranging (LIDAR), automated acoustic sampling, and environmental DNA (eDNA).

3 Biodiversity monitoring for policy: two examples

3.1 Introduction

This section describes two examples to illustrate biodiversity indicators development and monitoring. The first example is the definition of biodiversity indicators for the Marine Strategy Framework Directive (MSFD). While the marine approach to biodiversity indicators deviates quite a bit from the terrestrial approach, the type of analytical problems that have to be tackled in marine research are not fundamentally different. The marine approach is also struggling to get a grip on the multidimensionality of biodiversity, and as it stands, the impact chain from economic activities to biodiversity impacts has not been quantified. The second example is the description of the monitoring of nature policy in the Netherlands with a main focus on the terrestrial nature.

3.2 A marine approach to biodiversity indicators

3.2.1 Introduction

In the marine sector, indicator development started more or less independent from the terrestrial approach. The basic idea is that indicators should be policy oriented, and one is aware that many ecosystem services, originally mainly the amount of food for humans that is available, depends on the quality of the marine ecosystems. Therefore, the concept of ‘good environmental status’ has been developed, which is specifically related with the capacity of ecosystems to supply ecosystem services. Indicators that are being developed should be focused on measuring to what extend the ecosystem is in good environmental status.

3.2.2 Indicators to assess the state of the marine ecosystems

Natural capital consists of a biotic (ecosystem) and abiotic part and is responsible for the supply of ecosystem services. The ecosystem consists of different ecosystem components, as reflected by the biodiversity concept, which can be characterised by various biotic attributes representing both structure and functioning such as diversity, e.g. functional, taxonomic or (phylo-)genetic, the occurrence of specific taxa, specific population dynamics, habitat structure or community composition (including both size structure or species composition). This needs to be captured by an appropriate typology that is aligned to the various relevant policy frameworks so that the information can be used for the assessment of the status of the marine ecosystem as required in various policy frameworks (e.g. MSFD, Biodiversity Strategy) as well as its capacity to supply ecosystem services. To that end, the marine ecosystem and its biodiversity is thought to comprise of ecosystem components. For each of these ecosystem components (and possibly even subcomponents), a suite of indicators exists that can be used for the assessment of the marine ecosystem and its biodiversity. Here we focus only on the operational indicators, as these allow an assessment of ecosystem state and hence its capacity to supply ecosystem services.

Table 3.1 *Typology of ecosystem components distinguishing specific relevant subcomponents*

Ecosystem components	Ecosystem subcomponents
Marine mammals	Cetaceans
	Seals
Seabirds	Wading feeders
	Surface feeders
	Water column feeders
	Benthic feeders
	Grazing feeders
Fish	Commercial
	Non-target: Pelagic
	Non-target: Demersal
	Non-target: Elasmobranchs
	Migratory fish, e.g., diadromous, anadromous
Reptiles	
Seabed habitats including benthic communities	Benthic Infauna
	Benthic Epifauna
	Physical habitats (MSFD predominant or broad habitat types, from the European Nature Information System, EUNIS)
Water column habitats including plankton communities	Phytoplankton
	Zooplankton
	Physical habitats (MSFD predominant or broad habitat types)

Table 3.2 *Overview of potential indicators to assess marine biodiversity*

Source	Name of indicator	Ecosystem component	MSFD Criteria
OSPAR	Grey Seal Pup Production	Mammals	
OSPAR	Marine Bird Breeding Success or Failure	Birds	
OSPAR	Size Composition in Fish Communities	Fish	
OSPAR	Recovery in the Population Abundance of Sensitive Fish Species	Fish	
OSPAR	Seal Abundance and Distribution	Mammals	
OSPAR	Abundance and Distribution of Coastal Bottlenose Dolphins	Mammals	
OSPAR	Abundance and Distribution of Cetaceans	Mammals	
OSPAR	Marine Bird Abundance	Birds	
OSPAR	Size Composition in Fish Communities	Fish	
OSPAR	Pilot Assessment of Mean Maximum Length of Fish	Fish	
OSPAR	Pilot assessment of Changes in Plankton Diversity	Plankton	
OSPAR	Condition of Benthic Habitat Communities: the Common Conceptual Approach	Benthic fauna	
OSPAR	Proportion of Large Fish (Large Fish Index)	Fish	
OSPAR	Change in Average Trophic Level of Marine Predators in the Bay of Biscay	Predators	
OSPAR	Pilot Assessment of Production of Phytoplankton	Phytoplankton	
HELCOM	Distribution of Baltic seals	Mammals	D1C4
HELCOM	State of the Soft-Bottom Macrofauna Community	Benthic epifauna	D5C8
HELCOM	Abundance of Waterbirds in the Breeding Season	Birds	D1C2
HELCOM	Abundance of Waterbirds in the Wintering Season	Birds	D1C2
HELCOM	Abundance of Salmon Spawners and Smolt	Fish	D1C2
HELCOM	Abundance of Sea Trout Spawners and Parr	Fish	D1C2
HELCOM	Abundance of Key Coastal Fish Species	Fish	D1C2
HELCOM	Population trends and abundance of seals*	Mammals	D1C2
HELCOM	Nutritional status of seals*	Mammals	DIC3
HELCOM	Reproductive status of seals*	Mammals	D1C3
HELCOM	Proportion of Large Fish in the Community	Fish	D4C3
HELCOM	Abundance of Coastal Fish Key Functional Groups	Fish	D4C2
HELCOM	Zooplankton mean size and total stock	Zooplankton	D4C3
UNEP-MAP	Habitat distributional range & Condition of the habitat's typical species and communities	Benthic habitats	D6C4; D6C5
UNEP-MAP	Species distributional range – Marine Mammals	Mammals	D1C4
UNEP-MAP	Species distributional range – Seabirds	Seabirds	D1C4
UNEP-MAP	Species distributional range – Marine Turtles	Reptiles	D1C4
UNEP-MAP	Population abundance of selected species – Seabirds	Seabirds	D1C2
UNEP-MAP	Population abundance of selected species – Marine Mammals	Mammals	D1C2
UNEP-MAP	Population abundance of selected species – Marine Reptiles	Reptiles	D1C2
UNEP-MAP	Population demographic characteristics – Seabirds	Seabirds	D1C3
UNEP-MAP	Population demographic characteristics – Marine Mammals	Mammals	D1C3
UNEP-MAP	Population demographic characteristics – Marine Reptiles	Reptiles	D1C3

In the assessment of the status of the marine ecosystem, the structural aspects of the ecosystem (as captured by the concept of biodiversity) and its functioning (in the MSFD captured by the food web concept) can be distinguished. At present, most of the operational indicators representing structural aspects of the ecosystem are population-level indicators, while its functioning is deemed to be best represented by community level indicators (where the different communities represent specific trophic guilds). For most of the ecosystem components, these indicators are still being developed and tested against the criteria to derive a suite of community-level indicators deemed to adequately represent the functioning of the various trophic guilds that make up the marine food web. Within each community/trophic guild, the following aspects were identified that should not be 'adversely affected due to anthropogenic pressures':

- Diversity, i.e., species composition and their relative abundance
- Size distribution of individuals across the trophic guild
- Productivity

In addition, the balance of total abundance between the trophic guilds should not be adversely affected.

Notably, the indicators representing the functioning of the ecosystem and its food web are arguably the best candidates to represent the capacity of the ecosystem to supply ecosystem services. As such, the (further) development and testing of these indicators as part of an assessment framework should be prioritised.

3.2.3 Criteria for indicator selection

To assess the state of the marine ecosystem, two types of biodiversity indicators can be distinguished: the 'operational' indicators used to support implementation of the main marine policy frameworks, such as MSFD, and 'surveillance' indicators, which monitor key aspects of the ecosystem, but are limited in their capacity to underpin specific management advice (Shephard et al. 2015).

Operational indicators are expected to have well-understood relationships between state and specified anthropogenic pressure(s) and to have policy targets (i.e., in case of the MSFD reflecting Good Environmental Status, GES). Operational indicators track changes in the state of particular attributes of specific ecosystem components, and recovery to GES should be achievable by the introduction of specific measures to manage the known pressures. In contrast, for the surveillance indicators, there is either insufficient evidence to define targets, thus limiting their capacity to support formal assessment, or the links to anthropogenic pressures are not well understood, thus preventing their use to guide management. This distinction has consequences for the selection process of what can be considered suitable indicators, in that stricter selection criteria apply to the operational indicators, certainly pertaining to the 'management' criterion but to some of the sub-criteria of the 'quality of underlying data' criterion. For example, the two 'management' sub-criteria 'relevant to management' and 'management thresholds targets are estimable' do not apply to surveillance indicators, because no relationship between the ecosystem aspect covered by the indicator and a manageable pressure exists nor does any threshold to set a policy target. Neither is the 'quality of underlying data' sub-criterion 'reflects changes in ecosystem component that are caused by variation in any specified manageable pressures' a prerequisite for surveillance indicators. Something specifically advantageous of surveillance indicators compared to operational indicators is that they could provide an early warning signal for potential future change in an ecosystem attribute.

Table 3.3 *Criteria and sub-criteria used in the selection process for operational ecological indicators*

Criteria	Sub-criteria (issues)	Rationale
Availability of underlying data	Existing and ongoing data	Indicators are supported by current or planned monitoring programmes that provide the data necessary to derive the indicator. Ideal monitoring programmes should have a time series capable of supporting baselines and reference point setting. Data should be collected on multiple sequential occasions using consistent protocols
	Relevant spatial coverage	Data should be derived from an appropriate proportion of the regional sea, at appropriate spatial resolution and sampling design, to which the indicator will apply
	Relevant temporal coverage	Data should be collected at appropriate sampling frequency and for an appropriate extent of time relevant to the time scale of the process or attribute the indicator describes.
Quality of underlying data	Indicators should be technically rigorous	Indicators should ideally be easily and accurately determined using technically feasible and quality assured methods
	Reflects changes in ecosystem component that are caused by variation in any specified manageable pressures	The indicator reflects change in the state of an ecological component that is caused by specific significant manageable pressures (e.g., fishing mortality, habitat destruction). The indicator should, therefore, respond sensitively to particular changes in pressure. The response should be based on theoretical or empirical knowledge, thus reflecting the effect of change in pressure on the component in question; signal to noise ratio should be high. Ideally the pressure–state relationship should be defined under both the disturbance and recovery phases
	Magnitude, direction and variance of indicator is estimable	The indicator should exhibit a predictable direction, exhibit clear sense of magnitude of any change, and estimates of precision should allow for detection of trends or distinct locales—requiring that some measure of sampling error or variance estimator is available
Conceptual basis	Scientific credibility	Scientific, peer-reviewed findings should underpin the assertion that the indicator provides a true representation of process, and variation thereof, for the ecosystem attribute being examined
	Associated with key processes	The link between the indicator and a process that is essential to food web functioning should be clear and established, based on the current understanding of trophic dynamics
	Unambiguous	The indicator responds unambiguously to a pressure
Communication	Comprehensible	Indicators should be interpretable in a way that is easily understandable by policy makers and other non-scientists (e.g., stakeholders) alike, and the consequences of variation in the indicator should be easy to communicate
Management	Relevant to management	Indicator links directly to mandated management needs, and ideally to management response. The relationship between human activity and resulting pressure on the ecological component is clearly understood
	Management thresholds targets are estimable	Clear targets that meet appropriate target criteria (absolute values or trend directions) for the indicator can be specified that reflect management objectives, such as achieving GES. Ideally control rules can be developed
	Cost-effectiveness	Sampling, measuring, processing, analysing indicator data, and reporting assessment outcomes should make effective use of limited financial resources

3.2.4 Policy objectives and indicator targets

The overarching goal of the MSFD is to achieve ‘good environmental status’ (GES) for marine seas under its jurisdiction. To that end, the MSFD identified 11 qualitative criteria, the so-called descriptors (D1-D11), four of which describe the state of the ecosystem and can therefore be considered criteria for biodiversity:
- D1 Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions.
- D3 Populations of all commercially exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock.
- D4 All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity.
- D6 Sea-floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected.

The other descriptors describe human-induced pressures or their impacts.

In the MSFD good environmental status (GES) is defined as:

the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generations, i.e.:

- *the structure, functions and processes of the constituent marine ecosystems, together with the associated physiographic, geographic, geological and climatic factors, allow those ecosystems to function fully and to maintain their resilience to human-induced environmental change. Marine species and habitats are protected, human-induced decline of biodiversity is prevented, and diverse biological components function in balance.*
- *hydro-morphological, physical and chemical properties of the ecosystems, including those properties which result from human activities in the area concerned, support the ecosystems as described above. Anthropogenic inputs of substances and energy, including noise, into the marine environment do not cause pollution effects.*

The text above suggests that GES is a holistic concept, relating to whole ecosystem across large-scale spatial regions. The MSFD requires marine regions to be productive. Marine resources should be exploited, but only at sustainable rates: i.e., exploited at a level that can be maintained indefinitely and which conserves ecosystem structure and function. If this is the case, the ecosystem can be considered to be at GES. Given this imperative to make productive use of marine resources, albeit at sustainable rates, and considering that any human disturbance of marine ecosystems inevitably has an impact, it is clear that the state at which the ecosystem is considered at GES cannot be the same as the pristine state of the ecosystem prior to human intervention. Thus, GES must represent a permissible degree of deterioration in the state of the marine ecosystem within a region, away from the pristine state to a state that still maintains full ecosystem functionality and resilience, but which permits sustainable exploitation of the marine natural resources contained within the ecosystem.

In the MSFD, an 'environmental target' means a qualitative or quantitative statement on the desired condition of the different components of marine waters in respect of each marine region or subregion. The MSFD urged Member States to establish a comprehensive set of environmental targets and 'associated indicators' for their marine waters so as to guide progress towards achieving good environmental status in the marine environment. Prescribed indicative lists of pressures, impacts and characteristics have to be taken into account by these Member States as well as continuing the application of relevant existing environmental targets laid down at national, Community or international level in respect of the same waters. This ensures that these targets are mutually compatible, and that relevant transboundary impacts and transboundary features are considered, to the extent possible. Environmental targets relate to individual ecosystem components and the specific associated indicators used to monitor change in the ecosystem component in question. The integration of the information obtained from assessing the state of all ecosystem components, based on environmental targets for each component and prevailing values of their associated indicators relative to these environmental targets, provides the holistic overview to establish whether or not the marine ecosystem in a particular region is at GES. Thus, for any specific associated indicator for a given ecosystem component, the environmental target is the value of this indicator that would be expected if the ecosystem was at GES.

Where threshold values cannot be defined, the setting of trends-based targets can provide a pragmatic and operational alternative. In essence this means that where scientific evidence suggests that current values of the indicator in question reflect a sub-GES situation, an a priori directional change can be proposed as an alternative to setting an absolute target indicator value. When such an approach is adopted, meeting such trends-based targets does not mean that GES has been achieved. At best, it implies that the appropriate measures have been put in place to move the ecosystem attribute reflected by variation in the indicator towards GES.

Targets should not conflict across indicators within MSFD and international policy frameworks. Potential inconsistencies between Water Framework Directive (WFD) and MSFD have been analysed by Borja et al. (2010). The WFD ecological status classification of Good Ecological Status (GES) is based on biological and chemical monitoring results. The normative definitions of the WFD (Annex 5) set the descriptive definitions for the high, good, and moderate status for different water categories and quality elements. According to

Borja et al. (2010), normative definitions describing the desirable status for GES of biological quality elements as in WFD are not included in the MSFD. Instead, the MSFD applies qualitative descriptors to determine the GES, which, to a certain extent, can be related to some of the elements within the WFD. Proper alignment of WFD and MSFD is in particular relevant if both directives apply to the same area, e.g., overlap in the coastal zone between baseline and 1 nm. OSPAR (2012) and HELCOM considered three characteristics for GES for biodiversity to be equivalent to assessment of Good Ecological Status for the WFD and Favourable Conservation Status (FCS) for the Habitats Directive, which accommodate a defined deviation from reference state (i.e., the absence or negligible level of impact from anthropogenic pressures). Thus, GES can be expected to:

- Accommodate some level of impact, such that quality is not even across an entire region or subregion;
- Represent a defined deviation from a reference state, accommodating sustainable use of the marine environment, provided that there is no further deterioration from present state (at an appropriate scale of assessment).

Background work to define GES at individual descriptor level was undertaken by different Task Groups for each descriptor, including biodiversity (Cochrane et al. 2010). Criteria for assessing the relevant attributes and components of biological diversity are summarised in Table 3.4.

Table 3.4 *Criteria for assessing the relevant attributes, criteria and indicator classes to be selected*

Attribute	Criteria	Indicator classes
Species state (includes sub-species and populations where they need to be assessed separately; apply criteria to each sub- species/population)	Species distribution	Distributional range
		Distributional pattern
	Population size	Population biomass
		Population abundance (number)
	Population condition	Population demography e.g., body size; age class structure; sex ratio; fecundity rates; and survival/mortality rates
		Population genetic structure
		Population health (sub-lethal condition, e.g., disease prevalence; parasite loading; pollutant contamination)
		Inter and intra-specific relationships (e.g., competition, predator-prey relationships)
	Habitat distribution, extent and condition	Habitat distributional range
		Habitat distributional pattern
		Habitat
Habitat/community state	Habitat distribution	Habitat distributional range
		Habitat distributional pattern
	Habitat extent	Areal extent of habitat
		Habitat volume
	Habitat condition	Physical condition (structure and associated physical characteristics, incl. structuring species)
		Hydrological condition (incl. water movement, temperature, salinity, clarity)
		Chemical condition (incl. oxygen, nutrient and organic levels)
	Community condition	Species composition
		Relative population abundance
		Community biomass
		Functional traits
Landscape state (where assessed as 'Listed' habitats)	Landscape distribution and extent	Landscape distributional range
		Areal extent of landscape
	Landscape structure	Habitat composition and relative proportions
	Landscape condition	As for habitat condition and community condition, as appropriate
Ecosystem state	Ecosystem structure	Composition and relative proportions of the ecosystem components

Source: Cochrane et al. (2010).

Borja et al. (2013) addressed the question of what GES entails especially with regard to the level at which targets are set (descriptors, criteria, indicators), to scales for assessments (regional, sub-divisions, site-specific), and to difficulties in putting the GES concept into practice. Options for determining when GES has been met are indicated. The MSFD implementation can be based largely on existing data and can be focused on the activities of the Regional Seas Conventions (Borja et al. 2013), i.e. OSPAR, HELCOM.

Generic frameworks can be used to assess potential changes of GES by human impacts, see Figure 3.1.

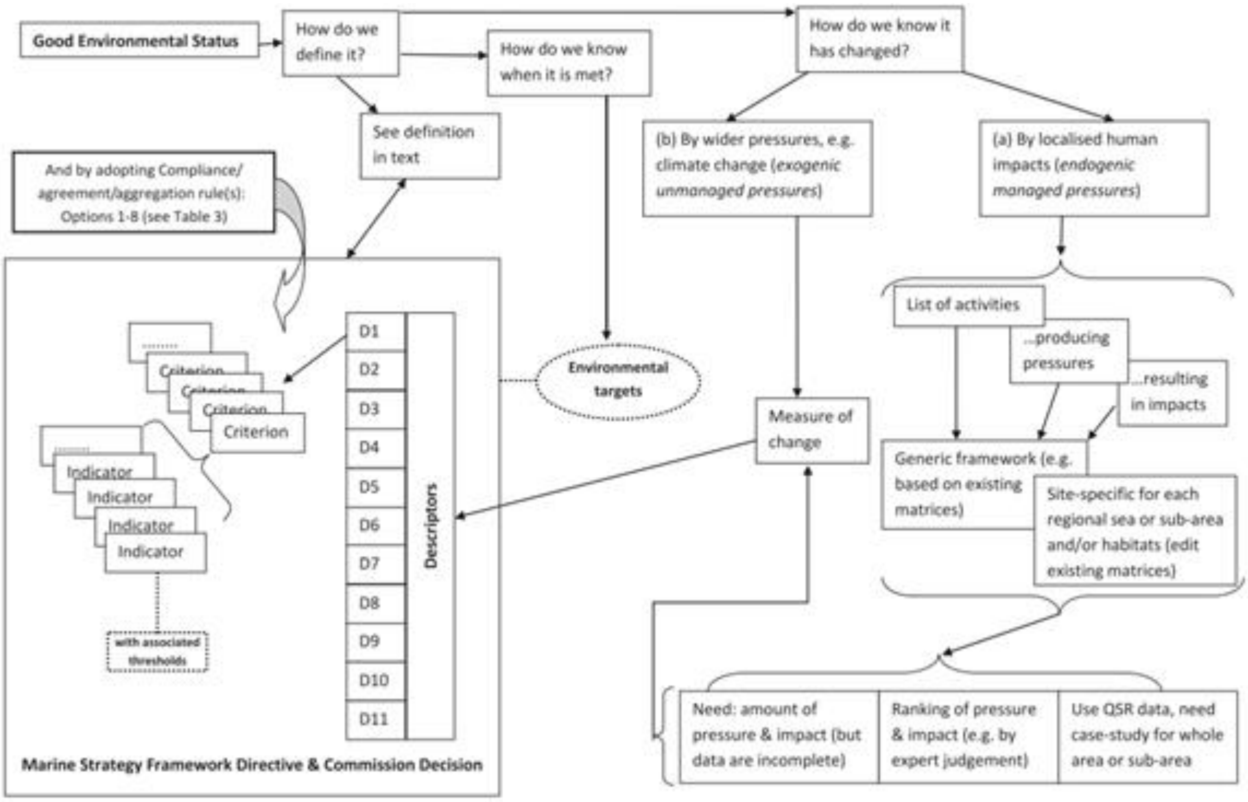

Figure 3.1 *Making operational the definition of Good Environmental Status: Quality Standard Reports Source: Borja et al. (2013).*

3.2.5 Conclusion

The marine approach to biodiversity indicators deviates quite a bit from the terrestrial approach. For example, a single biodiversity indicator has not been developed for marine impact assessments, and the whole impact chain from economic activities to biodiversity impacts has not been quantified. Nevertheless, the type of analytical problems that have to be tackled in marine biodiversity research are not fundamentally different, such as getting a grip on the multidimensionality of biodiversity.

3.3 Monitoring of Dutch nature policy

In this section, we set out the approach to monitoring nature management policy in the Netherlands. In doing so, we provide insight into what protocols are being used to monitor biodiversity in the Netherlands.

3.3.1 Policy for nature management

The policy for nature management in nature reserves is set out in the national Grant Scheme for Nature and Landscape (in Dutch: Subsidiestelsel Natuur & Landschap, abbreviated SNL). SNL is based on the principle of uniformity in nature management in all provinces. Through SNL, the provinces grant subsidies to nature managers. Grant schemes are tailored to the type of nature and management practices needed, based on abiotic nature conditions (water management and food wealth), biotic conditions and cultural history.

The National Nature Network (in Dutch: Natuurnetwerk Nederland, abbreviated NNN) is a network of existing and newly created nature reserves and was introduced in 1990 in the Nature Policy Plan of the Ministry of Agriculture, Nature and Food Quality (LNV) (originally, NNN was called Ecologische Hoofdstructuur, or EHS). The aim of the nature network is to halt the degradation of nature and biodiversity by creating a coherent network of nature reserves. This is done by increasing the size of nature areas and linking them together by acquiring, developing and managing adjacent and intermediate agricultural land. Large areas of nature help to improve water and environmental conditions (see CLO, 2017).

Natura 2000 sites are part of a network of nature reserves in the European Union that are protected under the Birds Directive adopted in 1979 (Directive 79/409/EEC) and the Habitats Directive (Council Directive 92/43/EEC) adopted in 1992. These guidelines indicate which types of nature (habitat types), and which species must be protected. The Netherlands is obliged to maintain or restore the species and habitat types at a favourable conservation status. Most of the Natura 2000 sites fall within the NNN. However, some of these areas (over 27,000 hectares) are not part of the nature network, mainly agricultural areas (see CLO, 2022).

3.3.2 Monitoring in Dutch nature areas

For all areas in the National Nature Network where management is subsidised by SNL, there is compulsory monitoring. However, parts of the nature network are excluded from this subsidy scheme, such as areas belonging to the Ministry of Defence, water companies, Rijkswaterstaat and municipalities. Guidelines for monitoring management in nature reserves that are financed by the SNL are listed in the document ‘Werkwijze Monitoring en Beoordeling Natuurnetwerk en Natura 2000/PAS’ (in English: ‘Methods of monitoring and evaluation assessment of the National Nature Network and Natura 2000/PAS’, see Van Beek et al. (2014)).

In the Netherlands, the monitoring and assessment of the data is the responsibility of the provinces (Van Beek et al. 2014). For example, each province must draw up a multi-annual monitoring plan, the so-called Provincial Monitoring Programme (PMP), for the monitoring of the various nature areas under management. At least once a year, each province must consult with the nature organisations to discuss progress on nature management. In addition, the provinces draw up monitoring reports. Based on these reports, it becomes clear whether the agreed objectives of nature policy have been achieved or that implementation needs to be adjusted.

Nature organisations can choose to carry out the monitoring themselves in the area they manage, or they can leave it to the province. If nature organisations carry out the monitoring themselves, they can apply for a subsidy from SNL. Collected data are stored in the National Database Flora and Fauna (NDFF).

Indicators for monitoring the National Nature Network (NNN)
Monitoring requires the collection of information on the quality of the ecosystems, the associated plant and animal species, vegetation and controllable conditions (environment, water, space and structure of the site). For areas within the National Nature Network, the following indicators are used (Van Beek et al. 2014):

1. *Vegetation*
   Vegetation mapping is required to determine this indicator. An inventory will be carried out to identify the types of vegetation that are present in an area. These are drawn on a so-called vegetation map. This provides information about both the biotic and the abiotic conditions in an area.
2. *Flora and fauna*
   Species groups are monitored for this indicator. These are supposed to give a picture of the ecological quality of an area. Four groups of species are distinguished (plants, breeding and migrant birds, dragonflies, and butterflies and grasshoppers). For each type of management, at maximum three species groups should be monitored. The grid cell method can be used for this purpose, where the number of species in a block of fixed dimensions is counted.
3. *Structural elements*
   The structural elements of a site concern matter such as the height of the vegetation, open places and

dead trees. This allows the multiformity of a site to be depicted. Information about the structural elements can be collected by means of aerial photographs or vegetation mapping.

4. *Site conditions*
   To examine how a nature area is influenced by its environment (in particular water and air), the site conditions should be determined. These include groundwater level, soil acidity and nutrient richness. By means of the free and publicly available ITERATIO application, the site conditions can be determined on the basis of the vegetation mapping.
5. *Spatial conditions*
   Sustainable populations of species benefit from a large surface area that is connected to the same or a similar nature management type in the environment. Spatial conditions are assessed based on the total surface area of the nature area and the distance from the area to another nature site with similar characteristics.
6. *Naturalness*
   This indicator is only used for the nature type 'large-scale dynamic nature'. The extent of naturalness depends on the human disturbance to the ecosystem. The indicator looks at landscape forming processes and the presence of key species. Usually, the naturalness of an area is analysed by a team of experts.

Indicators for monitoring Natura 2000 sites
Natura 2000 sites have been designated under EU nature policy. The national government must report to the EU on the progress towards agreed conservation objectives. The following indicators should be used (Van Beek et al. 2014):

1. *Habitat types: presence and surface area of vegetation types*
   For each management area, the number of vegetation types to be found and the surface area these vegetation types cover should be determined. For this purpose, vegetation mapping can be used.
2. *Habitat types: quality of vegetation types*
   Each habitat type has a so-called profile document, which states which vegetation types have a moderate or good quality. Using the vegetation mapping can then score for each type of vegetation in the management area.
3. *Habitat types: site conditions*
   This concerns the acidity, moisture level (or groundwater level), salinity, nutrient richness and flood tolerance. By using the free and publicly available ITERATIO application, the site conditions can be derived from vegetation mapping.
4. *Habitat types: typical species*
   Typical species are animal or plant species that are associated with a specific habitat type and that represent a certain quality. To check whether the number of typical species in a management area has remained the same, has increased or decreased, the number of typical species are counted according to the grid cell method.
5. *Habitat types: structure*
   Information on the structure of the management area can be derived from the vegetation mapping.
6. *Birds- and Habitats Directives species: numbers and habitat*
   To understand the development of species, information should be provided on the number of individuals per species and the extent and quality of the habitat of each species. The number of individuals per species can be determined using the grid cell method; information on size and quality of a habitat can be obtained on the basis of an expert analysis.

Each Natura 2000 site has a management plan. This plan determines which species will be monitored and in what way this information will be collected.

Similarities and differences between indicators Nature network and Natura 2000 sites
There is a certain overlap between the indicators to be monitored in the National Nature Network and in the Natura 2000 areas. All the indicators needed for the monitoring in the Natura 2000 sites are required for the monitoring of the Nature Network. In addition, for the monitoring of the Nature Network, information is required on the spatial conditions and the naturalness (Van Beek et al. 2014).

3.3.3 Standard costs for monitoring

As with nature management, monitoring of nature works with standard costs. This takes into account which species groups need to be inventoried. Based on the experience figures of the nature site managers, cost calculations have been made on the basis of quotations from market parties. The calculated costs are the bare costs of carrying out monitoring according to the provincial 'Working method for nature monitoring and assessment of the National Ecological Network and Natura 2000/PAS'.

In the system of standard costs for management, bare costs are taken into account, plus a percentage for additional costs for work supervision, among other things. Work supervision is involved in monitoring, but in particular costs for outsourcing and control (to guarantee the reliability of data at a later stage) are necessary to ensure that monitoring is carried out properly. To this end, a percentage has been calculated based on the experience of the site managers (see BIJ12 2023e).

The costs are based on outsourcing the fieldwork, digital delivery and a report on the fieldwork, excluding VAT. The fieldwork is done by professionals and does not include the costs of guidance, overhead, and organisational costs. The pricing is based on terrestrial monitoring. Differences per management type arise due to variation in species richness, accessibility, clarity and the like (see BIJ12 2023f).

Take for example the monitoring costs in bogs (see Table 3.5). The direct costs for monitoring are based on the frequency and area to be monitored and the type of activities that are undertaken (vegetation mapping, structure mapping, and monitoring of flora, breeding birds and dragonflies). The total direct costs are €27.53 per hectare per year; more than half of which goes to vegetation mapping. To calculate the standard costs, an average surcharge of 50% of the direct costs is added to the total. The actual subsidy for monitoring amounts to 75% of the standard costs (the remaining 25% is what the nature manager has to bear); in this case, the subsidy amount is €31.06 per hectare per year.

Table 3.5 *Composition of standard costs and subsidy amount for monitoring in bogs*

Bog	Costs per ha for 100% monitoring (€)	Frequency	% area to be monitored	Average costs per ha per year (€)
Vegetation mapping	174.38	1/12	100%	14.53
Flora	57.49	1/12	100%	4.79
Breeding birds	21	1/6	100%	3.51
Butterflies	-	-	-	-
Dragonflies	19.16	1/6	100%	3.19
Structure mapping	1.05	1/12	100%	0.09
Total direct costs				**27.53**
+50% surcharge for indirect costs				13.88
Standard costs				**41.42**
-25% own contribution				10.36
Subsidy amount				**31.06**

Source: BIJ12 (2023e).

3.3.4 Nature Quality Calculation Module for Nature Network Netherlands (RNN)

BIJ12, the joint working organisation of the provinces, has developed a calculation module for the NNN (RNN). The module aims to 'carry out the monitoring of NNN in a uniform, efficient and reliable manner' and 'a large part of the analysis of the monitoring data is carried out automatically (BIJ12 2023a).'

With RNN, it is possible to calculate the quality of nature using five indicators (characteristics of the nature reserve) in a standardised manner. This is done at the scale of an SNL management type within an assessment area. For the indicators on flora & fauna and site conditions, the required data are retrieved automatically. For the other indicators (structure characteristics, spatial conditions, and naturalness), entry must be made manually based on 'expert judgment'. The system then automatically calculates the nature quality for all management types (see BIJ12 2023a).

With this module, Dutch provinces will be able to calculate

> 'unambiguously the nature quality of SNL nature management in the Netherlands for their own assessment areas. Moreover, both provinces and land management organisations will be able to analyse the effects of nature management at a detailed level within their nature reserves' (BIJ12 2023a).

3.3.5 National Database Flora and Fauna (NDFF)

The National Database Flora & Fauna (NDFF) bundles, uniforms and validates monitoring data on the distribution of plant and animal species. About 150 million observations have been stored in the NDFF. The information in the database is public, but not freely accessible: the monitoring data can be requested via a subscription or a one-time delivery. The database provides information about plant and animal observations. Only validated data is made available in the NDFF (see BIJ12 2023c).

More than 100 databases are bundled in the NDFF. The NDFF is supplemented daily with recent observations. Many different organisations are involved in the NDFF. BIJ12 takes care of the management and exploitation of the National Database Flora and Fauna by order of a consortium of stakeholders.

The organisations and chain partners that are involved in the NDFF supply data, develop products and/or supply personnel for the NDFF. The subscribers, governments, water boards, et cetera, make use of the NDFF via the Import and Export portal. Non-subscribers, such as researchers, can contact the NDFF for data.

To guarantee the reliability of observations in the NDFF, all observations are checked and validated. For most of the observations this is done automatically, some of them need to be checked manually. To be able to validate as many observations as possible automatically, rules have been drawn up by the validation team, for example for checking familiar locations and/or for activities during the season. Observations that do not comply with the rules in the automatic validation are manually validated by a validation team. This is done per group of species.

The species experts always manually view observations with photographic or other evidence and make inquiries with the observers. If the observer and the species expert cannot work it out together, the assessment committee will decide about the observation in question. This will be communicated to the observer (BIJ12 2023b).

To improve and standardise the collection of nature data, the NDFF works with established protocols, standardised counting methods. This means that observations are always collected in the same way. Standardised counting has the great advantage that the data can be easily compared, both among themselves and from year to year, and that it is clear what can be done based on the observations. Observations made according to a set protocol can be recognised as such in the NDFF (NDFF 2023b).

Observations in which a certain species has not been seen or heard are called 'null observations'. These null observations are just as valuable as 'normal' observations. Null observations made using a protocol are validated and then entered into the NDFF. Null observations made without a protocol are not sufficiently reliable and are not entered into the NDFF (NDFF 2023c).

Protocols shall be established by a Quality Assurance Committee (in Dutch: Commissie Kwaliteitsborging) and assessed against the following criteria (NDFF 2023c):
- Is there a clear description of the procedure to be followed?
- Is the method of data processing prior to storage in the NDFF clearly described?
- What criteria apply to the use of this protocol for observers?
- What is the observer effort (for how long has been recorded)?
- Does the procedure provide data on the presence and absence of species or on numbers?

On top of the aforementioned protocols, several protocols have been designated for Agricultural Nature and Landscape Management (ANLb). The purpose of the NEM monitoring network for ANLb is to collect the data needed to assess the target realisation of the ANLb system. It compares the rural trends of target species within areas with ANLb and agricultural areas without management. This trend is measured using standardised measurement methods and locations. Measurement locations have therefore been designated across the country and are sampled in the same way everywhere. The protocols explain how the data should be collected. The ANLb monitoring contains protocols for amphibians and fish, butterflies and dragonflies, waterfowls, and breeding birds, among others (NDFF 2023d).

The NDFF works with established protocols to collect good quality nature data, so that these can be used for the licensing and exemption for the Flora and Fauna Act and the Nature Conservation Act. Three groups of protocols have been established, which can be recognised by the numbering (NDFF 2023c):

- The individual observations (species group number 000 where, for example, 01.000 stands for individual observations of amphibians).
- Inventory methods (species group number .001, .002, et cetera).
- Standardised inventory manuals drawn up in collaboration with CBS (Statistics Netherlands) as part of the Ecological Monitoring Network (NEM) (species group number .200, .201, et cetera).

Observations made according to a particular protocol are recognisable as such in the NDFF. The protocols are drawn up in collaboration with species experts (BIJ12 2023c).

Of all the 150 million observations that are currently in the NDFF (that is, as of August 2019), the majority consists of individual observations (84 million), see Table 3.6. Observations from inventory methods come second (36 million) and standardised inventory manuals within the NEM monitoring come third (29 million observations). For the last years, yearly around 9 million observations are added (please note that the number for 2019 is incomplete, as it is from August 2019).

Table 3.6 *Observations within the NDFF as of August 2019*

	Individual obs.	Inventory methods	Standardised (NEM)	Total
Before 2015	53.075.045	31.188.941	24.671.230	108.935.216
- 2015	6.205.141	1.046.827	1.395.931	8.647.899
- 2016	6.789.819	1.264.513	1.310.654	9.364.986
- 2017	6.992.472	1.145.768	1.024.506	9.162.746
- 2018	7.106.461	1.057.371	367.295	8.531.127
- 2019	3.976.906	575.774	78.569	4.631.249
2015-2019	31.070.799	5.090.253	4.176.955	40.338.007
Total	84.145.844	36.279.194	28.848.185	149.273.223

Source: NDFF, edited by Wageningen Economic Research.

Comparing the number of observations before and after 2015, it becomes clear that the largest group of observations has always been that of individual observations, see Table 3.7. In recent years, this share has increased from around 50% before 2015 to 77% in 2015-2019. Observations from standardised inventory manuals within the NEM monitoring remain relatively modest, with a share of 10% between 2015 and 2019.

Table 3.7 *Number of protocols and share of observations within the NDFF*

Group of protocols	Protocols	Before 2015	2015-2019	Total	Total (obs.)
Individual obs.	18	49%	77%	56%	84.145.844
Inventory methods	45	29%	13%	24%	36.279.194
Standardised (NEM)	48	23%	10%	19%	28.848.185
Total		100%	100%	100%	149.273.223

Source: NDFF, edited by Wageningen Economic Research.

Within the NDFF, different species groups are distinguished: from fungi to mosses and vascular plants and from invertebrates to fish, birds, reptiles and mammals, see Table 3.8. Some groups are (partly) interconnected: species group 01, 05 and 10 all have to do with amphibians and/or reptiles and 03, 04, 07 and 09 all consist of invertebrates. Generally speaking, vascular plants and birds are the groups that relatively have many observations, even though their share has rapidly declined over the last years. For mosses, lichens, bryopsida and liverworts, butterflies, dragonflies and damselflies, micro (harvestmen) and other invertebrates, amphibians and reptiles, fungi, fish, and mammals, little to no data are available. By far the largest group however is that of 'Other protocols', which is basically a mixed bag full of protocols.

Table 3.8 *Share of observations (%) per species group within the NDFF*

Species groups (number and description)		Before 2015	2015-2019	Total
01	Amphibians	0	0	0
02	Mosses, lichens, bryopsida and liverworts	2	0	1
03	Invertebrates: butterflies	2	1	1
04	Other invertebrates	1	0	1
05	Amphibians and reptiles	0	0	0
07	Invertebrates: dragonflies and damselflies	1	0	1
09	Invertebrates: butterflies and micro	1	0	0
10	Reptiles	0	0	0
11	Fungi	2	1	2
12	Vascular plants	28	8	23
13	Fish	0	0	0
14	Birds	20	6	16
17	Mammals	1	1	1
100+	Other protocols	42	83	53
	Total	100	100	100

Source: NDFF, edited by Wageningen Economic Research.

Most observations within this group are individual observations via the website waarneming.nl, via an inventory through the Year-round Garden Counting via tuintelling.nl, and via the input portal to the NDFF, see Table 3.9. The share of individual observations via the website waarneming.nl has increased from 65% before 205 to 82% between 2015 and 2019. The share of observations via the input portal to the NDFF dropped dramatically from 34% before 2015 to 7% in the period 2015-2019. The inventory through the Year-round Garden Counting via tuintelling.nl was marginal before 2015 and increased to 11% between 2015 and 2019).

Table 3.9 *Source of individual observations (%) within the NDFF*

Source of individual observations	Before 2015	2015-2019	Total
Input portal to the NDFF	34	7	22
Website waarneming.nl	65	82	73
Year-round Garden Counting	0	11	5
Other individual observations	1	0	0
Total	100	100	100

Source: NDFF, edited by Wageningen Economic Research.

The NDFF data provide no information on the species groups within the individual observations. However, for the observations that are reported via waarneming.nl, some relevant statistics are available on their website (Waarneming.nl 2020). According to this website, as of 5 January 2020, 67,165,269 observations have been provided by 107,841 users in total. For 14,272,981 observations, photos have been added by users, and sounds were added for 55,142 observations. Observations were reported on 108,818 locations. Of the 67 million observations, two thirds were of birds, 11% were of plants, 5% of moths, 4% of butterflies, and 2% of dragonflies, mammals, and fungi. Hymenoptera, diptera, fish, locusts and crickets (orthoptera), beetles, mosses and lichens, and reptiles and amphibians account for 1% of all observations. The group

‘Other’ contains bugs, plant lice and cicadas, other arthropods (Arthropoda), molluscs, other insects, algae, seaweeds and other unicellular organisms, other invertebrates, and disturbances.

As for the Year-round Garden Counting in the Netherlands: this is a dataset of recurring counts (weekly or daily) at a fixed location. Counts can be carried out on the basis of a limited timeframe or week totals, when observers note the highest number per species. This mostly applies to the gardens of participants, mostly in built-up areas. This yields a lot of information about biodiversity in cities and villages, because there is a lot of information about the gardens themselves (Vogelbescherming Nederland et al. 2023).

The data that participants have collected can be analysed in relation to the garden and its specific aspects, such as the size of the garden, exposure, how much of it is paved, how much is lawn, how many herbs grow there, and if it contains a pond. Those aspects determine which species can be observed in the garden (Vogelbescherming Nederland et al. 2023).

The observers state which species they looked for and report everything they actually saw. This means that the absence of species that are not noted can later be established. This creates a wealth of information about the probability rate and the occurrence of species (common and otherwise) in gardens (Vogelbescherming Nederland et al. 2023).

In conclusion, the NDFF contained about 150 million observations in 2020, of which the majority consisted of individual observations. The importance of so-called citizen science is large and growing, as many of the observations are registered by citizens via websites or apps. However, the information within the NDFF is rather skewed towards two species groups: birds and plants. There are little observations for other species, such as moths, butterflies, dragonflies, reptiles and amphibians, mammals and fungi. This raises the question whether this skewness adequately represents the biological diversity of the Netherlands. Is the number of observations limited because the populations of these species are modest? That seems unlikely, since there are for example innumerable insects in the Netherlands. Probably it has to do with the fact that birds and plants are relatively easy to observe or that the public can relate to these species’ groups better (with birds being cuddlier than insects).

4 Biodiversity impact assessment methods

4.1 Introduction

Once biodiversity concepts and indicators are defined and changes in the biodiversity indicators can be monitored, the next step requires the analysis of the causal relationships between human activities and biodiversity. This is necessary to get insights into the possibilities to prevent further biodiversity decline by better business decisions and government policies. The purpose of this chapter is to give an overview of the main approaches that relate socioeconomic activities with biodiversity impacts. This chapter also compares terrestrial approaches with more or less independently developed marine approaches.

The standard approach to evaluate environmental impacts is the LCA (Section 4.2). Such an assessment starts with a list of activities, relates emissions and resource extractions to these activities and then calculates pressure factors for biodiversity that only in some approaches is used to calculate biodiversity changes. Different methods are available that are used to calculate biodiversity effects from emissions and resource extractions by activities. ReCiPe is a method developed within LCA (Section 4.3), and GLOBIO is a method developed for global impact assessment, but may be applied to relate pressures with biodiversity in LCA (Section 4.4).

Within LCA some new methods have been developed, being LC-IMPACT and IMPACT World+, which especially provide more regional differentiation, with LC-IMPACT using an indicator for global extinction of species instead of intactness of local biodiversity. Both approaches are not sufficiently developed for direct implementation (Section 4.5).

Within the marine sector the so-called Cumulative Impact Assessment (CIA) has been developed, which tries to catalogue all pathways from activities to final biodiversity impact and emphasises that the final effect is the result of the combination of all these pathways (Section 4.6). Currently the focus is on recognising high risk pathways, and these risks are determined by experts by scaling on a number of criteria. While the terrestrial approaches of GLOBIO and LCA tend to focus on a limited number of empirical relationships that are quantified, the marine approach tries to catalogue all relationships and evaluates them based on semi-quantitative indicators using expert-judgement.

Both terrestrial and marine approaches are developing towards a more integrated and regional approach, with a focus on increasing the level of quantification (Section 4.7).

4.2 Life Cycle Assessment (LCA)

4.2.1 Introduction

The central question in impact assessment for biodiversity is what effect different decisions or activities have on biodiversity. LCA is a tool that is specifically focused on the impacts caused by the decision to buy products or services. LCA is a methodological tool to quantify the potential environmental impacts of a product, process or service over its full life cycle: raw material extraction, manufacturing, use, and end-of-life. It is based on the ISO 14040-14044 standards by the International Organization of Standardization, see Figure 4.1. with ISO 14040:2006 – Environmental management – Life cycle assessment – Principles and framework, and ISO 14044:2006 – Environmental management – Life cycle assessment – Requirements and guidelines.

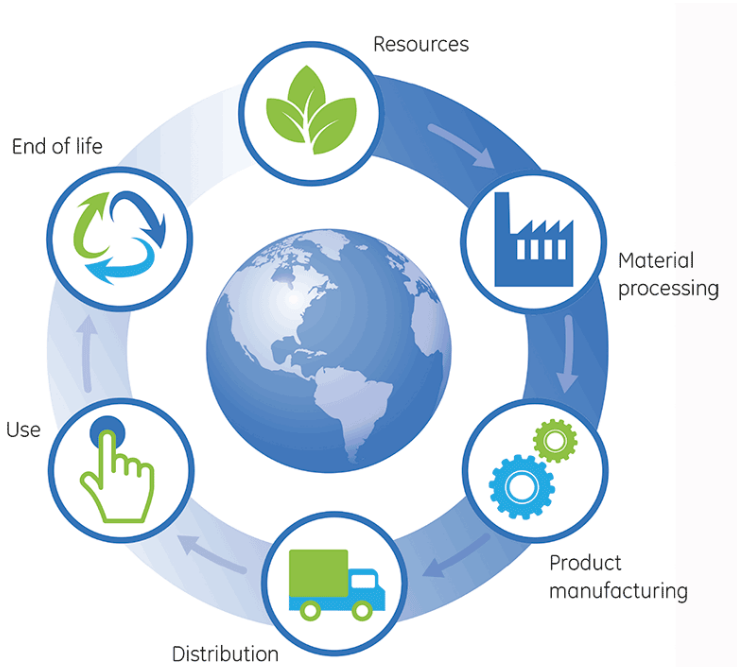

Figure 4.1 *The Life Cycle*

According to ISO standards (ISO 14040/14044), an LCA consists of four phases, see Figure 4.2:

1. Goal and scope definition
2. Life cycle inventory analysis (LCI)
3. Life cycle impact assessment (LCIA)
4. Interpretation

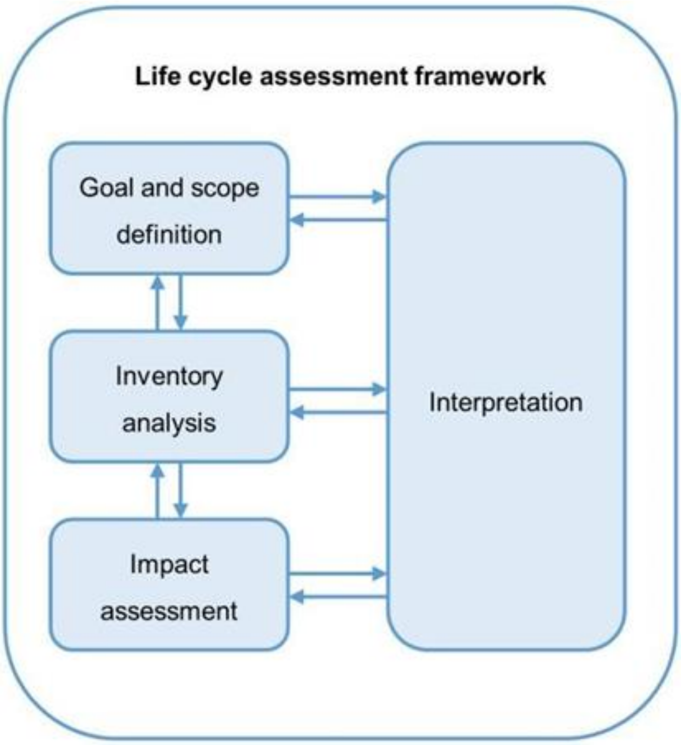

Figure 4.2 *Phases of LCA*

4.2.2 Goal and scope definition

In the goal and scope definition phase, first the aim, intended application and audience of the LCA is defined. Second, the functional unit must be defined. The functional unit is a measure of the function of the studied system, and it provides a reference point to which the inputs and outputs can be related. Third, the system boundaries must be set. This involves questions such as: is only the production process analysed (gate to gate), is the lifecycle till the output of the production process analysed (cradle to gate), or is usage and waste treatment included in the analysis (cradle to grave)? Furthermore, it is necessary to define what processes are included and which are outside the scope. The answer depends on the question how large the expected effects of the process are compared to the total effects. This is called the materiality of an effect. In addition, the temporal, geographical and technical scope must be defined. Finally, if multiple products are created by one process, it is necessary to divide the environmental impacts from the process between them. The International Reference Life Cycle Data System (ILCD), a common basis for consistent, robust and quality-assured life cycle data and studies by the EU, prefers system expansion as a methodology for taking care of multiple outputs (JRC and IES 2010).

System expansion considers expanding the product system to include the additional functions related to the co-products. Thereby including the additional functions related to the co-products and modelling the resulting changes (substitutions) in the product system. Especially by including the reduction in supply of the same product from the marginal supplier to the market for the co-product.

For example, if wheat is produced, straw is produced as co-product and this straw may be used to produce heat and electricity. The environmental effect of wheat production is then calculated as the total environmental effects of the agricultural activities to produce wheat and straw together minus the savings on environmental effects from heat and electricity production elsewhere. Hence, the system is expanded to include the processes for electricity and heat production which straw displaces.

When system expansion is not possible, allocation can be used. For example, it is stated that 80% of the total effect is allocated to wheat and 20% to straw based on the relative market values. One may allocate based on underlying physical relationships such as mass or energy, or on economic value. In practice, most LCAs use this less preferred method because it is easier to implement.

4.2.3 Life Cycle Inventory Analysis (LCI)

The second phase in an LCA is called the inventory analysis. The inputs and outputs to the system (energy and material uses and releases to the environment) are identified and compiled. The LCI is the listing of all resources consumed and emissions linked to a specific process. For each process, a listing is made of inputs used and outputs generated, where emissions are called outputs, and inputs may be economic inputs with an inventory of resources and emissions in the background, or resources. These inputs and outputs can be added to the flow diagram (Figure 4.3). In the inventory phase, one has to collect data about the relevant inputs and outputs and convert them to the functional unit defined in the scoping phase (for example per kg product produced).

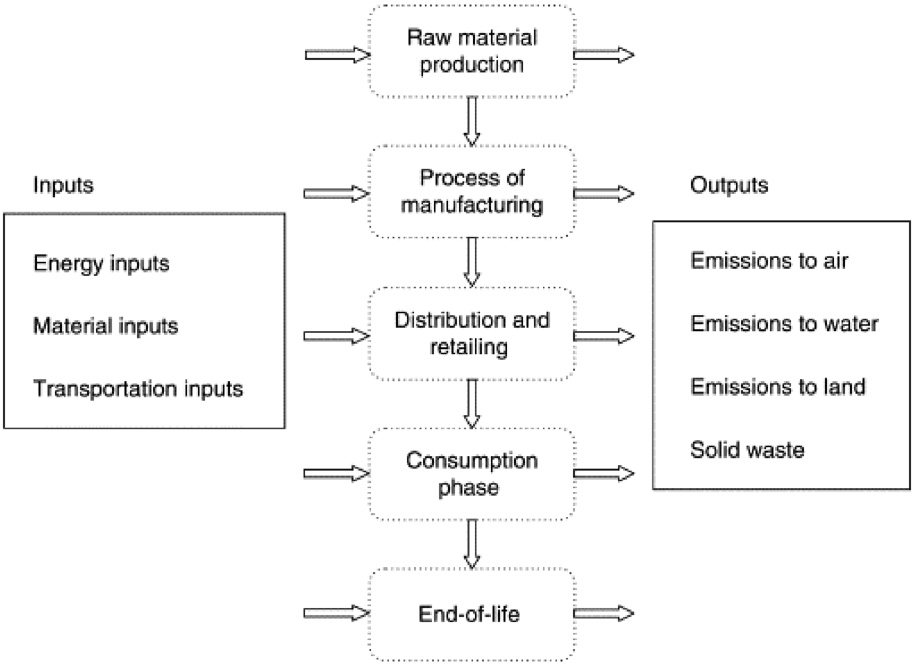

Figure 4.3 *Linking inputs and outputs to the life cycle phases*

4.2.4 Life Cycle Impact Assessment (LCIA)

The third LCA phase is the translation of the inventory of emissions and resource uses into environmental impacts. LCIA methods translate the emissions and resource extractions into a limited number of environmental impact scores. The emissions and resources of the inventory are assigned to the corresponding impact categories and then converted into quantitative impact indicators using so-called characterisation factors. Impact assessment method developers use the environmental mechanism as a basis for determining the characterisation factors. There are many LCA impact assessment methods available, and each can contain several impact categories (such as land-use, climate change, water use, eutrophication, ecotoxicity). Some methods, such as ReCiPe, have both mid- and endpoints. Endpoints refer to the final outcome of an environmental mechanism. Midpoint is in between the inventory data and the endpoint. For example, the CO_2 equivalents that express the radiative forcing is a midpoint impact category regarding climate change. In the end, it is needed to calculate an endpoint indicator (e.g., impact on biodiversity or human health).

Both levels have advantages and disadvantages. On midpoint level, a higher number of impact categories is differentiated, and the results are more accurate and precise compared to the endpoint level because the causal relations are more straightforward. Meanwhile, the endpoint level provides better information on the environmental relevance of the environmental flows. Endpoint categories are issues of environmental concern and called Areas of Protection (AoP) in LCA. Three AoPs are human health, ecosystem quality and resource scarcity. Most LCA studies are limited to midpoint impact categories, and therefore the loss of biodiversity is rarely assessed.

The midpoint indicators give direct information on environmental effects. For example, all emissions that influence climate are brought into one impact category called global warming/climate change and formulated in one unit, CO_2 equivalents, by having the same global warming potential (GWP). The GWP expresses the amount of additional radiative forcing integrated over time (20, 100 or 1,000 years) caused by an emission of 1 kg of greenhouse gas (GHG) relative to the additional radiative forcing integrated over that same time horizon caused by the release of 1 kg of CO_2. A period of 100 years is standardly taken for GWP. This already shows that aggregation requires some value judgments, in this case that the relevant period is 100 years and that it is irrelevant whether global warming happens now or after 100 years. These choices are only partly related with the costs that are caused by these impacts.

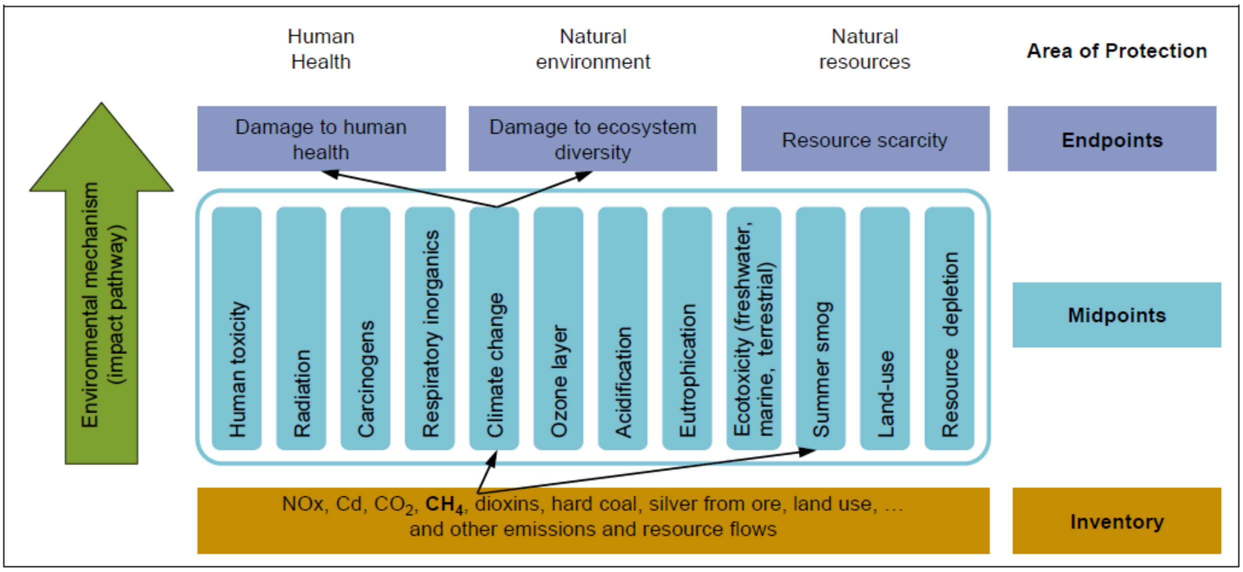

Figure 4.4 *Life cycle impact assessment*

The emissions and resources of the inventory first have to be assigned to midpoint indicators and then to endpoint indicators, although normally more steps are involved (JRC and IES 2010). This mechanism is called an impact pathway. For each step in the impact pathway parameters are needed. These parameters are called characterisation factors. These characterisation factors provide cause-effect relationships and make it possible to translate outputs from the inventory to midpoints or endpoints. For example, in ReCiPe (see Section 4.3) the cause and effect pathway for photochemical ozone formation is presented graphically as in Figure 4.4. The whole chain from emissions of nitrogen oxides (NO_x), and non-methane volatile organic compounds (NMVOC) to its effects on human health and terrestrial ecosystems is shown and parameters are estimated for each step in this chain to quantify the relationship.

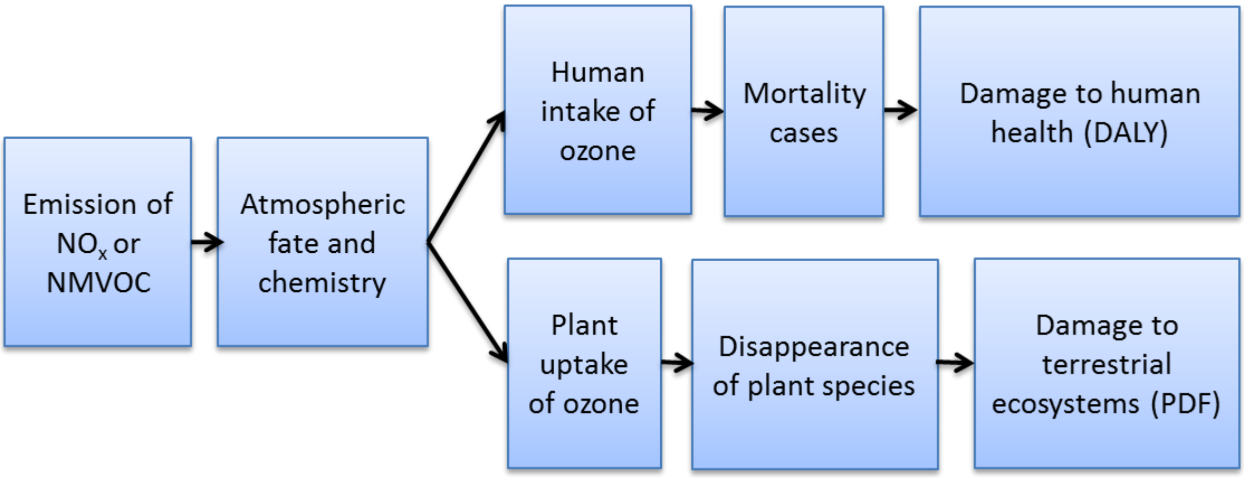

Figure 4.5 *Cause and effect pathway for photochemical ozone formation (LC-IMPACT 2023a)*

A life cycle impact assessment results in an 'environmental profile': showing the indicator results for all the predefined impact categories. This provides information on the environmental effects within the life cycle of a product and on the phases in the life cycle that contribute most to the different environmental effects.

4.2.5 Interpretation

When the impacts are calculated, the results should be interpreted carefully. First, everything must be checked on consistency and completeness. Is there double counting (consistency check), or are some relevant effects not included (completeness check)? What is the uncertainty of the outcomes, and how sensitive are the outcomes to certain input data or parameters? What is the impact of the chosen perspective? For example, in standard LCA, effects are evaluated over 100 years, while ReCiPe distinguishes a perspective of 20 years and 1,000 years. Using another perspective can have large consequences. Based on the analysis, one may draw conclusions about issues defined in the scoping phase and make recommendations on opportunities for improvement.

In the interpretation phase, results depend on the inventory data and its quality (accuracy, representativeness), the chosen allocation method selected (substitution, mass, energy, or economic allocation), the parameter values and the system boundaries that have been used. One has to look to what extent one may also derive other interpretations and conclusions based on the calculated results. If results are compared with other LCA results, careful checks should be made that the LCAs are comparable by using the same scope, allocation methods, impact categories and characterisation factors. Moreover, there should not be too much difference in probable deviations in the real world from the characterisation factors used. Furthermore, it may not be appropriate to assess local applications by using global or regional characterisation factors or data.

4.2.6 Available software tools and impact assessment methods

The most well-known software tools to do LCA analysis are SimaPro and Gabi. Moreover, there are numerous impact assessment methods, frameworks and standards available that can be applied to relate pressures with impacts in LCA among which:
- IPCC (GWP)
- USEtox method (human toxicity & freshwater ecotoxicity)
- Cumulative Energy Demand
- LANCA (land use)
- AWARE (water use)
- CML (abiotic depletion, GWP, ozone layer depletion, human toxicity, freshwater ecotoxicity, marine ecotoxicity, terrestrial ecotoxicity, photochemical oxidation, acidification, eutrophication)
- TRACI – US (ozone depletion, global warming, smog, acidification, eutrophication, carcinogenic, non-carcinogenic, respiratory effects, ecotoxicity, fossil fuel depletion)
- IMPACT 2002+ (14 midpoint, 4 endpoints (human health, ecosystem quality, climate change, and resources))
- ReCiPe (3 endpoint categories and 17 midpoint categories, see Section 4.4)
- Product Environmental Footprint
- LC-IMPACT
- IMPACT World.

The EU developed, for instance, a standard, the Product Environmental Footprint (PEF) to increase comparability of environmental impact analyses of products within certain product categories (EC 2017). In this standard, specifications are predefined for certain methodological aspects and therefore achieve better comparability and reproducibility of results. Comparability is further increased by developing Product Environmental Footprint Category Specific Rules (PEFCRs). However, biodiversity impacts were not included in this standard (EC 2017), see Table 4.1.

Table 4.1 *PEF Impact Categories, excluding biodiversity*

Indicator	Unit
Global warming potential GWP_{100}	kg CO_2 equivalents
Acidification	$molH^+$ equivalents
Freshwater eutrophication	kg P equivalents
Terrestrial eutrophication	mol N equivalents
Marine eutrophication	kg N equivalents
Freshwater ecotoxicity	CTUe
Deprivation-weighted water consumption	m^3world equivalents
Soil quality index	dimensionless (pt)no unit
Fossil resources depletion	MJ
Mineral resource depletion	kg Sb equivalents
Particulate matter	disease incidence
Ozone depletion potential	kg CFC-11 equivalents
Human toxicity, cancer	CTUh
Human toxicity, non-cancer	CTUh
Ionising radiation	kBq U^{235} equivalents
Photochemical ozone formation	kg NMVOC equivalents

Source: EC (2017).

4.2.7 Biodiversity inclusion in Life Cycle Analysis

To include biodiversity in LCA, it can be expressed as an endpoint category (ecosystem quality/health). Accordingly, the inventory data is translated to midpoint impact categories (such as land use, climate change, et cetera), which can then be translated into endpoint impact indicator results of biodiversity loss, expressed in potential disappeared fraction of species, or PDF.

In the EU standards aimed at harmonising footprint assessments among products and sectors, which are called Product Environmental Footprint (PEF) and Organisation Environmental Footprint (OEF) respectively. Biodiversity is currently not included as an indicator. It is suggested to assess biodiversity indicators separately from the other PEF impact categories (EC 2017). This would include an assessment whether biodiversity is relevant or not and if so, to report the assessment results under the section 'additional environmental information'. The PEF category rules (PEFCR) guidance, however, does not specify rules for assessing biodiversity (EC 2017).

Research into integrating biodiversity in LCA has already been initiated about twenty years ago. So far, the largest number of attempts to include biodiversity in LCA have been made by incorporating the impacts of land use as a pressure on biodiversity. Nevertheless, currently this cannot be comprehensively assessed in LCA. The UNEP-SETAC Life Cycle Initiative is driving global consensus on characterisation factors and impact indicators for biodiversity in the context of LCA (Jolliet et al. 2014). The characterisation model developed by Chaudhary et al. (2015) has recently been recommended by the UNEP-SETAC to estimate impacts on biodiversity related to land use in LCA (Verones et al. 2017). The research into other pressure points or midpoint impact categories is limited. Only a few methods exist to link these to biodiversity endpoints, as described above.

Middel and Verones (2017) integrated, for example, noise pollution impacts on marine ecosystems into the LCA framework. Their midpoint represents directly affected animals, whereas the endpoint represents the impact expressed as the number of affected harbour porpoises within a population (Middel and Verones 2017). To calculate impact in terms of the assessment endpoint, i.e. equilibrium abundance relative to undisturbed, Piet et al. (2021a) assumed semi-chemostat dynamics to model the abundance of the ecosystem component (see Section 4.6).

There is generally a trade-off between the geographical scope and taxonomic coverage. The methods considering global coverage have considerably less resolution, relying on less land use classes than the methods that focus on Europe or a part of Europe (FAO 2016). Another limitation is the biodiversity coverage. The methods currently developed focus on species richness compared with a reference state. The taxonomic coverage is often limited to vascular plants. The change in community composition across space and through time needs to be acknowledged as well (McGill et al. 2015). First attempts are made to include differences in conservation value of species in LCA. For example, Mueller et al. (2014) adapted the relative species richness method by De Baan et al. (2013) and calculated a biodiversity damage potential indicator by applying a biodiversity weighting factor based on absolute species richness, irreplaceability and vulnerability. The method developed by Chaudhary et al. (2015) is a step to include vulnerability factors for species. Furthermore, the different biodiversity levels of ecosystem or genetic diversity are much less studied.

As explained earlier, biodiversity is a complex concept and therefore assessments need to make simplifications to quantify the impact with current knowledge and data available. The methodological framework of LCA itself is posing particular difficulties. LCA has a very specific structure, which requires quantification in relation to a functional unit. It looks at aspects that flow in and out of a system. Therefore, it is problematic to include aspects that do not have a flow character (Haas 2006). LCA looks at the steady state and is usually generic in space, so a major challenge in using LCA is the inclusion of site specific and time dependent impacts (Curran et al. 2011). However, data availability to quantify impact relations is the most limiting factor.

Crenna et al. (2019) analysed the biodiversity impacts of food consumption in Europe by using the Product Environmental Footprint (PEF) standard for midpoint indicators and both ReCiPe 2008 and ReCiPe 2016 for the biodiversity endpoint indicator. It is obvious that further refinement of impact categories is needed, such as ecotoxicity, overexploitation of resources and invasive species. However, habitat fragmentation is not taken into account. PDF is far from perfect as an indicator for ecosystem quality. Furthermore, the cause-effect relations are based on a limited number of species groups and the vulnerability of species is not taken into account. The approach roughly follows Chaudhary et al. (2015), but uses global averages instead of a local approach, because the location where the impact occurs is very coarse.

4.2.8 Reflections

Even though impact assessment approaches and models integrating biodiversity into LCA are available and are sometimes integrated in LCA software tools (e.g. SimaPro), currently they are rarely used in practice (Winter et al. 2017). A reason for this could be that many stakeholders do not consider the credibility of the results strong enough (Milà i Canals et al. 2007). Moreover, there are currently hardly any incentives and pressure factors that would urge companies to assess and document the biodiversity performance of products. This calls for action at the country and/or international level.

LCA is a useful tool for assessing impacts on a large scale, but current methods and software do not have the flexibility to include specific local information. Furthermore, current LCA approaches do not differentiate between different production practices if they are on the same land use classes. This gives limitations for practical application for businesses and local policies. As a consequence, businesses and other organisations are developing methods derived from the LCA approaches but adjusting it to include the extra nuance they need (see Chapter 6).

4.3 The ReCiPe Method

4.3.1 Introduction

The main approach in LCA that includes biodiversity in a systematic way is ReCiPe. Therefore, this approach will be discussed in more detail. The ReCiPe methodology is a harmonised LCIA methodology developed in the Netherlands and commissioned by the former Ministry of Environmental Affairs. It was developed by a consortium of science organisations, such as Radboud University, CML Leiden, CE Delft and RIVM. It was published in 2009 and builds on the Eco-indicator 99 and CML methods. It includes a number of improvements, such as a biodiversity indicator. It offers results at both the midpoint and endpoint level. It was updated from the ReCiPe2008 method to ReCiPe2016 method with the aim to provide characterisation factors that are representative for the global scale instead of only the European scale (Huijbregts et al. 2016; 2017). It includes 3 endpoint categories and 17 midpoint categories (see Figure 4.6).

Midpoint indicators focus on single environmental problems, for example climate change or acidification. Endpoint indicators show the environmental impact on three higher aggregation levels. Converting midpoints to endpoints simplifies the interpretation of the LCIA results. However, with each aggregation step, uncertainty in the results increases.

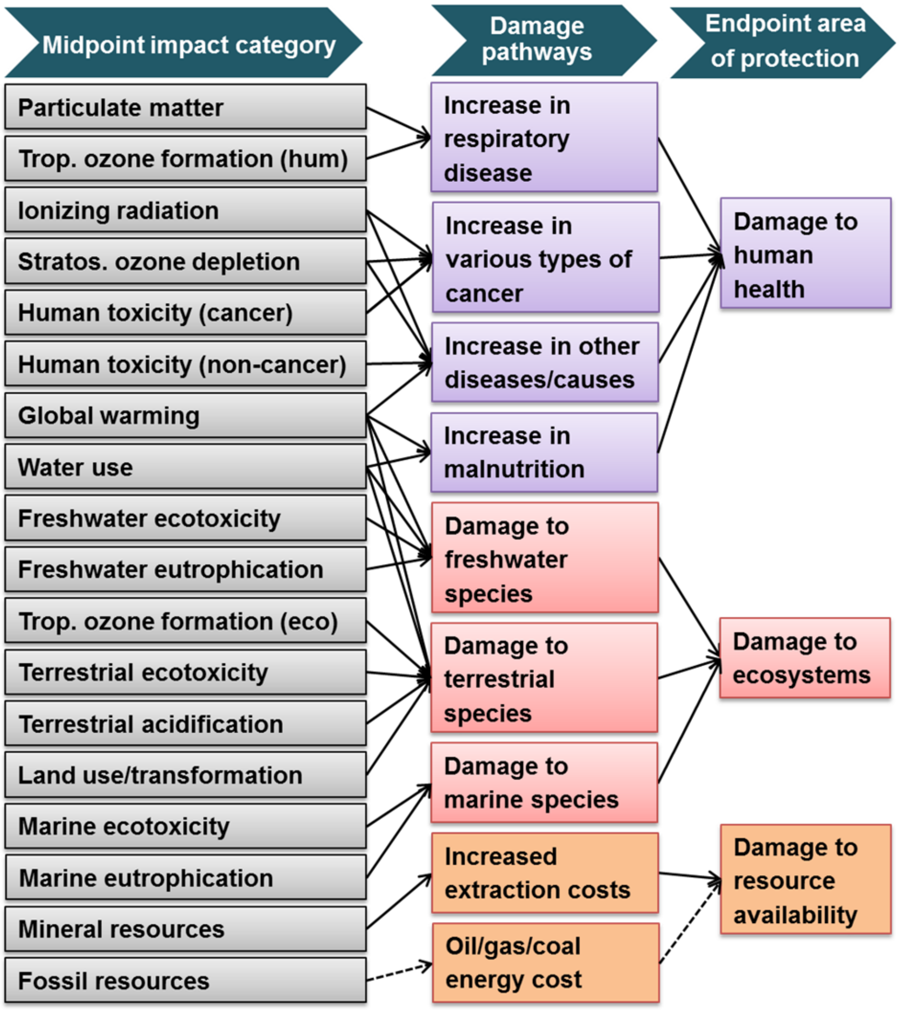

Figure 4.6 *Overview of the impact categories that are covered in the ReCiPe 2016 method*
Source: Huijbregts et al. (2016; 2017).

The inventory data is translated to midpoint impact results which can then be translated into endpoint impact indicator results in terms of impact on biodiversity with a unit describing the loss of species in a certain area or volume during a certain time, i.e. species.m^2.y or species.m^3.y in (Huijbregts et al. 2016; 2017). Consequently, biodiversity indicators are computed from the midpoint impact categories within three categories: damage to terrestrial species, damage to freshwater species and damage to marine species.

ReCiPe 2016 distinguishes six midpoint indicators that influence damage towards terrestrial species: global warming, water use, terrestrial acidification, land use/transformation, terrestrial ecotoxicity, and tropospheric ozone formation (eco). ReCiPe 2016 distinguishes four midpoint indicators that influence damage towards freshwater species: global warming, freshwater eutrophication, freshwater ecotoxicity and water use. Marine ecotoxicity and marine eutrophication are the midpoint indicators that are considered regarding damage to marine species.

4.3.2 Spatial differentiation

Country-specific characterisation factors for midpoints and endpoints were included for a number of impact categories in ReCiPe 2016, including fine particulate matter formation, photochemical ozone formation, acidification, freshwater eutrophication and water use. Particularly for the global models related to fine particulate matter formation and photochemical ozone formation, a higher spatial resolution at the global scale and with a closer spatial connection between fate, exposure and effects can further improve the reliability of LCIA. For the other impact categories, spatial differentiation has not been considered in ReCiPe 2016. The most prominent impact categories for providing regionalised results are land use and toxicity.

4.3.3 Climate change

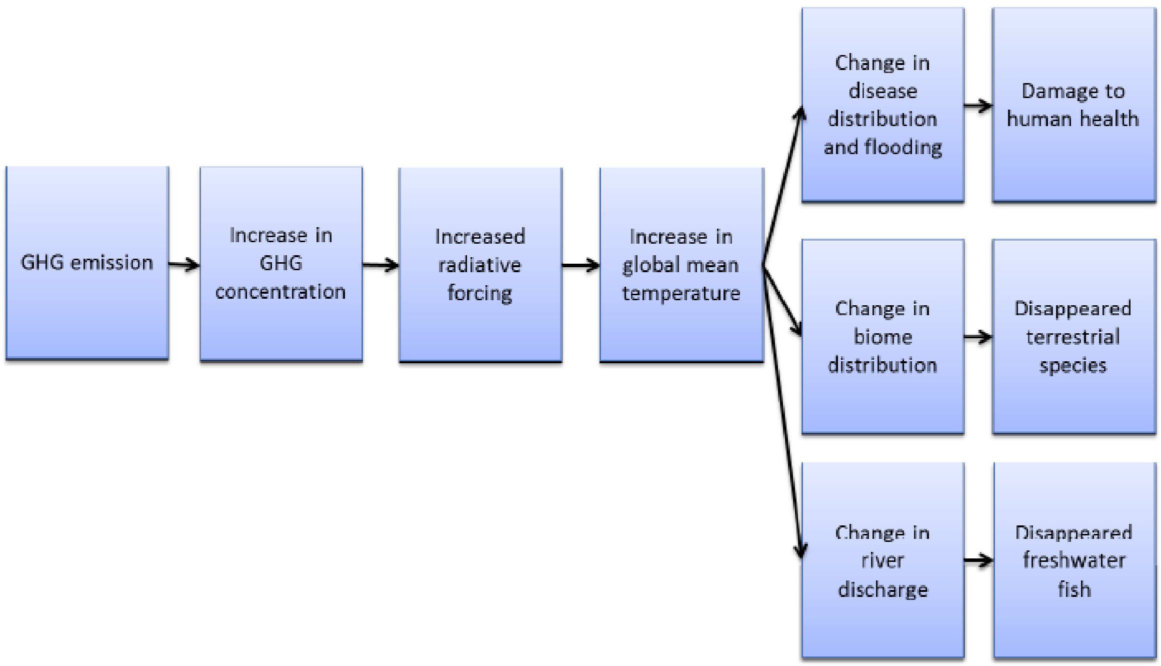

Figure 4.7 *From greenhouse gas emissions to climate change and biodiversity loss*
Source: Huijbrechts et al. (2017, p. 23).

The pathway from greenhouse gas emissions to biodiversity is through an increase in GHG concentration, resulting in increased radiative forcing, which is the difference between the energy from sunlight absorbed by the earth and energy radiated back to space. This leads to an increase in the global mean temperature. The change in temperature results in changes in biome distribution, which leads to changes in species diversity.

In the ReCiPe method all greenhouse gasses are brought to one numeraire, CO_2 equivalents, even though atmospheric lifetimes differ a lot. As a consequence, the period taken into consideration makes an enormous difference. For example, in a 20-year perspective, methane equals 84 CO_2 equivalents, but in a 100-year perspective this is 34 CO_2 equivalents and in a 1,000-year perspective it is 4.8 CO_2 equivalents. When working with long periods, feedbacks from temperature to natural carbon change may have large effects on the results. For CO_2 this is always taken into account and for the other greenhouse gas emissions only for the 100-year perspective. The reason is that the lifetime of CO_2 is very long, even more than 1,000 years, while for other greenhouse gasses the effect on radiative forcing is very small after 100 years.

The midpoint variable for climate change is the GWP. This is the additional radiative forcing integrated over time of a release of 1 kg of CO_2. The GWPs for the short time horizons are taken from IPCC (2013), while for a period of 1000 years the GWP of CO_2 is taken from Joos et al. (2013) and for the other greenhouse gasses as the multiplication of radiative efficiency per parts per billion (ppb), ppb per kg emission (together radiative forcing per kg emission), the life time of the GHG in years and (1-exponent of the lifetime divided by the time horizon). This last effect implies that it is assumed that decay follows an exponential function.

When the global warming potential is calculated, the second step is to calculate the so-called integrated absolute global temperature change potential (IAGTP). This implies that the relation between GWP and the temperature potential must be estimated. The IAGTP per kg CO_2 emission is taken from Joos et al. (2013). The temperature effect is in first instance relatively small, because the oceans take up most of the increase in temperature, but gradually the average temperature rises. Section 4.3 of Joos et al. (2013) discusses the uptake of carbon by the ocean and land biosphere. In year 20, 20% of a carbon impulse is taken up by the ocean, in year 100, 33%.

Finally, the effects on biodiversity have to be calculated. This is the temperature change multiplied by the relevant area and the effect factor. The effect factor is the PDF per degree temperature increase. The result is multiplied by the average terrestrial species density. The relation between temperature and PDF was taken from Urban (2015), who has an estimate of extinction risks at current temperature and at the temperature in the business as usual scenario of 4.3 degree increase in temperature (see Figure 4.8 below). For application in the linear LCA model, the function must be transformed into a linear function, and this is accomplished by interpolation of these two points.

Urban (2015) analysed 131 published studies on the relationship between climate and extinction risk, applying a Bayesian Markov chain Monte Carlo random effects meta-analysis. Most studies use the correlation between current climate and species distribution, while a smaller number of studies uses process-based models of physiology or demography, species-area relationships or expert opinions. The expert approach generates more pessimistic estimates with higher extinction rates than the other approaches. Most studies do not consider factors such as species interactions, dispersal differences in different landscapes, habitat degradation and evolution, although these may be key factors in explaining biodiversity. Figure 4.8 shows the relationship between temperature rise and the predicted percentage of extinction, where the rate of extinction risk increases with temperature rises. In the figure the current extinction % because of the temperature rise since industrialisation is 2.8, where the Paris target of 2-degree Celsius temperature rise in 2100 implies an average predicted extinction percentage of 5.2. The Representative Concentration Pathway (RCP) define greenhouse gas concentration trajectories defined by IPCC, where RCP 6 has 6 W/m^2 radiative forcing in 2100, and RCP 8.5 has 8.5 W/m^2 radiative forcing in 2100, where the temperature rise is the result of the radiative forcing.

Differences in definition of extinction thresholds explains differences in extinction risks in the models. Especially the moment one defines a species as extinct (the so-called extinction debt), i.e., by a habitat loss of 100% or for example 80%, really influences the results. Additionally, the dispersal assumption is influencing the outcomes, where at one extreme no dispersal of species to other regions is possible, and on the other extreme species disperse to all regions where the habitat is suitable.

The focus of Urban (2015) is on the number of species. However, the study mentions that substantial changes in abundance and distribution of species could affect ecosystems and the services they provide to humans.

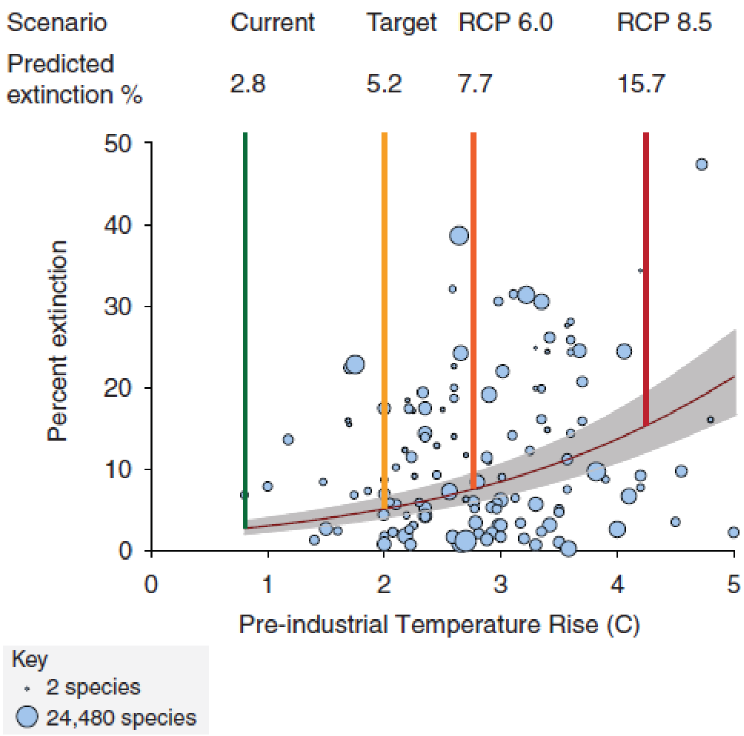

Figure 4.8 *Climate change and extinction risk in different studies*
Source: Urban (2015, p. 572).

For freshwater ecosystems, the approach is more or less the same, where the effects are calculated per river basin weighted by their volume, and relations are taken from Hanafiah et al. (2011). For health, an endpoint is calculated, based on De Schryver et al. (2009).

A paper by Nunez et al. (2019) has the same approach and is used for the GLOBIO biodiversity approach discussed in the next section, but not yet used in LCA analyses. This paper presents a meta-analysis of studies that use bioclimatic models for the analysis of the relationship between climate change and the fraction of remaining species (with 370 data points from 60 studies) and the fraction of remaining area (with 146 data points from 50 studies) for different species groups. The analysed studies are published between 1992 and 2015. The reference period is the situation around 1880, i.e., pre-industrial times, and results of studies that have a different reference period are adjusted to fit this reference period. The unit of measurement is the number of species (not the abundance of each species, as in the MSA indicator). The measure used is the fraction of the original biodiversity that remains after climate change, implying that new species in regions are not included, i.e., there is no dispersal of species.

In conclusion, both Nunez et al. (2019) and Urban (2015) estimate relations between climate change and biodiversity change based on meta-studies. However, results are very uncertain, because correlations are based on current situations instead of future developments or are model based.

4.3.4 Water shortages

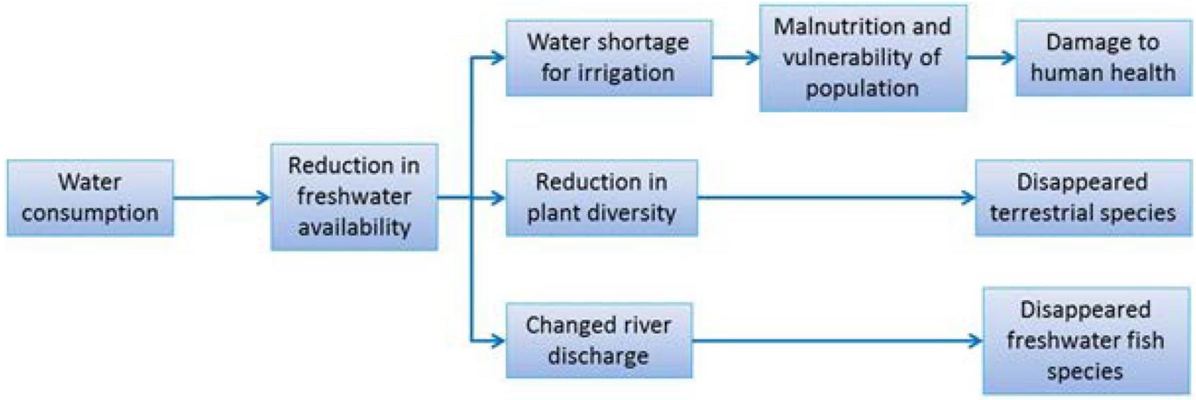

Figure 4.9 *From water consumption to biodiversity and human health*
Source: Huijbregts et al. (2016).

The starting point of the causal chain for the influence of water use on biodiversity is water consumption. Water consumption is the use of water in such a way that it is not available anymore in the original watershed. Water extraction is defined as the process of taking water from a source, but this can be either temporarily or permanently. Water consumption reduces the availability of freshwater. If part of the water is returned to the original watershed, then water consumption, defined as the reduction of water availability in the original watershed, will be less. Therefore, water requirement ratios are calculated, which is water consumption divided by water extraction. Water requirement ratios are based on AQUASTAT as discussed in Verones et al (2013). On a global level, 5 to 10% of industrial and domestic water extraction is consumptive, but for groundwater it flows in surface water and in that case, it is assumed that 100% is used.

An increase in water consumption increases water stress. To calculate water stress, first the sum of all water withdrawals is divided by the hydrological availability of water in the watershed, which is the withdrawal-to-availability index (WTA) and this index is multiplied by what they call a variation factor (VF). This variation factor is calculated per grid cell as a function of the standard deviations of monthly and annual precipitation in normal periods, with assumed lognormal distributions, and aggregated over grid cells based on the precipitation per grid cell. The adjusted WTA is calculated as the multiplication of WTA and VF. If in a watershed, the flows are strongly regulated, implying that storage of water is organised, then the square root of the calculated VF is used, because in that case variation in precipitation has less influence on water availability. It is mentioned that more storage implies more evaporation of water, but this is not taken into account. The final water stress indicator (WSI) is a logistic function of the adjusted WTA. In summary, water stress is calculated as the sum of all water withdrawals divided by the availability of water, taking into account that stress increases with the monthly and yearly standard deviation of precipitation, while the availability of regulation devices for water storage reduces this stress. The WSI is used for the endpoint human health. A decrease in the availability of freshwater implies that in first instance less water is available for irrigation. Less irrigation implies less crop production and therefore less food which results in poor countries with a low human development index (HDI) in malnutrition and an increased vulnerability of the population. The consequence is a damage to human health.

A reduction of blue freshwater availability (in lakes, rivers, aquifers and precipitation) results in less green water (soil moisture) and therefore results in less plant diversity on land and the related disappearance of terrestrial species. The NPP indicator was used for ecosystem well-being (see Section 2.4.4). NPP is correlated with plant species diversity (r = 0.6). This is based on data about watershed averages of NPP and plant species richness (Pfister et al. 2009), where a correlation was found for watersheds with fewer than 2 species per 10 km^2, but not for watersheds with higher species diversity. Finally, a translation to PDF is made, and this is multiplied by the average terrestrial species density to get species per year per m^2.

For aquatic ecosystems, less water in the rivers directly results in the disappearance of freshwater fish species in the river. In addition, it is calculated how much less water comes out of the river mouth when more water is consumed, and then the PDF is calculated based on Hanafiah et al. (2011). Fundamental is the relationship between the freshwater fish species richness, R, expressed in PDF, and the annual average river discharge at the river mouth, Q, expressed in m^3 per second, based on Xenopoulos et al. (2005). This relation can be described as $R= 4.2\ Q^{0.4}$, implying through differentiation that $dPDF/dQ = 0.4/Q$.

In summary, the empirical foundation of the relation between water uses and biodiversity requires some factors to relate water use to water consumption (where water that comes to another aquifer is regarded to be zero). Then a relation between water stress and net primary production is estimated, based on grid level information. Ultimately, a relation between PDF and net primary production is used. For aquatic ecosystems, the essential relation is between the amount of water in the river mouth and PDF.

4.3.5 Terrestrial acidification

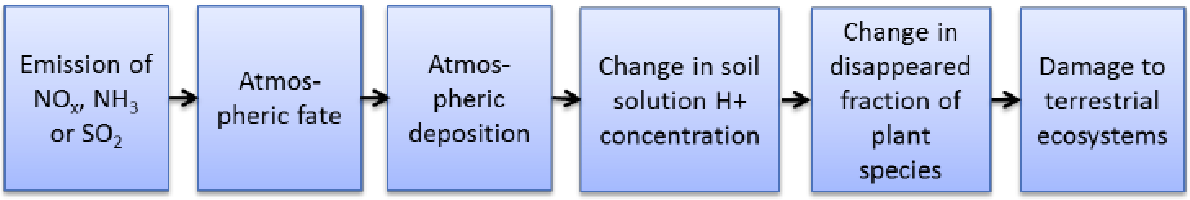

Figure 4.10 *From acidifying emissions to biodiversity*
Source: Huijbregts et al. (2016).

The basic line of argument regarding terrestrial acidification is that emissions of sulphates, nitrates and phosphates in the atmosphere change the atmospheric deposition of these inorganic substances. These emissions are transported to other regions, which is called atmospheric fate, i.e., the fate (destination) that the emissions reach in the end. The atmospheric deposition changes the acidity of the soil. All plant species have an optimal level of acidity, implying that if soil acidity changes, the species composition in ecosystems will change (Huijbregts et al. 2016, p. 57).

As a first inventory step in the LCA, the emissions per grid cell are determined. The second step translates emissions at a specific place to the change in concentration in the air (Roy et al. 2012b), using a simplified model (GEOS-CHEM). This model is based on data on emissions and depositions on a weekly base at a grid level of 2 by 2.5 degrees, so the grid level is not very detailed. Material balances are an key ingredient of the model used, meaning that it is guaranteed that all flows of material are counted for. The third step is to relate the deposition of acidifying substances in the air with acidity changes in the soil. Soil acidification is measured as soil H^+ concentration, where the choice of this measure is based on the sensitivity of the indicator and the relatively low parameter uncertainty (Roy et al. 2012a). The fourth step is to go from soil acidification to plant species diversity (in PDFs), using a dose-response function. This function is derived from logistic regression (Azevedo et al. 2013).

Azevedo et al. (2013) use field observational data from 2,409 species and soil pH in 140 studies over the last 20 years. Standardisation of pH values and species names was needed, and all information was coded in absence or presence dichotomies, where a species was considered absent if no observations were mentioned. All observations were allocated to biomes, where for each reported pH value (per 0.1 unit), the species richness was calculated as the number of species that were present. To relate this species richness to a numeraire and to focus on the species lost, the empirical potentially not occurring fraction of species (ePNOF) was calculated as 1 minus the number of occurring species in the biome at a specific pH value as fraction of the maximum number of species found in the biome (S_{opt}). The estimations were executed on sample of observation with pH values below the threshold value. The logistic regressions were explored based on this ePNOF and the pH values.

The results show a different optimal acidity for different biomes, and different sensitivities of changes in pH on species richness. This method is obviously not very sophisticated as only plant species are considered and not all species are reported. It is assumed that if species are not found in a pH range, that they are not growing there. The conversion factors used for different measures of acidity are not very precise, and the studies deal with present time, implying that adaptation strategies of plants were not considered explicitly. Finally, the estimation is based on species richness, not PDF, which only considers the species that are part of the ecosystem in optimal condition.

The characterisation factors were available at grid level, while the factors usually were presented at the national level or the global scale. Only plant species diversity is considered, while other species diversity indicators will change. If specific information on location is available, it seems logic to use the grid level data instead of the national level data. This is, however, not available in the SIMAPRO-software.

4.3.6 Land use change

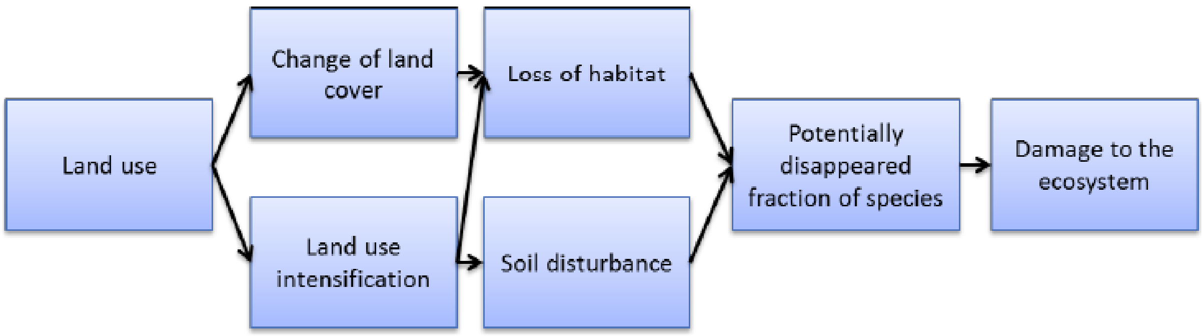

Figure 4.11 *From land use to biodiversity loss*
Source: Huijbregts et al. (2016).

The steps from land use to damage to the ecosystem are summarised in Figure 4.11 above. First, land is transformed from the current land use to the new land use. Then the land is occupied for a period of time and will require a period after use to come back to the old level, i.e. the relaxation time (Milà i Canals et al. 2007). In Figure 4.11, a difference is made between change of land cover and intensification of land use. Both have consequences for the loss of habitat, and on top of that, the change in land use intensification has consequences for soil disturbance. Subsequently, the potentially disappeared fraction of species is calculated, which is part of the damage to the ecosystem. Different reference states are possible (Koellner et al. 2013), but in the case of this PDF, the potential natural vegetation is used as point of reference. These states were approximated by using monitoring data in natural habitats in the same ecoregion (de Baan et al. 2013) or biome (Elshout et al. 2015).

De Baan et al. (2013) investigate the relative species abundance of different land uses compared to the current late-succession habitat state, which is widely used in restoration ecology and is a proxy for the potential natural vegetation. It uses the GLOBIO3 databases and a national biodiversity monitoring database of Switzerland, which has much more detail (BDM). BDM is based on a grid of 1,600 sampling points of 10 m^2 distributed evenly over Switzerland. The study uses 195 publications, providing 644 data points on land use and 254 data points on reference situations aggregated to 14 biomes. Different biodiversity indicators are calculated, as shown in Table 4.2.

Table 4.2 *Biodiversity indicators calculated for a subset of studies from the biome (sub-)tropical moist broadleaf forest*

Indicator type	Name and reference	Data requirement
Alpha diversity	Species richness	Species numbers
Sampling corrected alpha diversity	Fisher's α	Species numbers and total numbers of individuals
Diversity measure	Shannon's entropy H	List of species and their relative abundance
Abundance measure	Mean species abundance of original species (MSA)	List of species, original species and their relative abundance
Dissimilarity measure	Sorensen's S	List of species

Source: De Baan et al. (2013, p. 1219).

De Baan et al. (2013) present biodiversity measures of land occupation per biome, per land use type, per species group and per data source. Means and other statistical information about the effect on these measures can be calculated based on these measures.

The study criticises the use of one indicator for the analysis of product-related impacts of land use, where some indicators give positive effects of land use changes, while others have negative effects. On average, the effects are negative for all indicators. Moreover, one must be aware that there is a nonlinear relationship between the area sampled and species richness, implying that species richness must in some way be standardised. Because sampling methods may differ for different species, the sampling error may be very different from species to species, and this can only be partly corrected by using relative measures. Further issues are missing information on abundance, implying that MSA cannot be calculated, while it is more sensitive to changes in driving forces, the lack of a link between conservation targets and the defined reference situation, and missing information on species turnover. Furthermore, many measures focus on alpha diversity, while beta diversity is relevant. Another issue is that most indicators are compared to a natural situation, implying that all ecosystems get an equal weight. In conservation policy, some ecosystems are seen as more valuable than others. For example, ecosystems with more species or ecosystems supplying more ecosystem services are seen as more important than other ecosystems. Moreover, a focus on threatened species could be relevant in such a context. Finally, analysing the influence of land use on ecosystem functioning and ecosystem services would be relevant to complement the study.

One should be careful not to focus too much on specific species groups, because empirical studies did not find much predictive power of one species group on other species groups.

In the context of LCA, one must be aware that if the characterisation factors are regionalised, then the inventory data must be regionalised. Furthermore, in communicating results, one should be cautious to do so without emphasising the uncertainties.

Next to the occupation effect on biodiversity, the regeneration time may be taken into account as well, although currently the regeneration times seem to be very uncertain. Curran et al. (2016) analyse this regeneration time and they find that it can take a very long time, often many centuries, until the original state is reached. As a consequence, biodiversity offset policies will result in a net loss of biodiversity, while its purpose is to stop biodiversity loss.

Elshout et al. (2015) use data on species richness on croplands from 155 publications and organise them spatially to ecoregions and biomes, and biologically to taxonomic group, crop type and, if possible, management type. Based on this, formation characterisation factors were derived for the percentage loss in species richness compared to the natural state. In one approach, for each species group-crop-biome combination the median was calculated. However, for some combinations, no data were available. Therefore, in a second approach, the median was taken for a specific crop in all biomes and for all species groups, or for a specific biome for all crops and species groups, or for a specific species group for all biomes and crops. The study showed that vascular plants, which are used in many studies as a reference species group, are more sensitive to disturbance than other groups. This implies that the effect may be overestimated. Crop management was excluded in the study of Elshout et al. (2015), but field experiments show that this can have significant effects on biodiversity. Therefore, further research on crop management effects is needed.

In ReCiPe 2016, the point of reference for land use is crop land, while on the other hand the reference endpoint is the natural condition. This is based on field data on local species richness in different types of natural and human made land covers (Koellner and Scholz 2008; Koellner and Scholz 2007). The ReCiPe methodology takes into account the three stages of impact of land use, i.e., transformation, occupation and restoration. The characterisation factor when land is abandoned is half of the time until full restoration multiplied by the occupied land difference in species diversity. It may be smaller when restoration to an old stage is explicitly stimulated. In the current methodology, only two restoration times are used: for open vegetation and for forests, based on Curran et al. (2014).

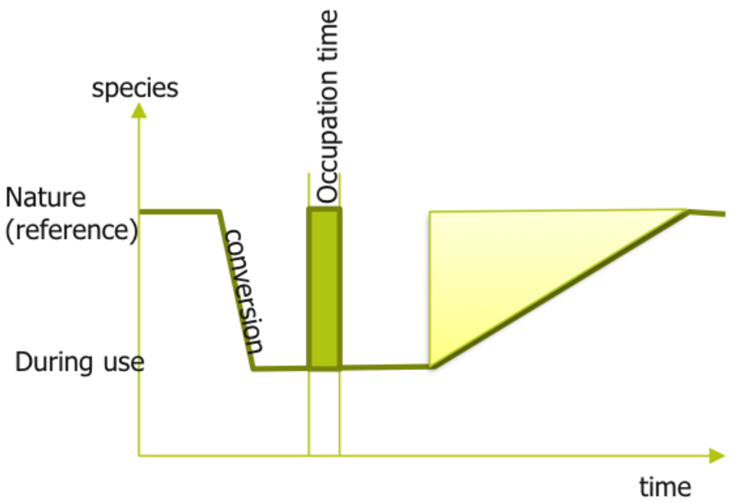

Figure 4.12 *Schematic overview of the three phases of land use and their impact on species richness*

ReCiPe provides combined ecological damage levels for 47 land use and intensity categories of the Ecoinvent database (Frischknecht et al. 2005). PDF impact factors focus on the species richness of plants and are based on data from two countries, namely the United Kingdom and Switzerland. It is assumed that the PDF impact factors that are based on data from these two countries, are relevant for the rest of Europe as well. No distinction is made between species with potentially different conservation values (e.g., Red Listed species). The study of Curran et al. (2016) is part of the UNEP-SETAC Life Cycle Initiative to build consensus on a shared modelling framework. They evaluate 31 models that are focused on biodiversity, and include models used in LCA as well as models that are currently not used in LCA. One of these models is GLOBIO3. Their conceptual model is shown in Figure 4.13. They evaluate the models on completeness of scope, biodiversity representation, impact pathway coverage, scientific quality and stakeholder acceptance. They indicated that it is not obvious that the natural state is used as a reference point. It is not value free and maybe counter-intuitive for nature policy in regions where nature and humans coevolved. In the last case, it may be better to use a culturally determined reference point. They discussed issues of biodiversity measures, including the issue of including extinction risk in the analysis.

In their analysis of data collection and modelling approaches, Curran et al. (2016) distinguish between species-area relationship (SAR), species distribution models relating environmental-climatic niches with species, meta-analysis of literature, species-energy relationship (SER, relating functional indicators such as Human Appropriation of Net Primary Productivity with species richness), indicators that measure habitat associations of species, expert judgment and regression analysis explaining threatened species density with land use composition.

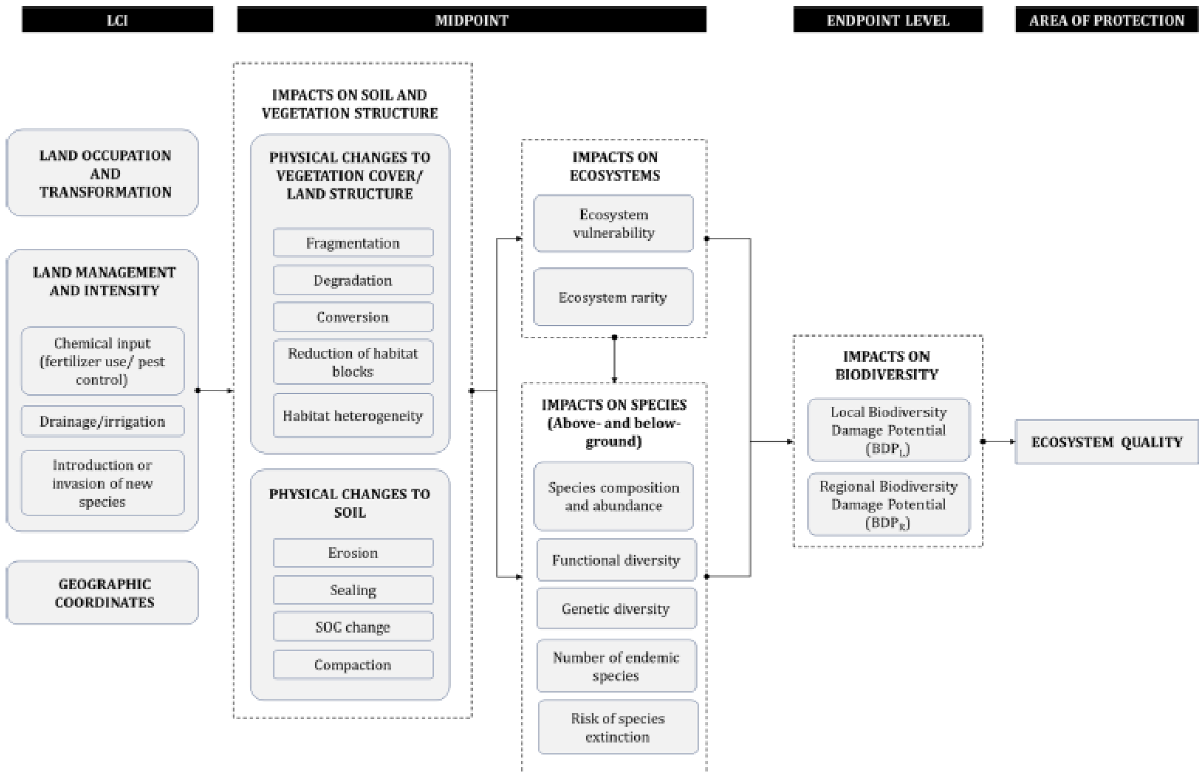

Figure 4.13 *Conceptual model on land use impacts in LCA*
Source: Curran et al. (2016).

The approach in ReCiPe is to first calculate species richness compared with cropland as a midpoint indicator and then to multiply this with a factor that may be location specific and calculates the species loss, based on the natural condition.

An interesting study on land use impacts of products is developed by Chaudhary et al. (2015). They acknowledge that global species extinctions are important, next to preserving local and regional biodiversity for resilient ecosystem delivery. The effects of land use on biodiversity are based on plot-scale biodiversity monitoring surveys comparing disturbed and undisturbed habitats. In this context, SARs must be taken into account, because the number of species increase with the area investigated. SARs predict species loss as a consequence of habitat loss. However, species often face habitat change instead of habitat loss. The classic SAR does not take this into account, while the Matrix SAR and the countryside SAR do. In the countryside SAR, the fact that some areas that are not in the original state, may nevertheless be suitable for some species, is taken into account. Furthermore, the marginal effect is relevant, not the average effect. Finally, characterisation factors for land transformation are calculated.

However, the most interesting part is the introduction of a vulnerability factor for species. First, the endemic richness in a region is the proportion of the investigated area divided by the total area where the species is living. This is then multiplied by the thread level according to the IUCN Red List of Threatened Species (based on a mapping of thread levels on a scale from 0.2 to 1). This factor is multiplied by the indication of the fraction of species that remains in the area. In this way, both the ecosystem and species level vulnerability are taken into account.

With respect to data assembled, the following information is used:
1. The local characterisation factor of the fraction of species that is left after land use change from natural circumstances to the current land use, based on five taxa, 14 biomes and 6 land use classes (de Baan et al. 2013; Elshout et al. 2014; Aronson et al. 2014)
2. Original species richness per ecoregion (804) and taxon (5) (Kier et al. 2005) with additional information from WWF Wildfinder database
3. The original natural habitat area for each ecoregion (804) (LADA database)

4. Remaining natural habitat area per ecoregion (804) and land use (Ellis and Ramankutty 2008)
5. Area per land use type (FAO-FRA), FAOSTAT
6. Z value (Drakare et al. 2006)
7. Regeneration time (i.e., time between the moment of land abandonment after use and the moment biodiversity is back at its original level) combinations of realm and biome (65), land use intensity (2) and taxon (5) (lognormal distribution)
8. Vulnerability score per ecoregion (804) and taxon (4) (IUCN, Birdlife International).

For distributions (for 1, 2 and 6), triangular distributions are used based on a minimum, median and maximum.

SARs do not take indirect effects of biodiversity loss into account, such as that when more roads are made for forestry management, they can be used for hunting. Furthermore, different land management practices are not taken into account, such as the difference between selective logging and clear-cut or conventional versus biological agriculture. In the study, the number of species is used multiplied by area, not the abundance of the species. Chaudhary and Brooks (2018) present a first attempt toward addressing the above five issues and calculate updated characterisation factors (CFs) for five broad land use types and their three intensity levels.

In conclusion, although the relation between land use and biodiversity is relatively well studied, only a very limited land use categorisation is used and relatively simple indicators such as PDF or MSA are used as dependent variable. Only recently attempts are made to broaden the scope. For example, extinction risks are included in recent methodologies and different land use management types are distinguished as different land use types. This is, however, far from sufficient to analyse the effect of organic compared with conventional agriculture on biodiversity.

4.3.7 Marine eutrophication and marine ecotoxicity

To get an idea of how the marine sector is included in LCA, Table 4.3 shows examples from effects on marine eutrophication and marine ecotoxicity. Here, no relation with biodiversity is included.

Table 4.3 *Marine eutrophication and marine ecotoxicity*

	Substance name		**emission compartment**	**kg N-eq to marine water/kg (all perspectives)**
Marine eutrophication	N		freshwater	0.30
	NH_4^+		freshwater	0.23
	NH_3		freshwater	0.24
	NO		freshwater	0.14
	NO_2		freshwater	0.09
	NO_3		freshwater	0.07
	NO_x		freshwater	0.09
	Substance name	**CAS**	**emission compartment**	**1,4-DCB eq. emitted to seawater**
marine ecotoxicity	P-CHLORONITROBENZENE	100005	urban air	1.45E+00
	4-NITROANILINE	100016	urban air	2.90E-01
	ethylbenzene	100414	urban air	4.57E-04
	styrene	100425	urban air	9.50E-05
	Benzyl chloride	100447	urban air	8.92E-02
	benzyl alcohol	100516	urban air	2.85E-03
	benzyldehyde	100527	urban air	2.34E-02
	HEXAMETHYLENETETRAMINE	100970	urban air	3.86E-05
	ANILAZINE	101053	urban air	5.29E+01

4.3.8 Reflections

The ReCiPe methodology provides a systematic way to relate different midpoint pressure factors to biodiversity. However, the method is still very rough and speculative, and uses only the PDF as indicator. In the description of different empirical relationships and indicators recent research is mentioned.

4.4 The GLOBIO methodology

4.4.1 Introduction

GLOBIO is formally not part of LCA but provides empirical relations that can be useful in the context of LCA by relating pressure factors, midpoints in LCA terms, with a biodiversity endpoint. The model uses MSA as biodiversity indicator, which is very similar to the PDF that is used in ReCiPe.

The Netherlands Environmental Assessment Agency (PBL), together with other research partners, has developed an approach for global biodiversity assessments, the GLOBIO model approach (Alkemade et al. 2009). It was developed for assessments at the global level, using a number of proximate drivers (or pressures) as a crude measure for ecosystem quality and dose-response functions to assess the impact of these pressures. These relationships between pressures and species abundance (dose-response relationships) are based on extensive systematic literature reviews and meta-analysis.

The driving forces (pressures) incorporated in the model are:
- land use (MSA_{LU}) (Alkemade et al. 2009)
- nitrogen deposition (MSA_N) (Bobbink et al. 2010)
- infrastructure development (MSA_I) (Benítez-López et al. 2010)
- fragmentation (derived from infrastructure) (MSA_F) (Verboom et al. 2014)
- climate change (MSA_{CC}) (Arets et al. 2014; Nunez et al. 2019).

If there would be no interaction among the drivers for a specific location, the overall MSA may be calculated by multiplying the MSAs related to the different drivers (Croezen et al. 2014):

$$MSA_i = MSA_{LU(i)} \cdot MSA_{N(i)} MSA_{I(i)} \cdot MSA_{F(i)} \cdot MSA_{CC(i)}$$

In GLOBIO version 4, the methodology cannot be combined with other environmental impacts and types of damages, such as impacts on human health, and cannot be combined with local pressures impacting biodiversity, such as water shortages. However, in its current shape it is quite comparable with the PDF factors applied in LCAs. Through the use of dose-response functions that relate pressure factors with impacts on biodiversity, it is possible to use this approach in an LCA context. Possibly, adjustments need to be made to be able to link to the specific midpoint indicators in LCA.

In previous assessments, the GLOBIO approach worked well in an LCA context with the use of midpoint indicators from either LCAs or company specific input data and dose-response functions to asses impact on biodiversity, see for example (Van Rooij et al. 2016; Arets et al. 2017; CDC Biodiversité 2019).

4.4.2 Role of productivity

In an assessment of the use of the related MSA.ha as a footprint indicator, Arets et al. (2017) indicated the importance of land productivity in the land-use component of the footprint assessments. Productivity plays a direct role in determining the size of the area that is needed to realise a certain functional unit of product (for example, a kg or litre of a certain product). As the area needed determines the biodiversity footprint to a large extent (MSA.ha), this has a large effect on the end result of the footprint assessment. If the productivity doubles, the required area is halved. In addition, the land use related biodiversity footprint is to a large extent determined by the quality of the area (in MSA terms). This depends on the intensity of the land use. Because more intensively managed areas usually have higher productivity, productivity plays a role on the quality side.

Due to the dual role that land productivity plays and the use of a functional unit, one must not only look at the biodiversity footprint in MSA.ha when comparing alternatives, but get a grip on the local impact. A highly productive, intensively managed land system can have a lower biodiversity footprint to produce a certain amount of product x. For example, because land productivity has a stronger effect than the negative effect on nature (MSA). An alternative that realises a higher naturalness (MSA) locally, but that covers a much larger area for the production of the same amount of product x, can thus score relatively poor. A higher overall biodiversity footprint can nevertheless lead to a more sustainable living environment at the level of local landscapes. That proved to be an important factor in the milk sector case in the biodiversity footprint report of Van Rooij et al. (2016). For social or political reasons, it may be desirable to maintain a higher natural value in certain areas at the expense of productivity.

In line with this, a lower land productivity with a higher biodiversity leads to a production system that can provide more other services and that has a sustainable production, which can be maintained over longer time periods. For example, intensive agriculture can in some cases lead to depletion of the soil, reducing productivity and ultimately making production impossible. This time factor is not included in the current GLOBIO method. Due to the more generic use of dose-response relationships, it is currently not possible to incorporate the effects of very specific nature-friendly measures with the applied GLOBIO method and dose-response relationships. To improve that, more studies on the long-term effects of nature-friendly/nature-inclusive measures are needed to be able to assess specific dose-response relationships for such systems.

4.4.3 Extensive land use

The footprint methodology based on GLOBIO has not yet been fully developed for highly extensive land use systems in natural or semi-natural ecosystems. For example, for calculating the land use related footprint in large semi-natural grasslands, or for an extensive logging system in a semi-natural forest. If the total area used for grazing or harvesting is used, the footprint indicator will give a large impact only because of the large area needed, while the impact is only very local. Aspects of spatial and temporal scales would need to be better included in this approach.

4.4.4 Restoration of biodiversity

GLOBIO does not use a transition or recovery period between two consecutive land use types. The conversion from more natural systems to fewer natural ones can occur suddenly, but conversely that is not the case. This makes it difficult to determine the MSA land use value of a (semi) natural area that recently had more intensive land use. The biodiversity of such an area will generally increase slowly. The speed with which recovery will take place will depend on several factors and may differ per location. This speed depends on the presence and accessibility of the area in question for original species.

GLOBIO lacks an age correction or recovery factor for older plantations with a longer rotation, that generally have a higher biodiversity. However, correction factors could be determined by re-subjecting the current data files used for the dose-response relationships to a meta-analysis for the above factors.

4.4.5 Economic allocation

When correcting the land use related footprint for multiple land use, the effects of the different land uses must be allocated to the different products produced on it. Sometimes, the value of the different products is used to determine the allocation of environmental effects of production of a specific commodity, for example for the value of meat of dairy cows when determining the footprint of milk. However, this is not correct, as meat production is a by-product of milk production, and therefore the prevented environmental effects as a consequence of producing the meat from milk cows instead of other meat, must be calculated for a correct allocation. Allocation is even more difficult to determine for the social value of a semi-natural natural landscape that is partly used by a company. For example, there may be nature conservation and recreation, for which it is not always easy to determine what land use has which effects.

The effect in the use of for instance rangelands, where animals live and graze in large (semi-) natural areas, is more difficult to assess. In such a case, the impact is only very local and will not affect the whole range area, while the range area may be very large. Especially with a large uncertainty in the effect function per unit of area, the overall effect may be overstated as a result of the large range area. More research would be needed to deal with such circumstances, focusing on more explicit quantification of the scale of the effects and trade-offs with ecosystem services that may benefit of the grazing with low intensities.

GLOBIO4 uses a detailed land use database and can run at a spatial resolution of 10 arc seconds (about 300 meters). In all databases used characteristics of parcels of land are related to the abundance of species. For each pressure factor, the abundance of species in disturbed parcels of land is compared with the abundance of species in an undisturbed situation for different natural conditions. This comparison is meant to be within the same study, because otherwise the data may not be comparable. For each species, the abundance in the disturbed situation is divided by the abundance in the undisturbed situation and truncated to 1 in case the abundance is larger in the disturbed situation. The arithmetic mean over all species is the mean species abundance (Alkemade et al. 2009; Schipper et al. 2020).

4.4.6 Included impact categories

The following impact relationships are included in GLOBIO4:
- Climate change
- Atmospheric nitrogen deposition
- Land use
- Impact of habitat fragmentation
- Road disturbance
- Hunting

GLOBIO4 focusses on terrestrial plants and warm-blooded vertebrates (birds and mammals) because for these species groups sufficient data are available to establish response functions. For land use and habitat fragmentation, the PREDICTS database is used (Hudson et al. 2017). For nitrogen the article by Midolo et al. (2019) is used, and for climate Nunez et al. (2019). Because the factors are analysed independently, for the parcels of land analysed the other pressures must be either very small or equal between treatment and control situation. In other words, pressure-impact relationships must be based on representative random samples of the community (Schipper et al. 2020).

The impact of land use on biodiversity is always taken into account, while some or all of the other factors may not be relevant for specific land uses. For example, for cropland the land use effect dominates all other pressure factors. In other cases, the relevant relationships are multiplied with each other, where factors are only taken into account as far as they are above the tolerance limit. Multiplication assumes that the pressures act independently and tolerances of organisms for different pressures are not correlated (Schipper et al. 2020).

Climate change
Schipper et al. (2020) provide in their supplementary material background on the calculations of climate change impacts on biodiversity in GLOBIO4. They use data from a meta-study by Nunez et al. (2019) from 31 bioclimatic studies with 135 values for biodiversity. They use the fraction of remaining species (FRS) as biodiversity indicator, which is the ratio between the number of species after climate change and the original number of species in the location. This is basically 1–PDF and is used as a proxy for MSA. The FRS of a region is calculated as the average FRS for a number of locations within a region. The global mean temperature increase in each climate scenario was used as the explanatory variable in a mixed beta regression model per taxonomic group.

Atmospheric nitrogen deposition
The relation between nitrogen deposition and biodiversity can be investigated by field experiments or by the analysis of observations of nitrogen deposition and biodiversity at different points in space or time. (Midolo et al. 2019) analysed 115 experiments in 85 studies and used four biodiversity metrics, being the change in

the logarithm of mean species richness, the change in the logarithm of the mean species abundance, the same truncated at 1 when abundance increases, and the geometric mean abundance.

The results show that different indicators of nitrogen deposition perform differently. For example, addition in the form of NO_3 may have different effects than the addition of NH_4. Next to N addition, other factors are relevant. Examples are the duration of the addition, the plot-size, the mean annual temperature, the mean annual precipitation, and cation exchange capacity (the number of cations that can be retained on soil particle surfaces, where a cation is an ion with a positive charge, i.e. with less electrons than protons; an indication of the ability of soil to buffer N-induced acidification). Only mean annual temperature had a significant interaction effect with N-addition. The study focuses on site-level biodiversity only, implying that only alpha diversity can be assessed.

In GLOBIO 4, the data of 37 studies that report individual species abundance are used to calculate 89 MSA values as input for a mixed beta regression model. In contrast with Midolo et al. (2019), a model without interactions between mean annual temperature and nitrogen addition performed best according to the Bayesian information criterion.

Land use
Schipper et al. (2020) use the monitoring data from the 2016 release of the PREDICTS database (Hudson et al. 2017). The following land use types are distinguished: 'Primary vegetation', 'Mature secondary vegetation', 'Intermediate secondary vegetation', 'Young secondary vegetation', 'Secondary vegetation of indeterminate age', 'Plantation forest', 'Cropland', 'Pasture', or 'Urban' as well as the following land use intensities: 'Minimal use', 'Light use', 'Intense use', and 'Cannot decide' Hudson et al. (2014). For GLOBIO4, the reference level for biodiversity was derived from studies on abundances of plant and warm-blooded vertebrate studies in primary vegetation with minimal use. The GLOBIO4 land use categories were mapped to those in the PREDICTS database, and when more sites were sampled in a land use category, the average abundance per species was calculated over the samples sites. For each species the abundance ratio was calculated as the abundance in the disturbed sites divided by the abundance in the reference sites and truncated at 1. The MSA values were calculated as the average of the truncated abundance ratios per land use type, species group and dataset. In this way for plants 55 MSA values from 32 studies were calculated and for warm-blooded vertebrates 85 MSA values from 48 studies. A mixed beta regression model was estimated on this dataset, where datasets within studies were random effects and GLOBIO land use classes fixed effects.

Basically, for this biodiversity measurements, one needs information on different species occurring in combination with certain land uses. To separate land use from other effects, one must be certain that other factors are more or less the same, or one must explicitly control for them.

Impacts of habitat fragmentation
For habitat fragmentation, GLOBIO4 uses data from the PREDICTs database. In this database, patch size is included, where a patch size of '-1' means that the patch size was too large to be measured (Hudson et al. 2014). In datasets with multiple patches, of which at least one more than 10,000 ha, the largest patch was seen as having no fragmentation problems. Based on this, 39 MSA values from seven studies were calculated in the same manner as with the land use classes. A mixed beta regression model was estimated on the basis of these MSA values, with the study as random intercept and the $\log_{10}$-transformed patch size (ha) as explanatory variable. According to Schipper et al. (2020) this is a fixed effect, but that does not seem in accordance with having numerous different patch sizes. Kuipers et al. (2019) discuss pathways to go a step further and provide a literature overview of attempts to include habitat fragmentation in LCA.

Road disturbance
GLOBIO4 includes the effect of road disturbance by measuring the distance to the closest road. Schipper et al. (2020) use the database by Benítez-López et al. (2010) as a starting point and extend it with observations from more recent literature. The focus is on studies on wildlife populations in areas adjacent to roads. Papers that include secondary impacts of road construction, such as hunting and human intrusion, were excluded from the database. The reference populations were the populations at a large distance from roads, and based on that, 204 MSA values from 34 studies were assembled. Again, a mixed beta regression

model was estimated, with study as nested random intercept and the log_{10}-transformed distance to the road was used as fixed effect.

Hunting
Hunting intensity is in general higher in the proximity of hunters' access points (Abernethy et al. 2013; Benítez-López et al. 2017), so the distance to a settlement is used as the explanatory variable for animal abundance (Benítez-López et al. 2019). Only studies that assessed the impact of hunting on wildlife abundance were included and each study has to provide data on at least one hunted and one control area. Studies that included other effects, such as areas that were logged, were not included, and studies that included areas where animal populations were managed for recreational hunting were excluded. In the calculation of MSA, all species were included irrespective if they are hunted. About 20% of the mammal species in the dataset had a body mass of less than 1 kg and therefore were normally not hunted (Ripple et al. 2016). About 38% of the bird species were normally not hunted (Redford 1992).

465 MSA values from 125 studies were used for a mixed beta regression with dataset within study as nested random intercept, country as crossed random intercept (to account for possible cultural differences in hunting; Benítez (Benítez-López et al. 2019) and the log_{10}-transformed distance to hunter'' access point (km) as fixed effect. In contrast with Benítez-López et al. (2019), inclusion of the log_{10}-transformed population density values from the Gridded Population of the World dataset (CIESIN 2017) generated better estimates than the estimates without population density.

4.4.7 Reflections

Both ReCiPe and GLOBIO use empirical relationships between pressures and biodiversity. Both use an indicator that is related with the fraction of biodiversity that is lost compared to a reference state. And both combine the effects of different pressure factors. However, while ReCiPe assumes additive relationships between the factors, GLOBIO assumes a multiplicative relationship. However, for relatively small changes this will not give a very different result in the outcome. Moreover, the pressure factors that are included differ, which basically implies that both approaches can learn from each other.

4.5 New developments in LCIA

4.5.1 Introduction

In the last years, efforts on building new LCIA methods besides the ReCiPe 2016 have come to the surface. The main reason for these efforts is to improve regionalisation and spatial differentiation in LCIA. In this section, two of the most prominent new LCIA methods are addressed, namely LC-IMPACT (see 4.5.2) and IMPACT World+ (see 4.5.3).

4.5.2 LC-IMPACT LCIA method

4.5.2.1 Introduction

The LC-Impact methodology has been developed in an EU FP7 project, in a collaboration between 14 partners (LC-IMPACT 2023). The aim is to provide a global LCIA methodology for the three main areas of protection (human health, ecosystems, resources), including spatially differentiated information, where applicable. It provides characterisation factors at the damage level for eleven (mid-point) impact categories related to three areas of protection (human health, ecosystem quality, natural resources), see Figure 4.14. Eutrophication covers both freshwater and marine eutrophication and toxicity covers freshwater, marine and terrestrial toxicity.

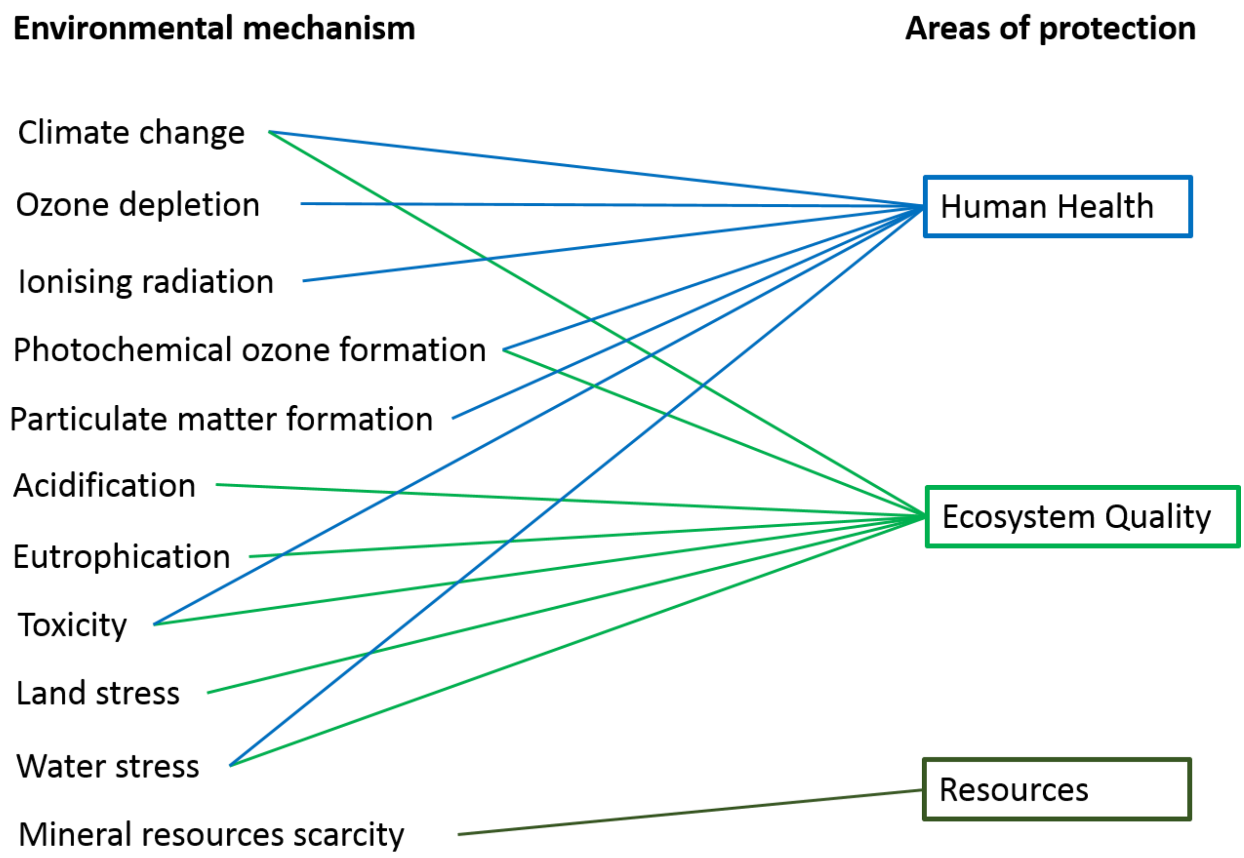

Figure 4.14 *LC-IMPACT method impact categories and areas of protection*
Source: Verones et al. (2020).

4.5.2.2 Ecosystem quality

The unit for ecosystem quality is measured with the PDF. Relative global species loss per unit of emission or extraction is calculated by the product of a vulnerability score VS, an exposure factor XF, as well as a fate factor FF and an effect factor EF. The VS varies between 0 and 1. A VS of 1 means that the species is highly threatened or probably endemic, while lower scores denote less vulnerable species (Verones et al. 2020).

$$CF_{ecosystem\ quality} = FF \cdot XF \cdot EF \cdot VS$$

With CF= the characterisation factor of ecosystem quality, VS=vulnerability score, XF=exposure factor, FF=fate factor and EF=effect factor.

What is special in LC-IMPACT compared to other LCIA methods, is that the CF quantifies the relative global species loss by putting the regional species loss in perspective of the global species pool. This is done for one or more taxa (fish, mammals, birds, amphibians, reptiles, and/or plants), depending on the data availability per impact category. The plants and animal taxa are given equal weight. Contributions of several animal taxa are included. The procedure and details for calculating taxon-specific and global vulnerability factors are described in Verones et al. (2020).

Three ecosystem types (terrestrial, freshwater, and marine) are distinguished. Impacts on terrestrial ecosystems are covered in five categories (climate change, photochemical ozone formation, terrestrial acidification, terrestrial ecotoxicity, and land stress). Impacts on freshwater ecosystems are represented in four impact categories (climate change, water stress, freshwater ecotoxicity, freshwater eutrophication), while the marine ecosystem is covered in two categories (marine eutrophication and marine ecotoxicity). Impacts of marine, freshwater, and terrestrial ecosystems, in terms of PDF, can be directly added under the assumption that these ecosystems have an equal value.

4.5.2.3 Spatial differentiation

For impact categories where impacts take place at a global scale only (climate change, ozone depletion, mineral resources extraction, ionising radiation), the characterisation factors are given on a global level only. For categories with a regionalised scope, such as water and land stress, spatially differentiated characterisation factors are derived. The level of spatial detail is varying greatly between the different impact categories, depending on the relevant scale of impact and the data availability, as is shown in Table 4.4. All spatially differentiated characterisation factors are available on a country, continental and global level to facilitate the link with standard LCIA data.

Differences between generic and regionalised impacts vary up to two orders of magnitude for some of the selected impact categories, highlighting the importance of spatial detail in LCIA (Verones et al. 2020).

Table 4.4 *Spatial resolution of the impact categories for the different parts of the environmental mechanisms*

environmental mechanism	Spatial resolution fate factor	Spatial resolution effect factor	Spatial resolution characterization factor
climate change (ecosystems)	none	none	none
climate change (human health)	none	none	none
stratospheric ozone depletion	none	none	none
ionising radiation	global values for air, freshwater, marine	none	global values for air, freshwater, marine
photochemical ozone depletion (ecosystems)	56 world regions (averages of base run of 1°x1°)	none	country level
photochemical ozone depletion (human health)	56 world regions (averages of base run of 1°x1°)	none	country level
particular matter formation	56 world regions (averages of base run of 1°x1°)	none	country level
terrestrial acidification	615'888 three dimensional compartments	2° x 2.5°	2° x 2.5°
freshwater eutrophication	0.5° x 0.5°	biogeographical habitats	0.5° x 0.5°
marine eutrophication	Country to large marine ecosystems (233 spatial units)	66 large marine ecosystems (5 climate zones)	Country to large marine ecosystems (233 spatial units)
freshwater ecotoxicity			sub-continental
human toxicity			sub-continental
marine ecotoxicity			sub-continental
terrestrial ecotoxicity			sub-continental
land stress	ecoregions	ecoregions	ecoregions
water stress (ecosystems)	more than 20'000 individual points	more than 20'000 individual points	0.05° x 0.05°
water stress (human health)	watersheds (11'000 units)	country level	watersheds (11'000 units)
mineral resources extraction	none		none

4.5.2.4 Reflections

Today's commercially used LCA software tools and LCI databases generally do not handle spatially differentiated data well, since they are mostly restricted to country scales. This hampers the applicability of this method. Only one software tool, Brightway 2, is able to handle the fully spatially differentiated CFs.

There are several uncertainties due to limited knowledge of the exact impact mechanisms. Therefore, new developments are planned to include new impact pathways and improve the already covered impact pathways. LC-IMPACT does not provide midpoint characterisation factors, unlike ReCiPe 2016, since midpoint factors are not available yet in a consistent way across all included impact categories.

LC-IMPACT uses the global PDFs as metrics for biodiversity impacts, whereas ReCiPe2016 combines absolute species loss at the local, regional, and global scale, using species.yr. In addition, rather than the cultural perspectives used in ReCiPe, LC-IMPACT provide four sets (depending on the impact category) of characterisation factors, distinguishing between time horizons and different effects.

4.5.3 IMPACT World+ LCIA method

4.5.3.1 Introduction

IMPACT World+ is the update of the IMPACT 2002+, LUCAS, and EDIP methods. It covers the entire world at different levels of spatial resolution, to analyse the magnitude of characterisation results for each impact category at the global scale and to quantify the relative importance of spatial variability compared to the overall spread of characterisation factors. It provides CFs at four hierarchical levels of spatial resolutions: global default (non-spatially resolved), continental, country, and native resolutions. At midpoint level, 18 impact categories are recommended (see Figure 4.15). For damage level, it provides the possibility to adopt an area of protection (AoP) or an area of concern (AoC) viewpoint. The three AoPs are human health, ecosystem quality and resources & ecosystem services. The AoC viewpoint considers grouping and expressing damage level impact categories in terms of water-related damages, carbon-related damages, and the rest of damages.

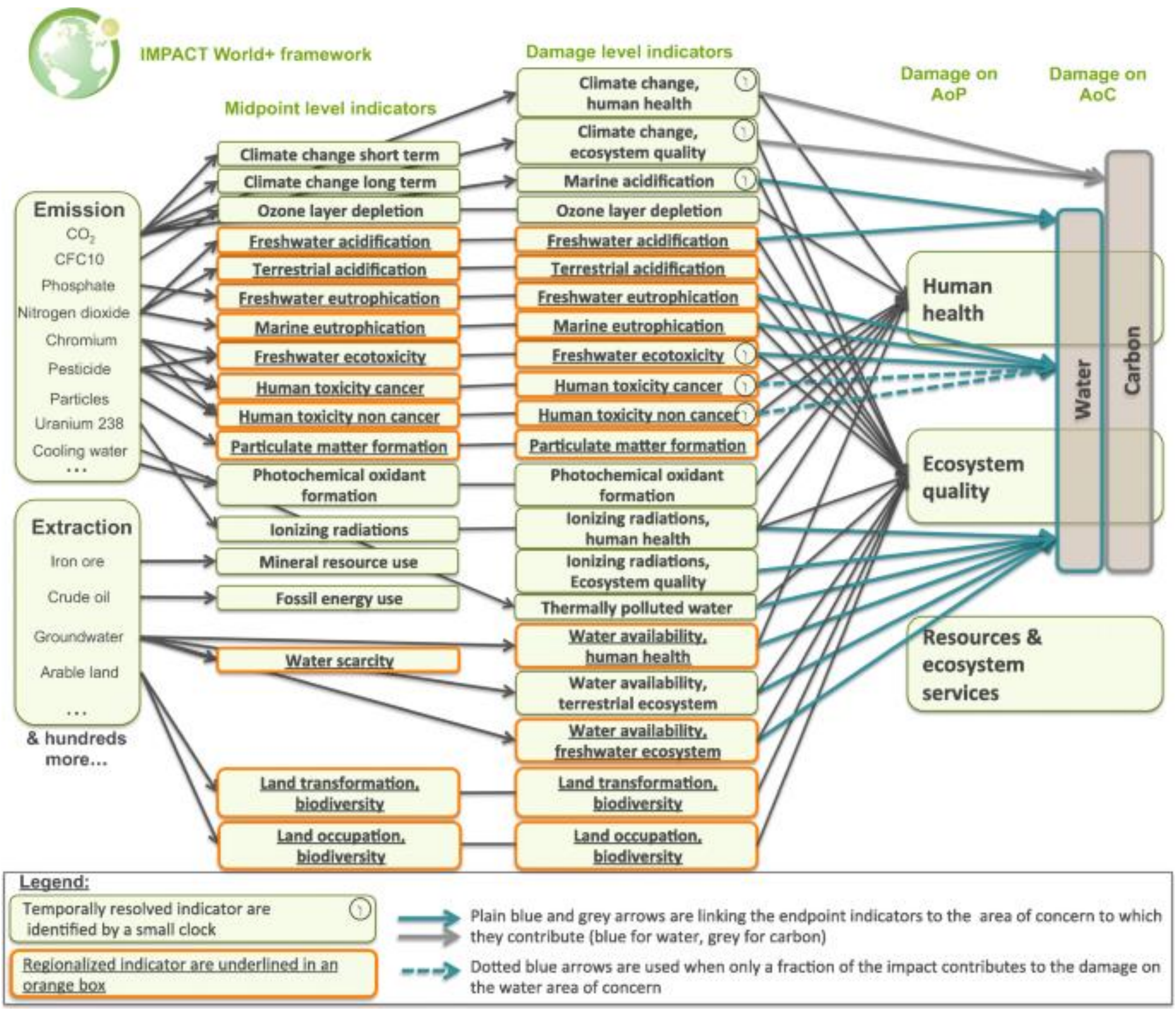

Figure 4.15 *IMPACT World+ method impact categories at midpoint and at damage level IMPACT World+ (2023)*

The IMPACT World+ method integrates developments in the following categories, all structured according to fate (or competition/scarcity), exposure, exposure response, and severity:

- Complementary to the global warming potential (GWP_{100}), the IPCC Global Temperature Potentials (GTP_{100}) are used as a proxy for climate change long-term impacts at midpoint. At damage level, shorter-term damages (over the first 100 years after emission) are differentiated from long-term damages.
- Marine acidification impact is based on the same fate model as climate change, combined with the H^+ concentration affecting 50% of the exposed species.
- For mineral resources depletion impact, the material competition scarcity index is applied as a midpoint indicator.
- Terrestrial and freshwater acidification impact assessment combines, at a resolution of 2° x 2.5° (latitude x longitude), global atmospheric source-deposition relationships with soil and water ecosystems sensitivity.
- Freshwater eutrophication impact is spatially assessed at a resolution grid of 0.5° x 0.5°, based on a global hydrological dataset.
- Ecotoxicity and human toxicity impact is based on the parameterised version of USEtox for continents. Indoor emissions are considered, and the impacts of metals and persistent organic pollutants are differentiated for the first 100 years from longer-term impacts.
- Impacts on human health related to particulate matter formation are modelled using the USEtox regional archetypes to calculate intake fractions and epidemiologically derived exposure response factors.
- Water consumption impacts are modelled using the consensus-based scarcity indicator AWARE as a proxy midpoint, whereas damages account for competition and adaptation capacity.

- Impacts on ecosystem quality from land transformation and occupation are empirically characterised at the biome level.
- SpatialAWARE as a proxy midpoint, whereas damages account for competition and adaptation capacity.
- Impacts on ecosystem quality from land transformation and occupation are empirically characterised at the biome level.

4.5.3.2 Spatial Differentiation

For regionalised impact categories affecting ecosystem quality (acidification, eutrophication, and land use), the characterisation factors of half of the regions (25^{th} to 75^{th} percentiles) are within one to two orders of magnitude and the 95^{th} percentile within three to four orders of magnitude, which is higher than the variability between substances, highlighting the relevance of regionalising (Bulle et al. 2019).

4.5.3.3 Reflections

23 impact indicators were considered still immature and not yet integrated into the LCIA method. This requires further research. The method is not implemented into LCA software, but the documents are readily available online to import IMPACT World+ manually into LCA software.

While ReCiPe 2016 is a representative of Western European conditions, IMPACT World+ considers spatial variability at a global level. Unlike ReCiPe 2016, IMPACT World+ is not sufficiently developed for direct application.

4.6 Cumulative Impact Assessment (CIA) method for the marine environment

4.6.1 Introduction

Marine analysis of the relationship between socioeconomic activities and biodiversity has been developed more or less independently from those for terrestrial biodiversity. Therefore, this section an provides an insight into the marine approach and evaluates the extent to which it differs from the terrestrial approach. The main approach for the marine sector is called cumulative impact assessment (CIA). The aim of a CIA is to reveal the vulnerability of the different ecosystem components in terms of their potential impact risk from the cumulative pressures across all human activities. The relationship between expected (cumulative) impacts and biodiversity targets can be investigated via establishing links between the ecosystem components and pressures included in a CIA, and the EU policy objectives, specifically those of Natura 2000 and the MSFD, which are the Good Environmental Status (GES) descriptors.

4.6.2 Method

The CIA method is in development since 2013 and it is based on peer-reviewed studies conducted in international collaborations. It started with the EU-funded project ODEMM (Options for Delivering Ecosystem-Based Marine Management),[1] e.g. Knights et al. (2015); this was followed by the EU-funded project AQUACROSS (Knowledge, Assessment, and Management for AQUAtic Biodiversity and Ecosystem Services aCROSS EU policies).[2] A major product of these projects is a comprehensive database containing 7771 causal impact chains for the North Sea, which were all semi-quantitatively assessed using (scientific) knowledge from literature supplemented by expert judgement of a large team of international experts (Borgwardt et al. 2019). This assessment was based on five AQUACROSS criteria: (i) extent, (ii) dispersal, (iii) frequency, (iv) persistence, and (v) severity (Borgwardt et al. 2019) and two additional criteria: (vi) resilience (Knights et al. 2015) and (vii) pressure load (Pload) developed as part of the ICES WGCEAM (Working Group on the use of Cumulative Impact Assessments for Management) (Piet et al. 2023). The next subsections briefly describe the linkage framework, pressures caused by human activities, impact criteria, impact risk calculation with type of results and confidence.

1 https://odemm.com
2 http://aquacross.eu

4.6.3 Linkage framework

CIAs require a mental model, sometimes referred to as the linkage framework, which connect the different categories of human/economic activities-pressures and ecosystem components through impact chains. Mental models represent the way in which people understand the world around them. A mental model can always be applied, and the comprehensiveness depends on its complexity in terms of the level of detail of the sectors and their activities, or of the ecosystem, or of the extent to which ecosystem services or the full social-ecological system are considered. An example of a relatively concise linkage framework comes from the CIA for the ecological and economic costs and benefits of spatial plans for the North Sea, carried out by Jongbloed et al. (2021a) and Roebeling et al. (2021), see Table 4.5 and Figure 4.16 Another example of a relatively comprehensive linkage framework can be found in Tamis et al. (2016). The number of activities, pressures and ecological components can be larger than chosen in Table 4.5, see Borgwardt et al. (2019).

Table 4.5 *Activities, pressures and ecological components in the CIA linkage framework for the North Sea*

Activities	**Pressures**
Aquaculture	Abrasion/Damage
Fishing: Benthic trawling	Artificialisation of habitat
Fishing: Nets	Barrier to species movement
Fishing: Pelagic trawls	Change of habitat structure/morphology
Oil and Gas	Changes in input of organic matter
Sand extraction	Changes in Siltation
Shipping	Death or Injury by Collision
Telecoms and Electricity	Disturbance (visual) of species
Wind farms/parks	Electromagnetic changes
	Extraction of flora and/or fauna
Ecological components	Input of light
Birds	Introduction of genetically modified species
Fish	Introduction of Microbial pathogens
Mammals	Introduction of non-indigenous species
Habitat Pelagic water column	Introduction of Non-synthetic compounds
Habitat Sublittoral sediment	Introduction of Radionuclides
Habitat Littoral sediment	Introduction of Synthetic compounds
Habitat Circalittoral rock and other hard substrata	Litter
	N&P Enrichment
	Noise (Underwater and Other)
	pH changes
	Selective Extraction of non-living resources: substrate
	Smothering
	Total Habitat Loss
	Translocations of species (native or non-native)
	Water abstraction
	Water flow rate changes

Source: Jongbloed et al. (2021); Roebeling et al. (2021).

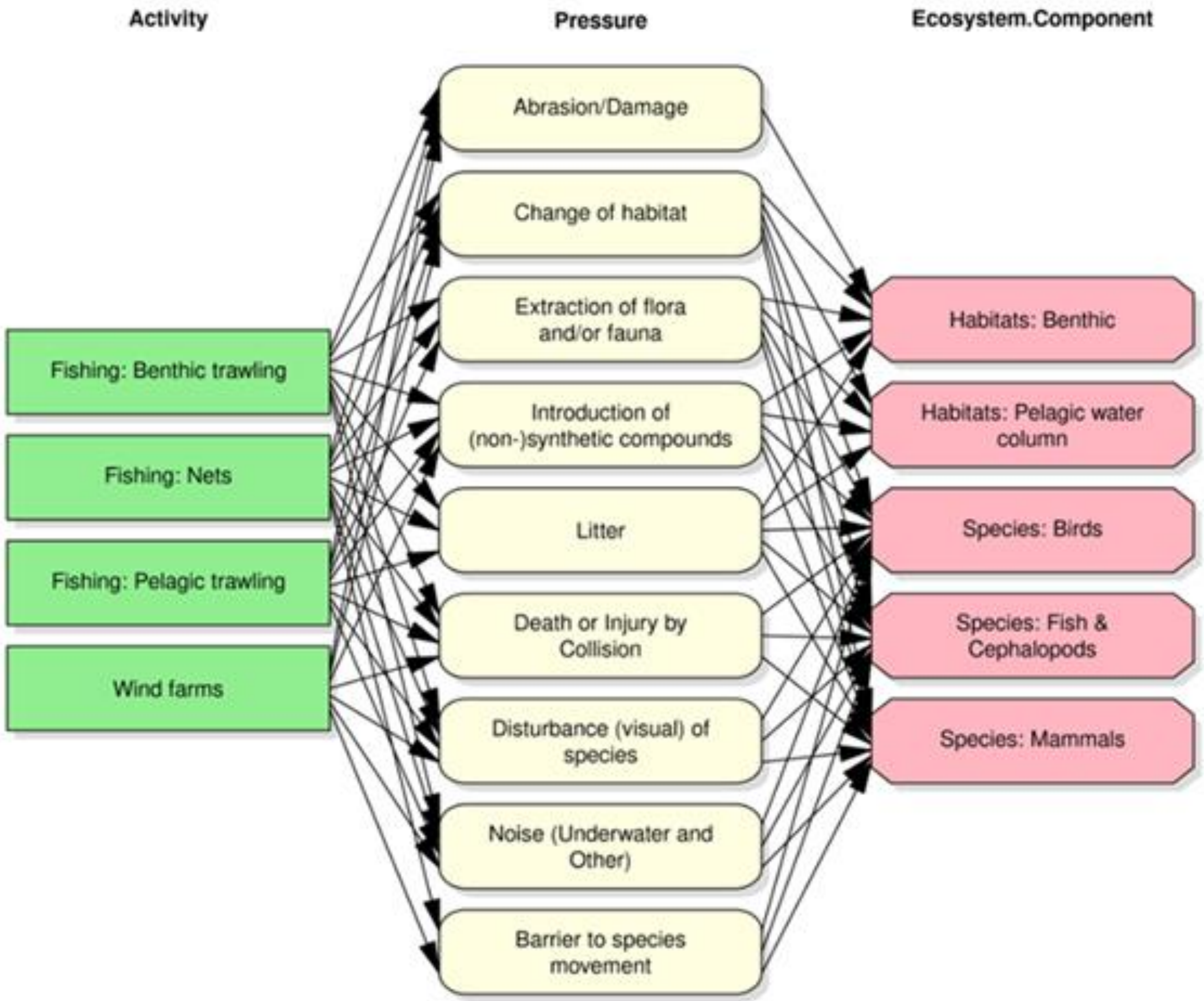

Figure 4.16 *Illustration of the linkage framework (mental model) for a CIA confined to four of the nine selected sectoral activities from Table 4.5 and their main pressures and their potential impact on the ecosystem components*
Source: Jongbloed et al. (2021).

When Figure 4.16 is compared with the LCA method, then the green part is the list of activities, the yellow part is the inventory, and the pink part are the endpoints.

4.6.4 Pressures caused by human activities on the North Sea ecosystem

Pressures are considered as 'the mechanism through which an activity has an effect on any ecosystem component' (Knights et al. 2015). A human activity may be the source of multiple pressures and any single pressure may be caused by more than one activity (Knights et al. 2015). Borgwardt et al. (2019) established a typology of human activities, a typology of pressures those activities introduce to aquatic ecosystems and a typology of aquatic ecosystem components impacted by those pressures. A total of 45 human activities were linked through 31 pressures to 82 ecosystem components, resulting in a linkage framework of over 22,000 activity-pressure-ecosystem component interactions across seven European case studies, of which one involves the North Sea (Borgwardt et al. 2019). The linear interaction between a sector, pressure, and ecological component is referred to as an 'impact chain' (Knights et al. 2015), consistent with LCA. Although it is recognised that indirect effects can play a role in the functioning of an ecosystem, only direct effects of sector–pressures on ecological components are considered (Knights et al. 2015). Knowledge on the interaction mechanisms of multiple stressors (additivity, synergy, antagonism) is lacking and therefore it is assumed that they will act in an additive fashion (following e.g. Judd et al. 2015)). This is consistent with the approach in LCA, which is an acceptable approximation for small changes.

Over 30 pressures can be identified and categorised in broad pressure types, see Borgwardt et al. (2019):

- biological (e.g., Introduction of Microbial Pathogens)
- chemical (e.g., Introduction of Synthetic Compounds)
- physical (e.g., Abrasion) and
- energy (e.g., Thermal Changes).

This approach is consistent with the LCA approach, although in an LCA the attempt is to limit the number of pressures as much as possible.

4.6.5 Impact criteria

This section describes the CIA criteria. Table 4.6 shows how for each of the criteria expert evaluations are being used, and which values are given to these different evaluations.

The *Spatial extent*, or overlap of each activity with each ecosystem component, was evaluated by considering the spatial distribution of human activities and ecosystem component in the study area, and how much spatial overlap there is between these (Borgwardt et al. 2019). The area of overlap is relative to the area occupied by the ecosystem component in question within the study area. The actual location of pressures and their impact pathways was considered when assigning spatial extent (e.g., accounting for the fact that not all pressures are introduced across the whole operating area of an activity; for example, abrasion is only introduced where fishing vessels are trawling or anchoring, while noise is introduced while steaming). The extent scores as applied by Borgwardt et al. (2019) were adjusted according to the spatial data gathered, if relevant (i.e., only for the selected activities and ecosystem components). The effect may be broader than the location of execution. *Dispersal* evaluates the potential of an activity-pressure impact to spread and increase its spatial overlap with an ecosystem component beyond that of the area of extent where the pressure and ecosystem component overlap initially (Borgwardt et al. 2019).

Frequency of interactions described the most likely number of times the activity interacts with an average square kilometre of an ecosystem component in an average year, where they overlap in space (Borgwardt et al. 2019). Moreover, would be relevant to consider the period of time necessary for the pressure associated with a particular activity to disappear after cessation of any further activities causing the particular pressure. This temporal component was described by *persistence*. For example, while habitat loss is persistent, organic enrichment is temporary.

Severity described the generic severity of an interaction in terms of its effects on the ecosystem component (Borgwardt et al. 2019). The type of response of the ecosystem component to the pressure was categorised as either ‘Acute’, ‘Chronic’ or ‘Low’. More details on the five Aquacross criteria and the classifications are presented in Table 4.6 The weighting of each impact chain was carried out by regional experts and co-ordinated by a core expert team that ensured consistency in the approach. Categorical weights were converted to numerical scores based on the justifications in Table 4.5 (Borgwardt et al. 2019).

The criterion *Resilience* (Knights et al. 2015) was reintroduced, which represents the recovery time of the ecological characteristic to return to pre-impact conditions. Recovery times for species assessments were based on turnover times (e.g., generation times). For predominant habitat assessments, recovery time was the time taken for a habitat to recover its characteristic species of features given prevailing conditions.

Furthermore, an criterion was added, *Pressure Load*, which has been developed within the ICES WGCEAM (Working Group on the use of Cumulative Impact Assessments for Management). Pressure load was introduced because the pressure intensity was not explicitly considered in the Aquacross North Sea ‘Impact Risk’ database. Therefore, an estimated (expert judgement) activity-specific contribution to the pressure load was introduced. Together all activities add up to 1 (or 100%) for the pressure.

Table 4.6 *Impact risk criteria with their categories and assigned numerical scores used to weight each impact chain*

Description		Standardised score
Spatial extent a): spatial overlap of each activity-pressure combination with an ecosystem component		
Exogenous	The activity occurs outside of the area occupied by the ecosystem component, but one or more of its pressures would reach the ecosystem component through dispersal	0.01
Site	The activity overlaps with the ecosystem component by up to 5% of the area occupied by the EC in the case study area	0.03
Local	The activity overlaps with the ecosystem component by between 5 and 50% of the area occupied by the EC in the case study area	0.37
Widespread patchy	The activity overlaps with the ecosystem component by between 50 and 100% of the area occupied by the EC in the case study area, but the distribution within that area is patchy	0.67
Widespread even	The activity overlaps with the ecosystem component by between 50 and 100% of the area occupied by the EC in the case study area, and is evenly distributed across that area	1
Dispersal a): effect of the dispersal of the pressure on realised area of spatial overlap		
None	The pressure does not disperse in the environment	0.01
Moderate	The pressure disperses, but stays within the local environment	0.1
High	The pressure disperses widely and can disperse beyond the local environment	1
Frequency a): temporal overlap of each activity-pressure combination with an ecosystem component		
Rare	Occurs approximately 1–2 times in a 5-year period but may (or may not) last for several months when it occurs	0.01
Occasional	Can occur in most years over a 5-year period, but not more that several times a year	0.11
Frequent	(1) occurs in most years over a 5-year period, and more than several times in each year, or (2) can occur in 1–2 years in a 5-year period but also in most months of those years	0.33
Very frequent	Occurs in most months of every year, but is not constant where it occurs	0.72
Continuous	Constant in most or all months of a 5-year period	1
Persistence a): length of time that is needed that a pressure disappears after activity stops		
Low	0 to <2 yr	0.01
Moderate	2 to <10 yr	0.06
High	10 to <100 yr	0.55
Persistent	The pressure never leaves the system or > 100 yr	1
Severity a): likely sensitivity of an ecosystem component to a pressure where there is an interaction		
Low	An interaction that, irrespective of the frequency and magnitude of the event(s), never causes a noticeable effect for the ecosystem component of interest in the area of interaction	0.01
Chronic	An impact that will eventually have severe consequences at the spatial scale of the interaction, if it occurs often enough and/or at high enough levels	0.1
Acute	A severe impact over a short duration	1
Resilience b): the recovery time of the ecological characteristic to return to pre-impact conditions		
None	The population/stock has no ability to recover and is expected to go ‘locally’ extinct. The recovery in years is predicted to take 100+ years	1.00
Low	The population will take between 10 and 100 years to recover. A raw value taken as the midpoint between the range boundaries	0.55
Moderate	The population will take between 2 and 10 years to recover. A raw value taken as the midpoint between the range boundaries	0.06
High	The population will take between 0 and 2 years to recover. A raw value taken as the midpoint between the range boundaries	0.01
Pressure load c): contribution (%) of activities to the total load of pressure in the studied area		
Extremely minor	Activities contributing 0.1% of the total pressure load	0.1
Very minor	Activities contributing 1% of the total pressure load	1
Minor	Activities contributing 5% of the total pressure load	5
Main	Balanced distribution of the remaining load after all minor activities have been scored. This should add up to a total of 100% for the pressure	>5% (adding up to 100%)

Sources: a) Borgwardt et al. (2019); b) Knights et al. (2015); c) Piet et al. (2023).

4.6.6 Impact risk calculation

Impact Risk is calculated through a risk assessment of the potential impact on nature as the combination of two aspects of risk, i.e., exposure and potential effect, where:
- The exposure is based on:
  - The area of the activity (Extent)
  - Potential spreading of the related pressures (Dispersion)
  - Relative contribution of the activity to the related pressures (Pressure load).
- The potential effect is based on:
  - The sensitivity of the ecosystem component to the pressures (Severity)
  - The number of times the activity interacts with an ecosystem component (Frequency)
  - The recovery time of the ecosystem component (Resilience)
  - The length of time it would take for the pressure to disappear after cessation of the activity(s) causing the particular pressure (Persistence).

Exposure reflects the proportion (%) of the ecosystem component that is potentially perturbed by the pressure. In case of quantitative information on a spatial grid, this can be estimated in terms of the Overlap, Likelihood of encounter, the Magnitude or the Severity of Exposure (Piet et al. 2021a), depending on the information available.

The *potential effect* represents the proportion (%) of the ecosystem component that is actually perturbed to a level where its contribution to ecosystem integrity and functioning is compromised. For each grid cell where both the ecosystem component and pressure occur, this is the equilibrium % abundance (numbers, biomass) of that ecosystem component relative to undisturbed. The potential effect is completely based on categoric scores as already included in the Aquacross database, Table 4.5.

Impact Risk is calculated for each impact chain as the product of exposure and the potential effect and is expressed as the proportional (%) change in the abundance of a particular ecosystem component due to a particular activity-pressure combination. The CIA then aggregates across impact chains to show for example the (1) vulnerability of each ecosystem (component) to the aggregated activity/activities or (2) the threat caused by each activity or activity-pressure combination to overall biodiversity, i.e., aggregated across ecosystem components. Vulnerability represents the likelihood that the state of the overall ecosystem and/or any of its components is reduced by a certain proportion (%) compared to a specific undisturbed or reference period and hence the likelihood that environmental policy objectives are not achieved. An impact risk equal to 0 implies the ecosystem component is undisturbed, a value of 100 or more implies (local) extinction. The ecosystem components cover the main biological features, which can be easily linked to the MSFD descriptors (see Table 2.6).

The CIA method can be applied to provide insight into the following type of research topics concerning scenarios and management measures on the impacts of human activities on the North Sea ecosystem:
- Relative contribution of human activities
- Relative vulnerability of ecosystem components
- Relative impact of Marine Protected Area (MPA) measures.

It should be noted that the output of the CIA:
- Only includes direct effects, i.e., effects via food web relations and other cascading effects are not included.
- Is especially informative on relative values, i.e., the differences between the reference situation and the alternative/future/potential situation.
- Provides ranking orders of contributors to the overall threat caused by human activities.
- Can be used to provide an integrated perspective on the (change in) vulnerability of the ecosystem as a whole (in a specific study area such as the Dutch part of the North Sea) as well as each of the different ecosystem components.
- Cannot be used to predict the actual values for specific indicators (e.g., MSFD).
- Has currently only limited value in providing spatially-explicit advice, but the development of a spatially-explicit and quantitative CIA method has started by Piet et al. (2021a).

4.6.7 Confidence

The CIA is based on categorised risk criteria. Scores are assigned to the qualitative categories to reach semi-quantitative estimates of impact risk (see Table 4.6). These categories and scores are rather course, and the scores may be arbitrary. In general, the criteria, their scores and estimated impact risks come with considerable uncertainty. This uncertainty can be assessed, e.g., in Piet et al. (2021a). There are possibilities to reduce some types of uncertainties. An example of this is elaborated in a fully quantitative CIA such as applied by Piet et al. (2021a), but this requires substantial quantitative information, and in the end the scores are mainly based on expert insights.

4.6.8 Case studies

The CIA method was applied in a number of case studies for the North Sea (e.g., Piet et al. 2017a; Jongbloed et al. 2019; Jongbloed et al. 2021; Roebeling et al. 2021; Piet et al. 2021a; Piet et al. 2021b).

Jongbloed et al. (2019) used the framework and data from Borgwardt et al. (2019) to analyse consequences of different scenarios for impacts on ecosystem components in the Dutch part of the North Sea, and this framework is used in a cost-benefit analysis of the North Sea (Roebeling et al. 2021). The impact risk is calculated for different scenarios (based on policy goals) and compared with a reference scenario (a recent year) and expressed in terms of increase/decrease of disturbance of the ecological components (e.g., fish, birds), which means that the state will deteriorate or improve. In case of an unfavourable environmental state/status (conservation state), only a decrease in disturbance may lead to a favourable environmental state over a certain period of time. The other way around, an increase in disturbance may change a favourable environmental state to an unfavourable environmental state. The consequences of the policy goals (for the state of the North Sea) are produced for each of the 11 MSFD Descriptors.

To predict the ecological and economic costs and benefits of spatial plans for the Dutch part of the North Sea, a CIA was carried by Roebeling et al. (2021). A key issue for the Dutch marine spatial planning (MSP) process is the siting of offshore wind farms and their impacts. The CIA was part of a case study, which described the process to guide the planning of offshore windfarms while balancing the demands for space from renewable energy with those for sustainable food (fisheries and aquaculture) and nature conservation (Jongbloed et al. 2021).

The cumulative impacts of off-shore wind farms and other main human activities on the North Sea ecosystem are assessed by Piet et al. (2021b). The objectives of the study by Piet et al. (2021b) are to evaluate for the North Sea marine ecosystem the knowledge base to assess the cumulative impacts of all the main human activities under various planning scenarios (until 2030 and 2050). To that end, a CIA is applied, and the outcomes are interpreted in relation to the achievement of (EU) biodiversity targets, and the concept of carrying capacity for the North Sea ecosystem. It contained a discussion on the possibilities to apply CIA in the context of an ecosystem-based approach to marine spatial planning aimed at achieving existing environmental marine policy goals. Most of the knowledge is still expert judgement-based, but now with the advantage of having a formalised methodology that can guide further scientific research to, in time, provide the information required and (further) improve the quality of such Cumulative Impact Assessments (CIAs).

The CIA case studies described above apply actual data on spatial distributions but rely on categorical scores based on expert judgement. Therefore, the impact is expressed as a relative measure that is difficult to interpret in absolute terms. To overcome this issue, Piet et al. (2021b) developed a first stepwise approach to conduct a fully quantitative CIA based on the selection and subsequent application of the best information available. This approach systematically disentangles risk into its exposure and effect components that can be quantified using known ecological information, e.g., spatial distribution of pressures or species, pressure-state relationships and population dynamics models with appropriate parametrisation, resulting in well-defined assessment endpoints that are meaningful and can be easily communicated to the recipients of advice. They applied the approach in a North Sea CIA focussing on two sectors: fisheries and offshore windfarms, and how they impact the ecosystem and its components, i.e., seabirds, seabed habitats and marine mammals through various pressures.

4.7 Conclusions

Even though impact assessment approaches and models integrating biodiversity into LCA are available and are sometimes integrated in LCA software tools (e.g. SimaPro), currently they are rarely used in common practice (Winter et al. 2017). Many stakeholders do not consider the credibility of the results strong enough (Milà i Canals et al. 2007). Moreover, there are currently hardly any incentives and pressure factors that would urge companies to assess and document the biodiversity performance of products. This calls for action at the country and/or international level. Current LCA approaches do not differentiate sufficiently between different production practices and do not have enough spatial detail to differentiate between different locations.

The ReCiPe methodology provides a systematic way to relate different midpoint pressure factors to biodiversity. The method uses only PDF.m^2 as an indicator which represents the fraction of species that is still available compared with the natural state and has limited spatial detail except for some pressure factors. The effects of all pressure factors are additive. GLOBIO uses an indicator that is related with PDF.m^2, MSA.m^2, but aggregate pressure factors multiplicatively. However, for relatively small changes, this will not give a very different result in the outcome. The pressure factors that are included differ, which basically implies that both approaches can learn from each other.

The two discussed new LCIA methods are not sufficiently developed yet. One of them weights the PDFs with the importance of ecosystems for global species survival, the global PDF, and both have more spatial detail than ReCiPe.

The marine approach is conceptually not fundamentally different from the terrestrial approach in being focused on impact pathways. However, while the terrestrial approaches of GLOBIO and LCA tend to aggregate biodiversity towards one general indicator, and is looking around for alternatives now, the marine approach starts with describing all types of indicators (they distinguish 82 ecosystem components) and is still in search to bring them into one framework. Furthermore, while the terrestrial approaches of GLOBIO and LCA tend to focus on a limited number of empirical relationships that are quantified, the marine approach tries to catalogue all relationships, uses mainly semi-quantitative indicators using expert-judgement based scoring of qualitative categories, and is only starting to search for opportunities for quantification. However, this shows that the terrestrial and marine approach are gradually converging to the same target: a further focus on quantification, with sufficient relationships and with more than one biodiversity indicator included in the evaluation.

5 National biodiversity accounting

5.1 Introduction

There has been an emerging interest to incorporate environmental emissions, ecosystem services and biodiversity in measurable indicators to inform policy decision-making. At the national level, there are all kinds of initiatives to inform governments on the status of biodiversity (e.g. IPBES reporting) and how pressures and impacts of biodiversity are developing (e.g. GBOs of the CBD). In the last three decades, there have been several attempts to include environmental issues in national accounts inspired by the report of the World Commission on the Environment and Development (Brundtland 1987), but System of Environmental and Economic Accounts (SEEA) is the first generally accepted environmental accounting system.

The general advantage of a consistent ecosystem and biodiversity accounting framework is that it includes all ecosystem flows and stocks, prevents double counting, organises information in a broadly agreed manner through consistent classifications and that it explicitly relate to the economic activities that generate the changes in ecosystems and biodiversity. Moreover, the SEEA Applications and Extensions (UN et al. 2017) illustrate how information from SEEA can be used in decision-making, analysis and research. Subsystems of the SEEA are developed to give more detail on specific resources or sectors and to integrate the system with insights from experts on specific topics. For example, there is an air emissions account, a land account and a water account (UN 2021). In the end, the environmental accounting system is meant to be relevant for sustainability analysis and in this context it would be beneficial to couple the system with social aspects of development and progress (UN 2021).

The UN has developed several guidelines to connect environmental pressures, ecosystem services and biodiversity to the System of National Accounts (SNA). Firstly, environmental flows and pressures were linked with the SEEA. Secondly, ecosystem services were incorporated with the SEEA Ecosystem Accounting (SEEA EA). Finally, there are recent attempts to associate biodiversity to the national accounts with the SEEA Experimental Biodiversity Accounting (SEEA EBA). In a study on the role of biodiversity and national accounting, Hamilton (2013) argued that the wish to incorporate values of biodiversity in national accounts might be elusive, but an aggregate value of nature might be feasible. All approaches will be presented in this chapter.

As all accounting systems are linked to the SNA, Section 5.2 starts with an introduction of the SNA. The SEEA consists of two parts. Firstly, the SEEA Central Framework (United Nations et al. 2014b), that was accepted by the UN Statistical Commission and has a focus on the use of natural resources in the economy and the consequences of economic activities for the environment, see Section 5.3. The Central Framework is focused on individual environmental assets, such as water and energy. Secondly, the SEEA EA focuses on ecosystems as systems where different individual assets interact and which deliver ecosystem services not only for production, but directly to the consumers (UN 2021) in Section 5.4. Section 5.5 presents the SEEA BA, which is a relatively less developed accounting system, although it is essential for understanding ecosystems, their conditions and the ecosystem services they provide (UNEP-WCMC 2015). Because marine ecosystem and biodiversity accounting has its own challenges, a separate section is focused on the example of ecosystem and biodiversity accounting for the marine sector in the EU in Section 5.6. Section 5.7 concludes.

5.2 The System of National Accounts (SNA)

The SNA is a consistent accounting system for economic transactions and it serves as a base for other accounting system including environmental, ecosystem and biodiversity accounting. The system is described in this report because it shows the difficulty of the decisions that have to be made to make an accounting system consistent, and especially the difficulties to make wealth and capital accounting consistent with accounting of flows. The same problems emerge in natural capital and ecosystem accounting.

The SNA registers the transactions in the economy in a consistent manner UN et al. (2014b). The foundation is the supply and use table, where an accounting identity forces that supply equals the sum of all uses. The supply consists of the production and import of goods and services and is equal to the total use of the products, being the sum of consumption by households and government, intermediate consumption of products by producers, exports and gross capital formation in both fixed assets (machines, buildings) and inventories (UN et al. 2014b). Goods and services are classified according to the Central Product Classification (CPC). For sectors, the International Standard Industrial Classification of All Economic Activities (ISIC) is used. One sector may produce multiple products and services, while one product or service is produced by one single industry by definition.

Although the SNA is usually applied at the national level, there are examples of systems of regional accounts.

In the accounting system, the distinction between stocks and flows is essential. Flows that result in storage and gross capital formation (investment) increase the stock of assets, while the extent to which the assets are used up (depreciation) reduces the stock of assets. Although depreciation is essential to account for stocks, depreciation is not a transaction and therefore is be measured directly. In national legislation, there are rules to calculate depreciation. As these rules are based on political decisions, these rules are not very reliable. In accounting, there is a difference gross accounting results, such as Gross Domestic Product (GDP), and net accounting results, such as Net Domestic Product (NDP). The discrepancy between GDP and NDP is depreciation.

Accounting identities play a crucial role in the measurement of some variables. For example, the difference between the value of the production and the value of the intermediate consumption is called value added. Thus, for each sector, the sum of the input values and value added equals the output value. This is the input-output identity. The sum of sectoral value added equals GDP. All these calculations are made in what is called the Production Account.

Based on the Production Account, a sequence of other accounts can be derived. The value added calculated in the Production Account is distributed over wages, taxes, subsidies, capital income and rent. These payments from value added may partly go to foreigners, while Dutch citizens may receive income from value added in foreign countries. In the distribution and use of income accounts the income calculated is either spent for final consumption or saved, where the balancing item is called net savings. By definition, savings are the difference between income and consumption.

Savings are used to acquire assets in the capital account. These assets can be both produced assets and environmental assets, such as land, livestock and plantations. Net lending is acquisition of assets minus savings, which is equal to exports minus imports in the Production Account. The last account, the financial account, records all transactions involved in lending and borrowing, i.e., deposits, loans, shares and equities. The financial account is balanced by net lending.

The stock side of the national accounts is registered in the balance sheets. The balance sheets register the values of all assets and liabilities at the beginning and the end of the accounting period, and the difference between them is net worth. Net worth is the value of all assets minus the value of all liabilities. This balancing item equals the total value of all capital goods and inventories in the economy.

Changes in wealth are determined by net savings as well as exogenous changes in the value of the assets and liabilities, which may be physical or monetary in character. For example, a price change of land increases wealth without a change in the physical amount of capital, while a storm may cause physical destruction of wealth that is not caused by production activities.

It is common for comparisons over time to have accounts in real terms, where all monetary values are adjusted for changes in prices over time. For example, the growth of GDP is normally formulated in real terms, i.e., corrected for changes in prices. Advanced methodologies have been developed to distribute changes in value between changes in prices, changes in quality and changes in quantity (see UN (2021)).

For prices, one must be aware that there is a difference between basic prices, i.e. the price that the producer receives, and, market prices, i.e. the price the producer has to ask after paying indirect taxes and receiving subsidies.

The valuation in the SNA is based on market prices, but not for all relevant transactions' prices are available, such as for labour of self-employed farmers, the consumption of own-produced food products. If there is no market price, one has to estimate what price willing buyers and sellers would accept in equilibrium. For consumers who produce their own consumption goods, the production is registered as production using prices derived from other prices of the same commodities, and then delivered to the consumer himself. When compiling national accounts of exchange values are not directly available (Obst et al. 2016), it is advised to first look for values of similar trade items (for example value in housing stocks based on traded houses, i.e. imputed rent, or valuation of subsistence agriculture). A second option is to look at production cost, such as for health, education and national defence (UN 2021). When changes in capital have to be accounted for, one may use depreciation schemes of investment goods (UN 2021), or one may estimate changes in the discounted value of future returns, i.e. the net present value of the goods.

Wealth accounting uses other concepts than national accounts based on estimated shadow prices, which are the marginal contributions of assets to well-being. The reason to use exchange values in national accounts is that they are relatively easy to observe and that they guarantee consistency in the accounts, in the sense that the price that the buyer pays is the same as what the buyer gets (Obst et al. 2016). In economic theory, the market price is set at the point of production/consumption where marginal benefits are equal to marginal cost of a commodity. For small changes, the prices used in market transactions provide information on equilibrium prices, i.e., prices where marginal costs equal marginal benefits.

One must be aware that the main objective of the SNA is to compile metrics of economic activity that are based as much as possible on exchange values, not to measure welfare. For example, consumer surplus, i.e. when the price that consumers pay for a commodity is less than the price they are willing to pay, is not included in the national accounts (Obst et al. 2016). Externalities are not included in the national accounts, because externalities occur without a market transaction.

In conclusion, the SNA consistently measures the transactions in an economy. However, the accounting system requires a number of decisions that are not always logic from an outsider point of view. Nevertheless, the logic of national accounts can be an inspiration for consistent manners of account of ecosystems and biodiversity.

5.3 The Central Framework of the System of Environmental-Economic Accounting (SEEA)

5.3.1 Introduction

Because the market system generates negative effects for the environment not included in the system of national accounts, an accounting system has been developed that relates the negative effects on the environment with economic activities in the national accounts. This is the System of Environmental-Economic Accounting (SEEA), where the Central Framework of these accounts focus on the direct environmental effects (UN 2017). The goal of the Central Framework is to compile and contrast monetary and physical data in a meaningful way and to analyse different economic and environmental strategies with respect to their trade-offs. A key motivation is to understand the potential of ecosystem and other environmental assets to provide services now and in the future that contribute to individual and societal well-being (UN 2021).

The Central Framework of the SEEA is a framework that integrates environmental flows into the SNA. Pollution, waste and the use of natural capital are allocated to processes in the SNA. For implementation, one may for example allocate greenhouse gas emissions to consumption of specific commodities, production of sectors, or the use of inputs to produce the commodities. Classifications of commodities and sectors are the same as those used in the national accounts, and regions are defined at the same spatial level as in the national accounts, normally at the country level.

The data coupled to the items of the national accounts can be expressed both in physical and monetary units. To analyse the consequences of economic policies or developments on the environment, it is sufficient to have the data in physical units. In economic models, one may put restrictions on these physical units, and this will generate prices if restrictions generate scarcities. As a consequence, a market for the externalities may emerge. The restrictions put into the models can be seen as definitions of property rights for the unpriced externalities, although one should be aware that the definition of good property rights might be debatable because of transaction cost (Coase 1960).

As far as the use of natural resources generates rents, this can be allocated to the environment and this rent is then included in the national accounts. However, if the use of environmental services is not paid for, it is not included in the monetary system of accounts, although it is included as a physical flow. Thus, activities for protection or restoration of the natural environment are implicitly incorporated in the SNA but are considered explicitly in the SEEA. Furthermore, activities within firms that are normally not registered in the national accounts, such as emissions and resource use, are accounted for explicitly in SEEA.

The remainder of this section focuses on specific aspects of the SEEA. First, the environmental flow accounting into the supply and use tables of the SNA will be discussed. Then the extension with environmental asset accounting as parallel with produced asset accounting in the SNA will be discussed. This includes a summary of difficulties to delineate natural resource inputs and residuals (such as pollution and waste) from economic inputs and to have a good classification system for them. Finally, some attention is given to opportunities to separate economic activities that improve the environment from other activities within the SNA.

5.3.2 Environmental supply and use tables

The most obvious way to relate to the environment is through the supply and use table in the SNA. However, most flows of goods and services for which there is no market and thus no market price exists, are excluded from the supply and us tables in the SNA. The SEEA makes these flows explicit in the 'Environment' sector in the supply and use tables, see Table 5.1. Flows of materials and energy are included in the supply and use tables of the SNA. The SEEA supply and use table has the Environment as an extra sector. Supply by the environmental sector is defined as the sum of all flows (of natural inputs from the environment), see top of Table 5.1. Therefore, supply of natural inputs is by definition equal to flows from the natural environment. Furthermore, in the use table, the 'Environment' sector includes all residuals that flow into the natural environment such as water, soil and air pollution.

Table 5.1 *Basic physical supply and use table from SEEA a)*

	Industries	Households	Accumulation	Rest of the World	Environment	Total
Supply table						
Natural inputs					Flows from the environment	Total supply of natural inputs
Products	Outputs			Imports		Total supply of products
Residuals	Residuals generated by industry	Residuals generated by final household consumption	Residuals from scraping and demolition of produced			Total supply of residuals
Use table						
Natural inputs	Extraction of natural inputs					Total use of natural inputs
Products	Intermediate consumption	Household final consumption	Gross capital formation	Exports		Total use of products
Residuals	Collection and treatment of waste and other residuals		Accumulation of waste in controlled landfill sites		Residual flows direct to environment	Total use of residuals

a) Empty cells are null by definition.

Source: UN et al. (2014b, p. 17).

Accounting identities are essential for consistent registration. As in the SNA, two identities hold. First, everything that is supplied is used in some way; it is either consumed, exported and what is not used is accumulated somewhere. Second, the sum of all inputs equals output, the input-output identity. This represents the laws of conservation of mass and energy in the same manner as this holds for the monetary flows of goods and services. While natural resources must always be extracted from the environment, residuals may flow into the environment, but may be collected by industries to process the waste or may be stored in controlled landfill.

Just as in the SNA, in the SEEA CF one has to consider relations with the rest of the world when accounting for a specific country or region. Materials that flow into the regional economy have to equal the materials that flow out of this economy (UN et al. 2014b, p. 44):

'Materials into the economy = Flows from the environment + imports + residuals received from the rest of the world + residuals recovered from the environment

is equal to

Materials out of the economy = Residual flows to the environment + exports + residuals sent to the rest of the world'.

5.3.3 Environmental asset accounts

A reason to develop the SEEA was to assess to what extent economic activities deplete or degrade environmental assets. The supply and use tables provide insights in the flows in and out of the environment. However, one may be interested in the development of natural capital in this context, where natural capital is the value of the natural assets. The environmental asset accounts are developed as counterpart of the balance sheets in the national accounts.

Natural assets change because of residuals flowing into the environment and resources taken out from them. On top of this, the stock of natural assets changes by other dynamics. However, not all use of environmental assets necessarily results in a reduction of the amount of assets (UN et al. 2014b, Section 2.5). Only if the stock of natural resources decreases as a consequence of using more than what is regenerated, the use of natural assets becomes a problem. For fossil, mineral and metal resources, the depletion equals the amount mined, but for fish depletion, it is all about the difference between the catch and the natural growth of the fish population. For offshore wind energy, resources are not depleted at al. The stock of water in the oceans are neglected, because the stock is too large to be meaningful. However, stocks of fish, minerals and energy are considered. Processes of natural growth of stocks of natural assets are not part of the supply and use tables but are registered explicitly in the asset accounts. Furthermore, catastrophes may reduce stocks independently from normal natural processes. This has to be accounted for explicitly in the environmental asset accounts.

Some statistical issue may change the accounted stock of environmental assets. For example, knowledge about the stocks may change, implying that new discoveries add to the stock, while stocks that were originally seen as relevant may not exist (for example the stock of crude oil reserves). Therefore, estimated stock may change because of reappraisal of the size of the stocks. Finally, some stocks may change because of reclassification. For example, when agricultural land is changed into urban land, this implies a reduction in the stock of agricultural land and an increase in the stock of urban land. Finally, if stocks are measured in monetary units, the value of the stock may change because of price changes, i.e., the revaluation of stock.

5.3.4 Classification of natural resource inputs and residuals

Natural resources inputs

As with classifications of industries and commodities, environmental assets have their own classifications. Table 5.2 below shows the classes of natural inputs.

Table 5.2 *Classification system for natural resource inputs*

1 Natural resource inputs
1.1 Extraction used in production
1.1.1 Mineral and energy resources
1.1.1.1 Oil resources
1.1.1.2 Natural gas resources
1.1.1.3 Coal and peat resources
1.1.1.4 Non-metallic mineral resources (excluding coal and peat resources)
1.1.1.5 Metallic mineral resources
1.1.2 Soil resources (excavated)
1.1.3 Natural timber resources
1.1.4 Natural aquatic resources
1.1.5 Other natural biological resources (excluding timber and aquatic resources)
1.1.6 Water resources
1.1.6.1 Surface water
1.1.6.2 Groundwater
1.1.6.3 Soil water
1.2 Natural resource residuals
2 Inputs of energy from renewable sources
2.1 Solar
2.2 Hydro
2.3 Wind
2.4 Wave and tidal
2.5 Geothermal
2.6 Other electricity and heat
3 Other natural inputs
3.1 Inputs from soil
3.1.1 Soil nutrients
3.1.2 Soil carbon
3.1.3 Other inputs from soil
3.2 Inputs from air
3.2.1 Nitrogen
3.2.2 Oxygen
3.2.3 Carbon dioxide
3.2.4 Other inputs from air
3.3 Other natural inputs n.e.c

Source: UN et al. (2014b, p. 45).

For natural resource inputs, it is a challenge to define at what moment they start to be part of the economy. For biological resources that are in nature, the take-up of for example carbon and nitrogen is part of the flows in the environment, while if the biological resources are cultivated, the uptake of carbon and nitrogen are the resource inputs for the economy. The release of carbon from the soil by cultivated land is seen as an input from the soil that is immediately released.

The purpose of the use of natural resource inputs is to make products. However, not everything that is taken, is used. The part that is not used is called residuals. There are residuals as a consequence of losses during the extraction process (for example gas leakages), residuals that are not useful anymore, such as discarded catch of fish or mining overburden, and residuals from reinjection, i.e., residuals that are directly brought back at another place for later use.

Because of the difference in logic between material flows and monetary flows, some principles of accounting in SNA have to be adapted. For example, if in standard national accounting a firm burns its own waste, it is seen as an activity within the firm. This is not registered in the SNA, because in the SNA only the final inputs and outputs are registered (UN et al. 2014b). However, from an environmental point of view, it may be useful to make explicit that a firm is burning or using its own waste, because it has environmental consequences. This shows the challenges to make the monetary and physical accounts consistent.

Classification and definition of residuals
Residuals can be defined as ‘flows of solid, liquid and gaseous materials, and energy that are discarded, discharged or emitted by establishments and households through processes of production, consumption and accumulation’ (UN et al. 2014b, p. 49). These residuals may be captured, collected, treated, recycled or reused by economic units and this may result in new products, even if the original residual had no value. One has to distinguish between payments for discarding residuals to an industry and the physical flow of the residuals.

In the SNA, it is agreed that consumer durables are statistically consumed in the first year. Therefore, there is not a direct relationship between the formal consumption of the durable in the SNA and the emissions of the consumer durable. The treatment of the consumer durable at the end of its life is in many cases not visible in the SNA. The same holds for capital goods of firms, because depreciation is not directly related with use.

With respect to the recording of scrapped and demolished product assets: they may go to a waste treatment enterprise, go to landfill, go as exports to the rest of the world, or flow directly into the environment (UN et al. 2014b). For all these scrapped and demolished product assets, one would like to be able to trace the sector that has used these products before as capital good or durable consumption good.

There are many specific examples of the application of the SEEA, for example:

- When legal or illegal dumping of waste or for example waste by wreckage or tanker cleaning on sea happens, this is registered as a flow of residuals.
- When products are explicitly emitted as part of the production process, such as fertilisers that are spread on the soil, the part that is used by the plants is part of the product, and the remaining part is registered as a residual flow into the environment. Particles lost for example by tyres when driving, are considered residual flows to the environment.
- Residuals may be recovered from the environment, and in that case a flow of residuals from the environment to the economy is possible.

Residuals have by definition no economic value. In case the discarder receives money for the residual, it is not a residual anymore, but a commodity. The purpose of a circular economy is to transform residuals into commodities as much as possible.

The allocation of the residuals to specific countries is rather hard. The SEEA follows the SNA and therefore attributes residuals to the country of residence of the emitting or discarding household. This implies that emissions and resource use from international transport are allocated to the country of residence of the owner of the international transport company, while the consumption of a tourist in a foreign country is attributed to the country of residence of the tourist, including the fuel used by a hired car. Purchases of a tourist are recorded as exports to the tourist's country of residence. This holds for harvests from natural aquatic resources.

There are some small differences between the registration in supply and use tables in the SNA on the one hand and the physical flows in the SEEA on the other hand (UN et al. 2014b, p. 239). The SNA registers for international transport of goods only the transport service provided. However, for the physical flow registration, the international movements of the products must be registered (UN et al. 2014b, p. 59 and p. 239). Flows within companies are sometimes useful in environmental flow accounts, but are not included in the national accounts, and sometimes there are financial transactions on water between water collectors, water treatment and water suppliers that have no physical flow parallel (UN et al. 2014b, p. 239).

Example of the air pollution in the Netherlands
To give an example, Table 5.3 shows a table with air emissions and how it is related to consumption, a sector and an environmental asset for 2005 and 2017.

Table 5.3 *Air pollution emissions table for the Netherlands in 2005 and 2017*

		2005			2017		
Air pollution emissions	**Units**	**Private consumption**	**Agriculture**	**Landfills**	**Private consumption**	**Agriculture**	**Landfills**
GHG emissions							
CO_2	Mn kg	40,721.0	9,759.0	1,031.0	37,677.0	10,199.0	456.0
N_2O	Mn kg	0.9	23.6	0.0	0.7	21.3	0.0
CH_4	Mn kg	22.9	462.0	233.5	208	541.7	102.8
CO_2 eq.	Mn kg	41,840.0	28,581.0	6,868.0	38,816.0	20,363.0	3,027.0
Acidifying substances							
NO_x	Mn kg	66.1	76.5	0.3	41.7	57.2	0.1
SO_2	Mn kg	0.7	5.9	0.0	0.5	0.3	0.0
NH_3	Mn kg	16.5	134.2	0.0	15.1	114.2	0.0
Acid-eq.	Bn kg	2.4	9.7	0.0	1.9	8.0	0.0

Source: Statistics Netherlands (CBS).[3]

Environmental goods and services and SNA
In SEEA, explicit attention is given to parts of the SNA that are related with environmental goods and services, such as the output of environmental goods and services, expenditures on protection and resource management of natural resources, and taxes and subsidies related with the environment. These accounts are functional accounts. These accounts reorganise information in the SNA for a specific functional purpose (UN et al. 2014b). As discussed before, reorganisation of information is not sufficient for the Central Framework of SEEA, and for ecosystem accounting discussed in the next section even more extra information needs to be added to the SNA system.

5.3.5 Summary

The natural resource use and residuals accounting in the SEEA follows as much as possible the accounting systematics of the SNA, both for flows and for stocks. While the information is for a large part additional, one has to find solutions where direct allocation is not easy. For environmental accounting not only, additional environmental information is needed, but additional economic information has to be added to the national accounts.

3 https://opendata.cbs.nl/statline/#/CBS/nl/dataset/83300ned/table?dl=2B7DD, visited 22 June 2023.

5.4 The SEEA Ecosystem Accounting (SEEA EA)

5.4.1 Introduction

An ecosystem is defined by the CBD as 'a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit' (UN 2021). As an ecosystem is a system, EA is based on a system approach, where different characteristics of the ecosystem are interdependent, in contrast to the focus on individual environmental assets by the Central Framework (UN 2021). While the Central Framework shows direct relations between the economy and the individual environmental assets, the SEEA EA has a system perspective of the environment and emphasises the importance of accumulation, degradation and self-regenerating capacities of the environmental system (UN 2021). For example, while the Central Framework considers only depletion of natural resources, EA considers degradation, i.e. a decrease in the functioning of the system (UN 2021). The Central Framework measures the physical flows between the environment and the economy, the stock of environmental assets and its changes, and economic activities and transactions related to the environment. The Ecosystem Accounting adds to these services from ecosystem processes, such as the regulation of climate, filtration and flood protection, and human engagement with the environment, such as recreation. The ecosystem accounting framework is meant to be consistent with the Central (UN 2021). This implies that the issues discussed in the SEEA Central Framework are relevant for the ecosystem accounting framework.

In contrast with the Central Framework, that basically adds data on emissions and resource use to data on the economy, EA system starts with tracking changes in ecosystems and then relates them to the economy and human activities. EA emphasises the importance of accumulation, degradation and self-regenerating capacities of the environmental system. While in the Central Framework countries are the foundation of measurement, in the EA the spatial area is the basic unit of measurement.

A conceptual framework for ecosystem accounting is represented in Figure 5.1 that is based on the ecosystem accounting approach as implemented by the EU in 2016, which is referred to as Mapping and Assessment of Ecosystems and their Services (MAES). Biodiversity is presented as the key concept to describe ecosystems. The ecosystem delivers ecosystem services, which have benefits for human well-being, and based on the evaluation of the socioeconomic system, one may respond with activities or changes in activities to improve or reduce degradation of the ecosystem. For analytical purposes of understanding the causal chains changes in the accounting period must be attributed to the pressures that cause them. Hereby one has to distinguish human and natural causes and extraction as distinct from regeneration and growth (UN 2021).

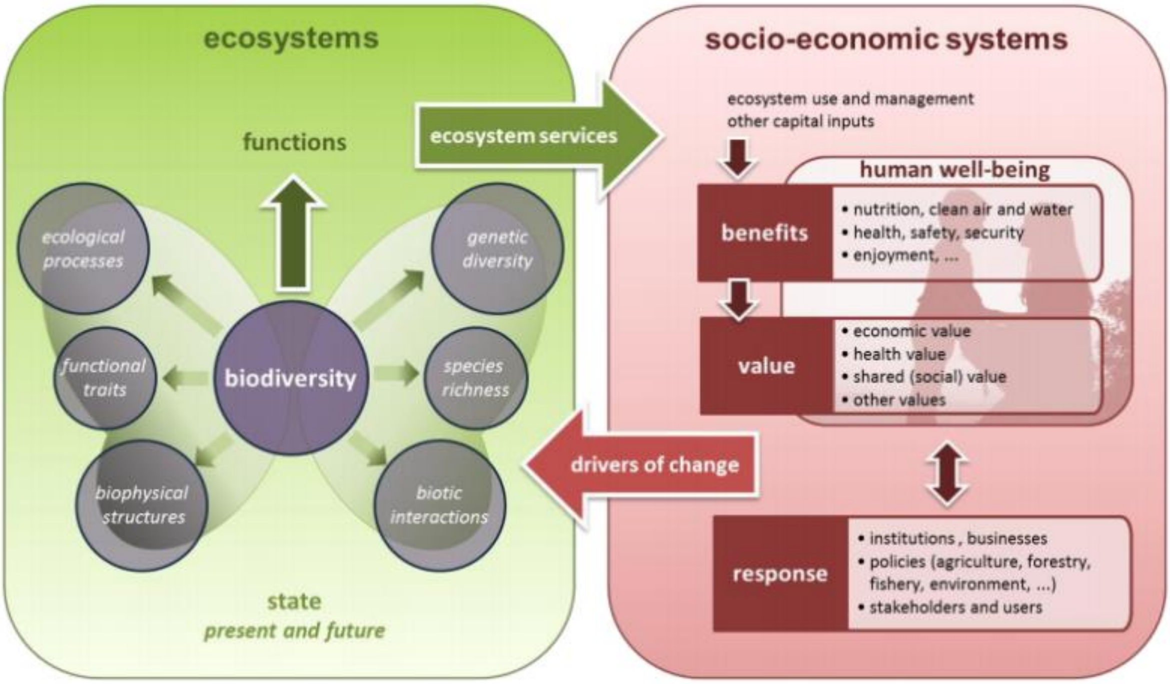

Figure 5.1 *Conceptual framework for EU wide ecosystem assessment*
Source: Maes et al. (2016).

The SEEA EA describes the relationship between ecosystems, the economy and society as shown in Figure 5.2. It shows that ecosystems assets are part of the environment of society and can be described by the extent and condition resulting in specific characteristics that generate the possibility to supply ecosystems. The final ecosystem services depend on demand by society, and these services generate benefits for the economy and society as a whole.

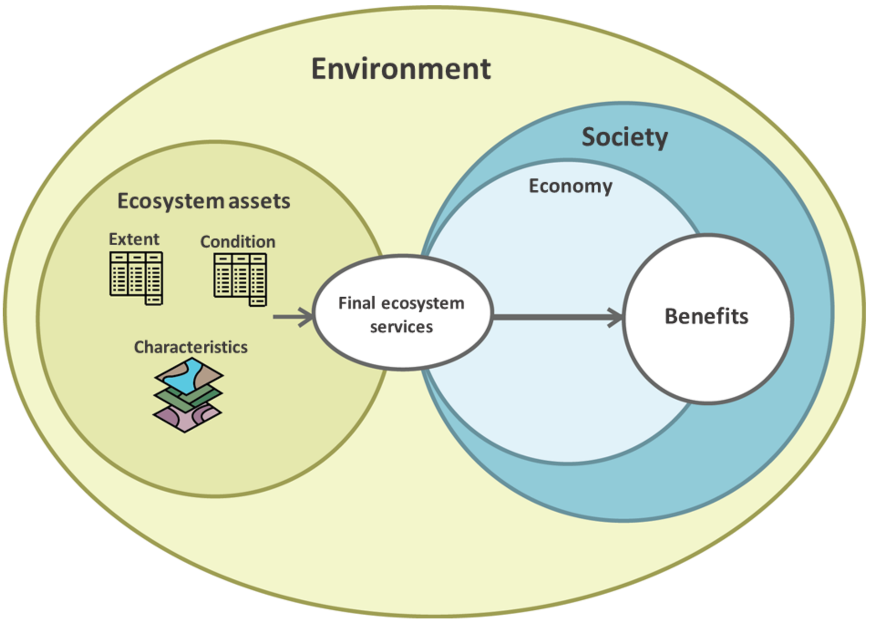

Figure 5.2 *The general ecosystem accounting framework*
Source: UN (2021).

The SEEA EA defines five core ecosystem accounts, of which three are non-monetary (see Figure 5.3). First, there is the ecosystem extent account, i.e., the areas occupied by different ecosystem types. Second, the ecosystem condition account, which describes the condition of each ecosystem. Third, the physical ecosystem services flow account, which shows the supply and use of the ecosystem services and is the parallel of the supply and use table in the SNA and the Central Framework. This supply and use table of ecosystem services may be valued in monetary terms to get the monetary ecosystem services account. One must be aware that the supply of ecosystem services is just the ecosystem services delivered, which is determined by the capacity of the ecosystem to supply ecosystem services and demand for these services. However, one may want to know the capacity of the ecosystem to supply ecosystem services. This capacity is strongly related with the quality of the ecosystem assets. This quality is, for its part, related with the maximum amount of ecosystem services that may be delivered, which is called the sustainable yield. For fisheries, the definition of sustainable yield is relatively easy: it is just the number of fish that one may catch, without reducing the capacity to catch fish in the future. However, since sustainable yield is much more difficult to define for other ecosystem services, the ecosystem capacity account has not been developed yet. Finally, the assets may be valued based on the expected future ecosystem services in monetary terms, to get the monetary ecosystem assets account.

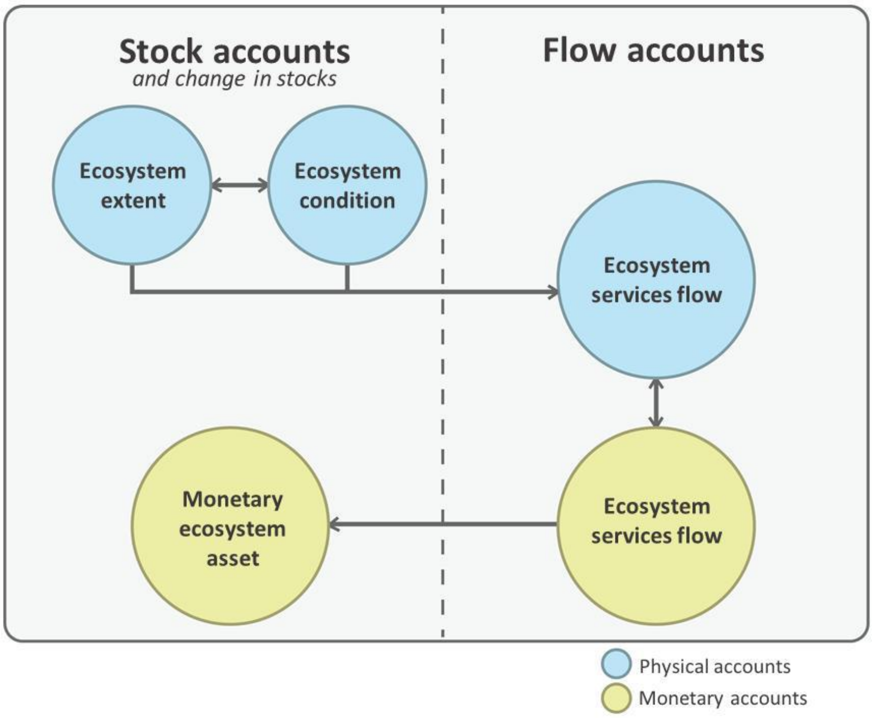

Figure 5.3 *Connections between different accounts in EA*
Source: UN (2021).

Related with these core accounts, thematic accounts may be developed that give information on characteristics of ecosystems. In 2023, accounts have been made for land, water, carbon and biodiversity. Based on the accounting system, one may create different accounts that integrate information of the other accounts. The most simple way is to make combined presentations; a step further is to make extended supply and use accounts and integrate these in the whole sequence of accounts of the SNA, to integrate the ecosystem asset accounts with the balance sheets in the national accounts.

In the rest of this section, the spatial units used in ecosystem accounting will be defined. Ecosystems do not operate at a national scale, but at their own logical spatial scale. Then we go into ecosystems as natural capital in the ecosystem assets accounts and relate these ecosystem assets with the ecosystem services they provide. After a digression on carbon accounting, we finally discuss the monetary valuation of ecosystem services and assets, that has to be consistent with the monetary valuation approach in the SNA. Biodiversity accounting as part of ecosystem accounting is discussed in a separate section (Section 5.5).

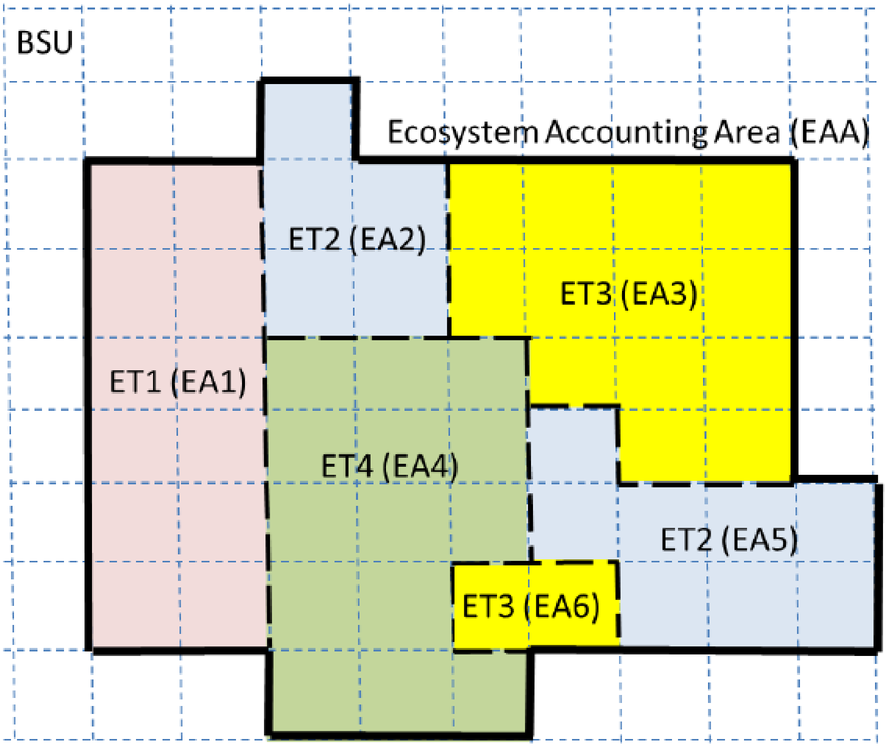

Figure 5.4 *Three levels of spatial detail in ecosystem accounting*
Source: UN (2017).

While the SEEA and SNA have a country or region as the basic spatial unit, for ecosystem accounting, the basic spatial unit is different, and relations have to be developed between this detailed spatial information and the spatial level of the SEEA and SNA. 'Conceptually, for accounting purposes, each area covered by a specific ecosystem type is considered to represent an ecosystem asset. Ecosystem assets are considered to be contiguous, and bounded spatially with each asset comprising all of the relevant biotic and abiotic components within those bounds that are required for it to function and to supply ecosystem services' (UN 2017).

Therefore, ecosystem accounting uses three different spatial scales, as represented in Figure 5.4 (UN 2021, 2017). The first level is the basic statistical unit. 'A BSU is a geometrical construct representing a small spatial area. The purpose of BSUs is to provide a fine-level data framework within which data about a range of characteristics can be incorporated' (UN 2021). This may be grid cells, such as hectares or parcels in the cadastre. The second level, the ecosystem assets, has homogeneity as focal point. Ecosystem Assets (EA) are defined as 'contiguous spaces of a specific ecosystem type characterised by a distinct set of biotic and abiotic components and their interactions', where an ecosystem type (ET) 'reflects a distinct set of abiotic and biotic components and their interactions' (UN 2021). However, statistics need aggregation towards areas that are economically or politically relevant, the Ecosystem Accounting Area (EAA). These are large areas for which there is an interest in understanding and managing changes over time because of social-economic reasons, such as countries, river basins or national parks.

BSUs provide the lowest level of information, and for example remotely sensed data are collected the easiest at that level. However, information is collected at higher levels. Because BSUs are not defined based on coherence, they must in some way be allocated to eAs and EAAs. Normally, predominance is used a criterium for this allocation. To allocate characteristics to a BSU, one may have to downscale from larger areas. For example, to downscale temperature bands to BSUs one may use elevation of the area to estimate the average temperature per BSU (UN 2021).

The Ecosystem Asset (EA) is the logical unit to measure ecosystem characteristics because it is defined to get some coherence with respect to characteristics and therefore it has more system characteristics in common. The EAA is the logical unit for accounting, because accounting regions are stable and therefore measurement is stable as well. Ecosystem services have a specific problem with respect to space. Attribution of ecosystem services for EA requires sometimes first downscaling from EAAs towards BSUs, to get results at EAA level. In ecosystem accounting, the location of beneficiaries of ecosystem services is relevant. If they are outside the area, then those must be considered as imports or exports of ecosystem services (UN 2021).

Aggregation of characteristics of Ecosystem Assets should be done after careful description of measurement, characteristics and the relationships involved. Because of interdependencies between ecosystem characteristics, there is a risk of double counting when different ecosystem services are aggregated into a single ecosystem. To prevent double counting, one may use net recording compared with gross recording as what is exactly measured, but it is advised to report gross data as much as possible, because net data are more elaborated and therefore less reliable. For the same reason, gross domestic product is reported often instead of net domestic product in national accounting, even though net domestic product is the best measure from an analytical point of view (UN 2021).

5.4.2 Ecosystem asset accounts

Ecosystem assets are the basic units of ecosystem accounting, because they represent more or less coherent systems. These ecosystems have characteristics and processes that are related with each other, and therefore have to be described in a logical manner. Together, these characteristics summarise the quality of the ecosystem assets. From this perspective, defining thresholds where ecosystems have a risk to collapse, and resilience (the potential to regenerate) of ecosystems are relevant (UN 2021).

To define quality, in many cases a reference condition is used. The reference condition could be the start of the accounting period, but is often the situation before the industrial revolution, or another moment in time. There is a difference here between the target condition, that implies a preference, and a reference condition, that is just a moment in the past (UN 2021). To define the quality of ecosystems one has to understanding the dynamics of ecosystems.

Human activities, including activities focused on restoring ecosystems or environmental conditions, contribute to the development of ecosystem assets, while different ecosystem assets have relations between themselves. Inter-ecosystem spatial features, such as connectivity and landscape configuration, may therefore be relevant ecosystem characteristics (UN 2021).

Information on ecosystem assets may be collected through remote sensing, on-ground assessment, surveys of landowners and administrative data. If information is not available, one may use value transfer, scaling up from a sample area, or meta-analysis of studies that allocate characteristics based on other characteristics. Furthermore, statistical practices such as sampling, weighting, editing and imputation may be used. Measurement of ecosystem condition is difficult and must be as scientifically sound as possible. The UN therefore advices an accreditation process to guarantee the quality (UN 2021).

Because measuring ecosystems is cost-intensive, one must use information from existing sources as much as possible (see Chapter 3). Therefore, a starting point for characterising ecosystems may be the resource accounts from the SEEA Central Framework, because these give an indication of the conditions that determine the quality of the ecosystems. Examples of these are land accounts, carbon accounts, water accounts, soil accounts, nutrient accounts, forest accounts and biodiversity accounts. For ecosystem extent, land cover (change) accounts are the starting point. For ecosystem condition, water, land cover and carbon accounts may be a good starting point. UN et al. (2014b) provided illustrations for a land cover account (Table 5.13 on p. 179 of UN et al. 2014b), and for a water account (Table 5.25 on p. 214 of UN et al. 2014b). Both illustrations are at an aggregate level of land cover and water classifications, but both the land cover account and the water account can be applied on in more detail for particular classes.

When one has an idea of the extent of ecosystems, one could add the quality of the ecosystem, i.e., the ecosystem condition. Figure 5.4 gives an idea of how a combined ecosystem extent and condition account may look like. The combination of several characteristics of the ecosystem condition gives an idea of the ecosystem quality.

Table 5.4 *Ecosystem extent and condition in one table*

	Ecosystem extent	Characteristic of ecosystem condition				
	Area	Vegetation	Biodiversity	Soil	Water	Carbon
		Examples of indicators				
		LFI, biomass, mean annual increment	Species richness, relative abundance	Soil organic matter content, soil carbon, groundwater table	River flow, water quality, fish species	Net carbon balance, primary productivity
Forest tree cover						
Agriculture land						
Urban and associated developed areas						
Open wetlands						

Source: UN et al. (2014a).

The development of different indicators such as for example in Table 5.4 could be expanded in a table where per EA opening stock, improvements by nature and humans, decreases by humans and nature, catastrophes caused by humans and, nature may be accounted for to get the closing stock. The separation of human influences from other influences is essential to be able to relate the changes in ecosystem condition with human activities and thus the SNA (UN 2021).

5.4.3 Ecosystem service supply and use accounts

Principles
The services that ecosystems provide to humans are called ecosystem services. Because ecosystems generate useful services for humans, they are a type of capital that is comparable with produced capital. Many ecosystem services become only useful if they are combined with human activities. Therefore, this is a type of coproduction (UN 2021).

Per Ecosystem Accounting Area (EAA), one may register per EA the amount (and value) of ecosystem services split into provisioning, regulating and cultural services. Figure 5.5 shows the relationship between ecosystem extent and ecosystem condition, and its capacity to generate ecosystem services. However, the actual production of ecosystem services depends on demand. In that sense, it is confusing that in the ecosystem accounting system, ecosystem service production is called ecosystem service supply. The ecosystem supply and use table shows the ecosystem services produced per EA, and what services are supplied to which sectors of the social-economic system.

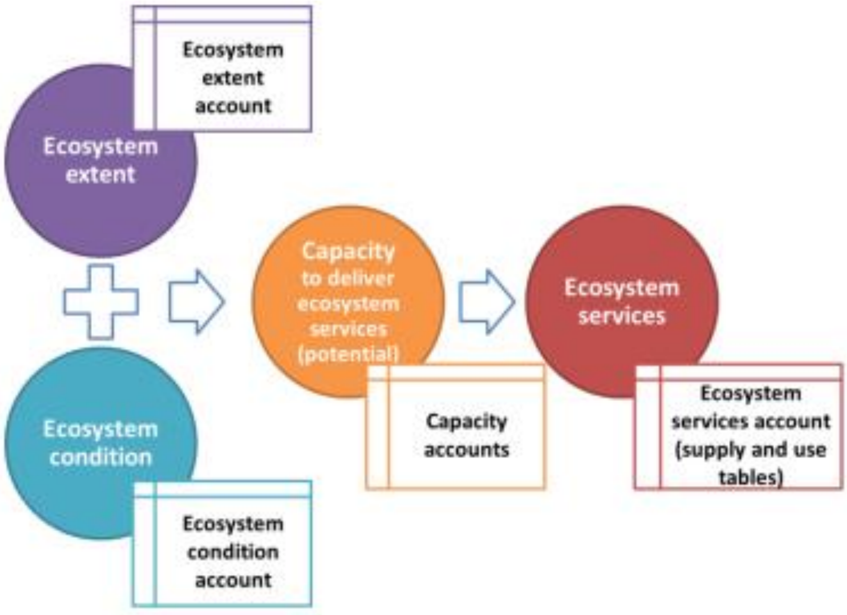

Figure 5.5 *Relation between ecosystem extent, condition, and services*
Source: Maes et al. (2018).

Figure 5.6 shows how in agroecosystems pressures that are partly caused by the socioeconomic system, influence the ecosystem extent and condition. Subsequently, the ecosystem condition influences the ecosystem services. Next, a reduction in ecosystem services could be a motivation to develop policies to improve the ecosystem quality by formulating policy objectives, which is outside the accounting framework, but is one of the motives for developing such an accounting system in the first place.

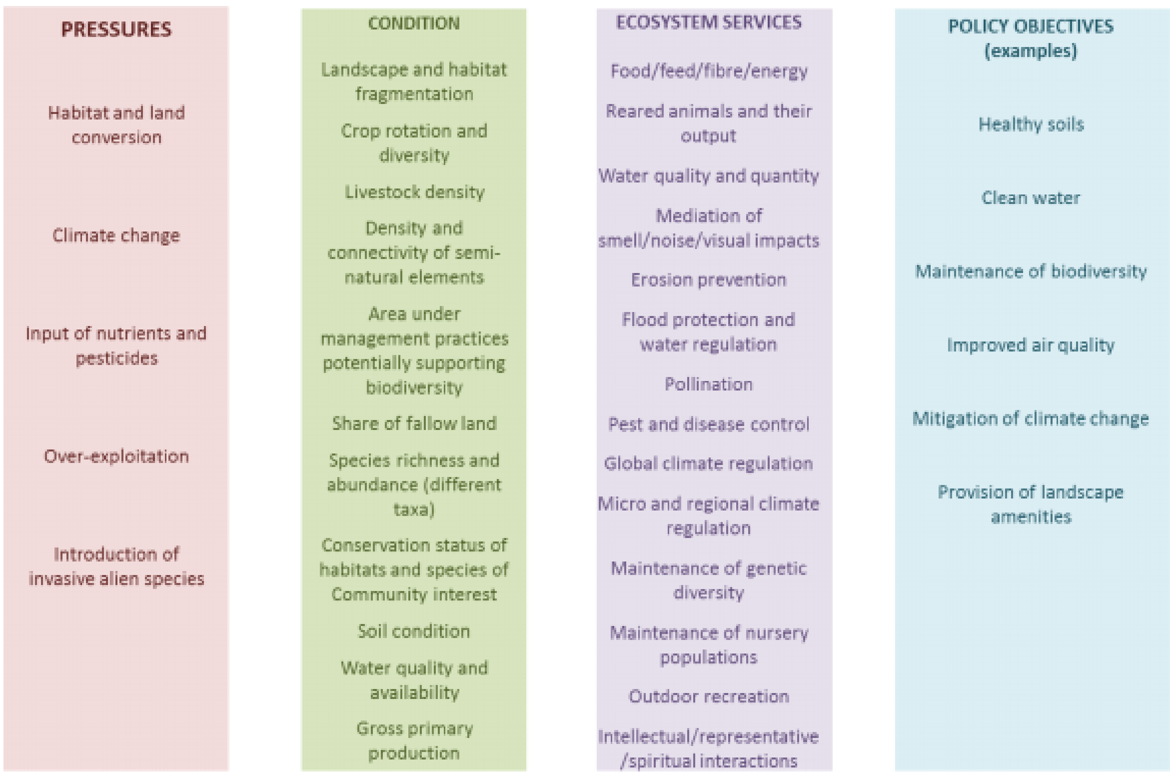

Figure 5.6 *Pressures, condition and services in agroecosystems*
Source: Maes et al. (2018).

One must distinguish between ecosystem benefits that are registered in the SNA, and others that are not. Ecosystem benefits that result in traded commodities are registered in the SNA, while ecosystem services that benefit well-being directly are not. In ecosystem accounting, supporting services for the functioning of ecosystems that have no direct human benefits are excluded from the definition of ecosystem services, because they become only effective at the moment the final ecosystem services are delivered. These supporting ecosystem services are comparable with intermediate deliveries in the economy, which are excluded from final aggregates. However, these supporting services are part of ecosystem assets, because these services influence the functioning of these assets. In current ecosystem accounting, the focus is on final ecosystem services, to prevent double counting and because inter- and intra-ecosystem flows are difficult to measure and handle in accounting (UN 2021). However, in the future they may be analysed from their own logic in the same manner as intermediate deliveries are used for analysing the consequences of final consumption in national accounting.

An ecosystem asset has the capacity to generate ecosystem services. This is called the sustainable yield. However, ecosystem services are measured as the current flows (or the flows in different scenarios), not the potential or sustainable flows. This implies that ecosystems are not valued based on the capacity to provide ecosystem services, but on the actual services expected to be delivered. Without beneficiaries, ecosystem services would not exist. For example, a walking track has higher value when more people use it. Regulating services, such as filtration, materialise only if there is pollution and if people live nearby to benefit from the reduction in pollution. However, the potential of ecosystems to deliver services may be valuable to know, and an ecosystem may deliver inter-ecosystem flows to other ecosystems, thus delivering ecosystem services indirectly (UN 2021).

To have consistent statistical analysis, common classifications are needed. A difference is made between provisioning, regulating and cultural services, where provisioning and cultural services are considered final services, while regulating services may be either final or intermediate (Obst et al. 2016). Accounting for disservices has not been developed but may be relevant. Disservices are basically the opposite of services, outcomes of ecological processes that affect humans in a negative way, such as the spread of diseases, or pest damage to crops (Saunders and Luck 2016). Some sectors have benefits from disservices. For example, pesticide and pharmaceutical industry benefit from health problems generated by ecosystems, such as natural pathogens (UN 2021). However, for the moment SEEA EA handles disservices as decreases in capital in the same manner as depreciation. The Common International Classification of Ecosystem Services (CICES) is used to classify the different ecosystem services in a more detailed manner.

From an analytical perspective, it is relevant to know to what extent provision of ecosystem services is determined by public or private property rights and to what extent ecosystem services are supplied to private agents or to the general public. Especially public benefits on private land may give problems, because it generally is not the intention of the private owner to provide these services to the public (UN 2021).

It is very difficult to analyse all ecosystem services, given that there are so many. The choice of which ecosystem services to analyse may depend on environmental importance, the ability of policies to influence it, and availability of data and methods to measure it (UN 2021).

Example of carbon accounting in the Netherlands

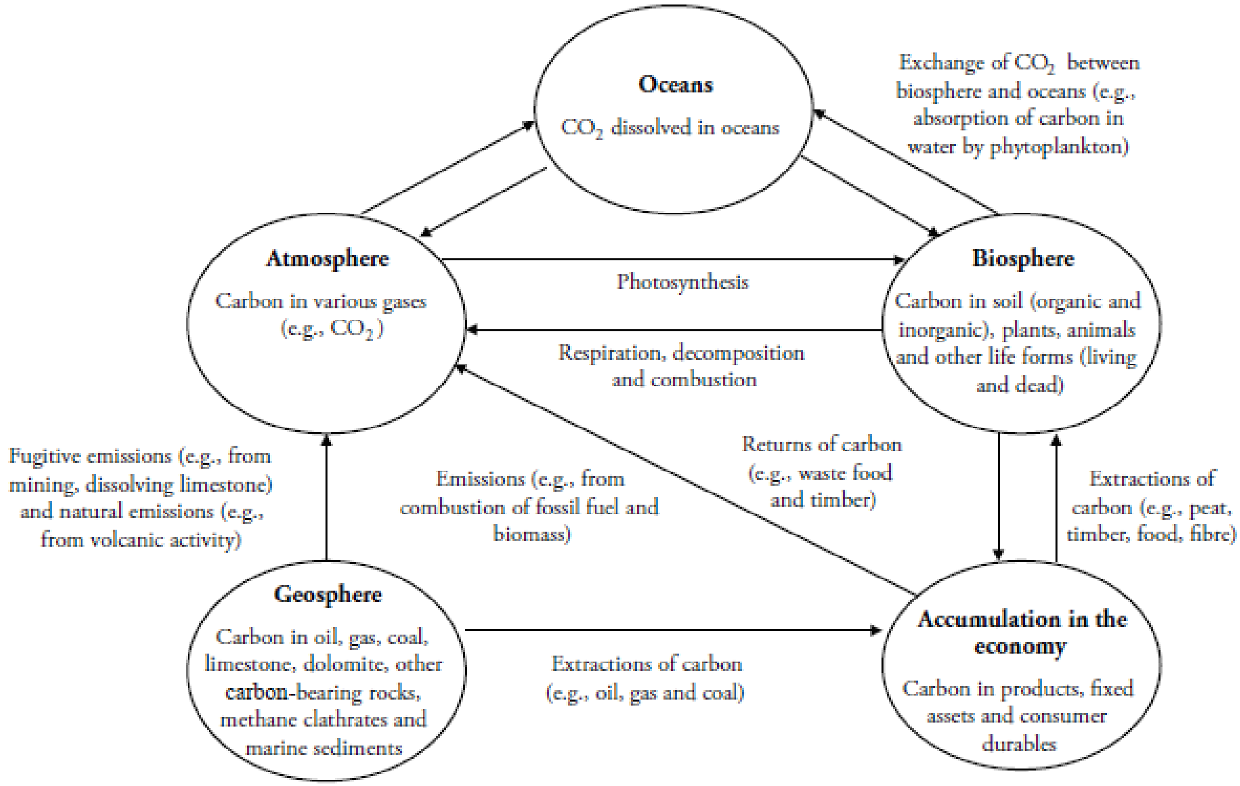

Figure 5.7 *The main components of the carbon cycle*
Source: UN et al. (2014a).

Carbon accounting is a relatively well-developed approach for environmental accounting and may illustrate the difficulties involved in developing a consistent accounting system and the challenge to find the relevant data. CBS (Statistics Netherlands) has developed carbon accounts for the Netherlands (Lof et al. 2017). Figure 5.7 gives an overview of the main flows and components of accounting for the carbon cycle, including the economy. The borderline between the economy at the one hand and the biosphere at the other is not very strict, so there are components and flows that could fit in both the economy and the biosphere. Regarding the carbon accounts, it has been decided to include managed forest in the biosphere, while agricultural products are in the economy. Clear definitions are needed to prevent double counting. Marine and lacustrine carbon and stocks of biocarbon below 30 centimetres are excluded because of practical reasons. Many parts of the accounts are available on a very low spatial aggregation level, although economic data are not available at the same level.

To assess the amount of biocarbon, or the biological storage of carbon, in many cases, look-up tables are used in combination with an ecosystem unit map. Data on stocks were used from Lesschen et al (2012) and De Groot et al (2005). Above ground, biomass and carbon sequestration estimates are based on a look-up table by Remme et al. (2014). Carbon emissions from peat soils are based on a formula that estimates the subsidence in mm as 15.5 * ditch water level (in mm below surface) and a constant of 2.7 for peat without a clay layer and a constant of -3.5 if there is a clay layer. The carbon emission is 0.706 tonne C/ha/year (Van den Akker et al. 2010). Healthy natural bogs (peatlands) sequester carbon (0.27 tonne C/ha/year), but this relates only to a very small part of Dutch peatland. Based on this information, tables on stocks and flows of carbon can be calculated, both on a provincial level and on an ecosystem level (total and per hectare). Maps on a higher level of detail can be made as well. Since carbon sequestration is spatially explicit, it can be represented by a map such as in Figure 5.8. One may aggregate the data behind this map to a table at the provincial level, as in Table 5.5.

Conijn and Lesschen (2015) have more detailed models for carbon balances of grasslands. Tidal marshes have very high sequestration rates. A comparison is made with the LULUCF approach (Coenen et al. 2016). Land use change was excluded from the analysis.

For the economy as a whole, the material monitor, the material flow accounts, and the water and emissions accounts are used for assessing imports and exports. Recycling is included in the analysis. Relations are made with inventories, fixed assets, consumer durables and solid waste. Also, fossil, bio-based and other resources are distinguished. Bio-based products are materials that have carbon compounds derived from plants or animals. Carbon conversion factors for products are used from the Dutch emission inventory and the Phyllis database, developed by ECN.

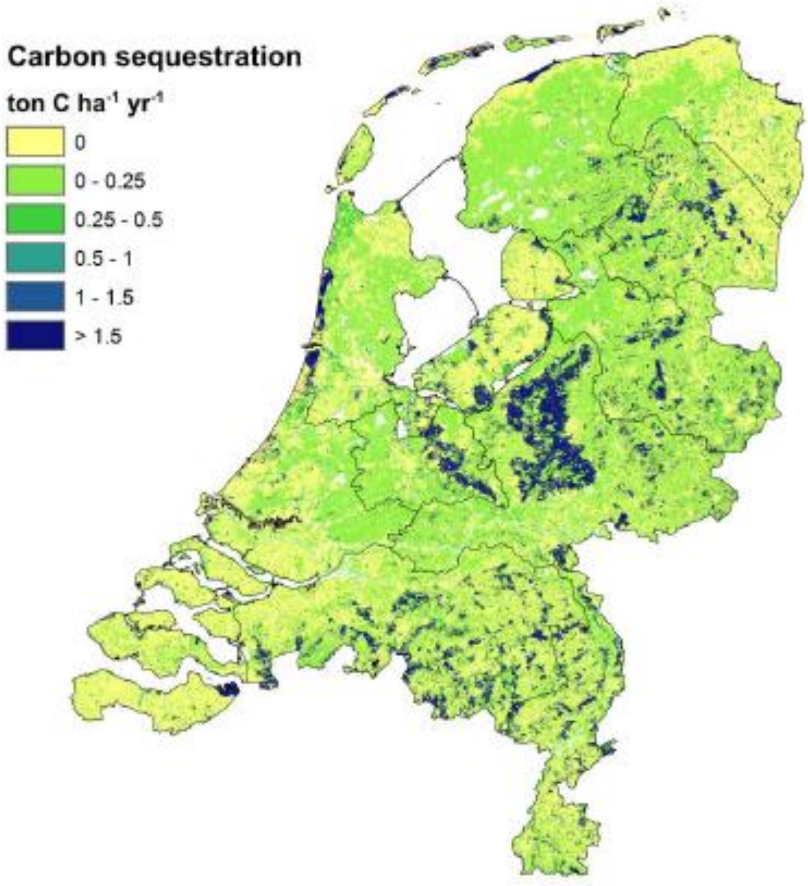

Figure 5.8 *Map of carbon sequestration in biomass for the Netherlands*
Source: Remme et al. (2018, p. 23).

Table 5.5 *Carbon sequestration and carbon sequestration per ha for provinces in the Netherlands*

Province	Total carbon sequestration (ktonne C/yr)	Carbon sequestration per ha (tonne C/ha/yr
Zuid Holland	41	0.13
Groningen	38	0.16
Zeeland	30	0.16
Noord Holland	60	0.21
Flevoland	36	0.25
Friesland	90	0.26
Overijssel	100	0.29
Drenthe	81	0.30
Utrecht	47	0.33
Noord Brabant	163	0.33
Limburg	73	0.33
Gelderland	216	0.42
The Netherlands	975	0.28

Source: Remme et al. (2018).

5.4.4 Valuation of ecosystem services

Principle

In ecosystem accounting, the purpose of valuation is to estimate the value of ecosystem services and assets in monetary terms. Because the ecosystem accounting framework is part of the System of National Accounts, valuation should be based on market values. In theory, market prices are based on supply and demand conditions and market prices equal both marginal cost and marginal benefits. Therefore, the valuation of ecosystem services should be based on valuation of marginal cost or marginal benefits, which implies that consumer's surplus is not included. What valuation approach is appropriate, is strongly determined by what one wants to value; either comparing ecosystem values with national accounting values or deriving aggregates, such as degradation adjusted savings or national income. Sometimes in cost-benefit studies, consumer surpluses are included in the estimates of environmental prices and in this case one should re-estimate values in SNA. In summary, for accounting of ecosystem services, one needs to estimate missing prices, using economic accounting principles, including identification of prices that are implicitly in the values of marketed goods and services (UN 2021).

In analysing ecosystem services, one must be aware that some ecosystem services are used to produce products included in the SNA, but that some ecosystem services are directly supplied to households and governments. These ecosystem services, the final non-SNA ecosystem services, are treated as final consumption of households and governments or exports (Obst et al. 2016, p. 12) and result in a higher estimate of GDP than in the normal national accounts. For estimating the two types of ecosystem services, different methods are needed.

For valuation in ecosystem accounting, one must be aware that ecosystem services have both direct use and indirect use values, where indirect use values include option values and non-use values (UN 2021). All should be tackled in valuation approaches.

Sometimes, ecosystem services are already priced in a way, such as on carbon markets or biodiversity markets. However, these prices normally do not reflect the social willingness to pay, but are specific for the institutional setting of the markets (UN et al. 2014a, p. 120). Therefore, even when markets seem to exist, there may still be externalities to consider.

Different approaches are available to value ecosystem services in a national accounting framework. These may be substitutes to estimate the same value, or they are just different methods for different purposes (Obst et al. 2016, p. 7-8):

- Resource rent
- Indirect estimation through production function estimation with natural resources included

- Replacement cost approaches, i.e., additional costs of the next best alternative for replacing the ecosystem service. For example, the ecosystem service can be replaced by deploying capital and labour instead of using the ecosystem and
- Marginal valuation from revealed demand functions.

Resource rent
A rent is the difference between the sales value of a product and the costs involved in producing the product. For example, the rent of agricultural land equals the value of the products produced and the costs of intermediate inputs, capital and labour. If land is rented, this rent is paid to the landowner. If the land is owned by the producer, the rent may be estimated based on the rent that farmers pay that do not own the land, or by calculating the difference between revenues and costs. Rents are included in the values of the System of National Accounts but are only made explicit in ecosystem accounting as resource rents.

Rent is not always easy to disentangle. For example, the rent paid for land may be partly a reward for the location of the land and partly for the quality of the soil and the biodiversity services in the neighbourhood. As a result, one amount of rent may be paid for more than one ecosystem service (UN 2021, p. 115-116). However, if you have varying combinations of these ecosystem services, one may use this variation to disentangle these different sources of rent.

Although estimating rent can be a useful method, the calculation as the difference between many uncertain accounting values makes it difficult to estimate properly. As a consequence, estimated ecosystem values as rents may sometimes result in negative or very low prices of ecosystem services Obst et al. 2016, p. 12 and p. 16). Therefore, even if the ecosystem service can be seen as a rent, one should be very careful to estimate the value of the resource rent as rent.

If ecosystem assets are not owned by someone, one does not have to pay for it, and therefore no rent emerges. This implies that the values will not be in the SNA. For example, if no charge is paid to a land owner for picking mushrooms, then the value of the ecosystem service provided will be zero (UN 2021). Therefore, the generation of ecosystem services is a productive activity, but it is not automatically priced and therefore an externality. When ecosystem services are properly priced, the rent method may work. Otherwise, other methods should be used as a measure (Obst et al. 2016; Edens and Graveland 2014).

Indirect estimation through production function estimation
When ecosystem services are used in production and these services are not priced properly, one has to focus estimating the value of these ecosystem services in production in another way. One opportunity is to estimate a production function, where natural resources are included as a factor of production. Such a function must be expressed in functional units, because if the prices of natural resources are not part of the cost, they may not be in the prices paid for the products. This already indicates that such an estimation of the production function will not be an easy task.

Replacement and treatment cost approaches
In estimating the value of ecosystem services, one may argue what the costs would be if the same service would have to be provided in another way. This is the replacement cost method, the estimate of manmade alternatives for the ecosystem service. For example, if river water is filtered into tap water quality by the dunes, one may compare the cost of doing this with the cost of directly filtering the water from the river. The difference in the costs is an estimate of the benefit of the filtering service of the dunes. Other examples are manual pollination as replacement of insect pollination, dykes instead of dunes against flooding, pumping instead of natural water storage (UN 2021).

A related approach is the treatment cost method, which is the cost of repairing damages that would occur without the ecosystem service in place. For example, what would the cost of flooding be if the ecosystem service of natural water storage would not be supplied? Out of these two cost methods, one may choose the cheapest as the best estimate of the value of the ecosystem service.

Marginal valuation from revealed demand functions
When a service is directly supplied to humans, one may use methods to estimate the values for the people who benefit from the ecosystem services. Basically, one may differentiate between revealed preference and stated preference methods. Revealed preference methods use information from market decisions to get information on the value of ecosystem services indirectly. For example, one may use differences in prices of houses near nature and further away from nature to estimate the value of nature (hedonic pricing method) or derive the value of a nature reserve by using information of the travel costs to go to the nature reserve (including the cost of time).

Stated preference methods ask directly about valuation, either through a questionnaire (contingent valuation) or by putting people in experimental settings, where they have to make decisions that give information about their preferences such as with choice experiments (UN 2021). Another approach is not to use the valuation approach, but to develop a simulated value of an exchange value in case a market would exist.

This simulated exchange value approach (developed in Spain for forests) estimates demand curves and supply curves and calculates equilibrium prices and quantities based on that (UN 2021). This is a type of scenario approach.

We would like to emphasise that in the context of ecosystem accounting, monetary values should be values in marginal changes, just as is assumed to be the case for normal prices. However, non-linear behaviour of ecosystem services may make it difficult to estimate these marginal prices: for example, a little bit less forest has not much effect on floods when the forest is large, but if the forest is small in size, a little bit less trees will have much more impact. The value of the ecosystem services depends on the amount of people involved in potential floods (UN 2021). Therefore, it is relevant to make a difference between the marginal and average value of ecosystem services. In practice, discontinuities may be less of a problem in estimating marginal cost and benefit curves because of uncertainty. When there is a probability distribution of tipping points, including the damage created when reached, the estimated expected marginal damage function will have no tipping point in it as it is a continuous function.

Because is extremely costly to get information on the value of ecosystem services, it may be useful to use information that is gathered elsewhere. This is called benefit transfer (UN 2021). The easiest way is to use the price used for an ecosystem service at one place at other places. However, this approach would be too easy in most cases, because values provided by ecosystem services depend on the biophysical, economic and institutional context and may depend on the proximity of other ecosystems. Therefore, one would like to have a benefit function that relates different factors to the value of the ecosystem service. Meta-analysis studies may be used to develop this type of function. However, the studies used for meta-regression may differ a lot in terms of quality, methodological approach and valuation concept, and therefore meta-regression is not an easy task (UN 2021). This calls for a critical evaluation of meta-studies.

One final remark: in the SNAs, all flows have positive values. If negative values are not allowed, negative ecosystem services cannot be included (Obst et al. 2016, p. 18). However, if positive ecosystem services would be included, then negative ones should be included, because otherwise it is impossible to calculate net flows of ecosystem services. However, if negative ecosystem service values are added to the SNAs it deviates from the accounting standard in the SNA.

Example of nature-based recreation ecosystem accounts
Ecosystem services accounting is still experimental and requires the development of a number of accounts for different types of ecosystem services. Practical examples of ecosystem services accounts are required to further develop the conceptual and methodological framework proposed by the SEEA EEA (Vallecillo et al. 2019). In their study, Vallecillo et al. (2019) use nature-based recreation as a practical example. It provides a spatially explicit assessment of ecosystem potential to provide nature-based recreation and leisure. It integrates the two components shown in Figure 5.9: (1) ecosystem-based potential, which estimates the potential capacity of ecosystems to support nature-based recreation activities; and (2) human inputs, which integrates a proximity-remoteness concept in relation to road networks and residential areas. Recreational areas close to these infrastructures can be reached more easily and have greater potential for daily nature-

based recreation. This spatial component related to build infrastructure is especially relevant for assessing daily nature-based recreation. Both ecosystem-based potential and human inputs are combined to assess daily recreation opportunities as a measure of recreation potential.

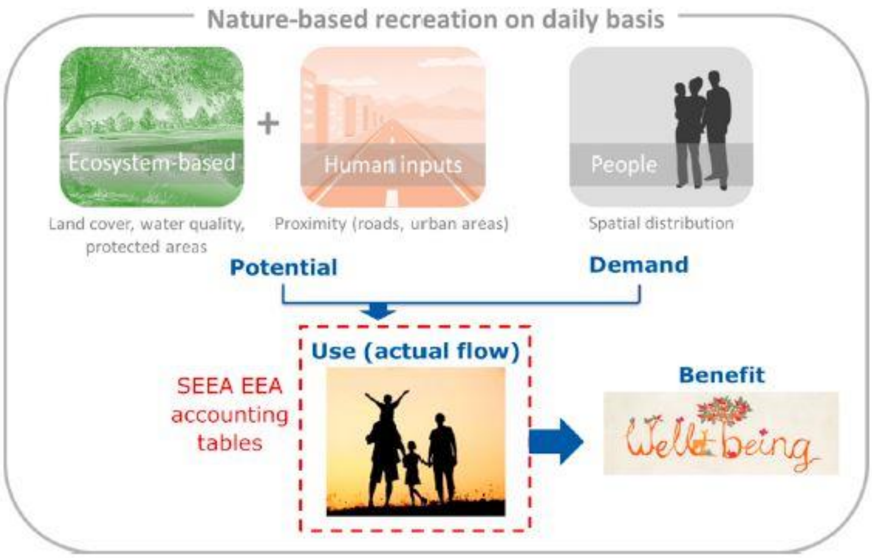

Figure 5.9 *Conceptual model of the components of nature-based recreation accounting*
Source: Vallecillo et al. (2019).

Vallecillo et al. (2019) thus highlight the importance of spatially explicit models for ecosystem services accounts, in which the different components of ecosystem services can be mapped, i.e., potential, demand and flow. Spatial models of ecosystem services are required to properly address the drivers of change in ecosystem services: changes in ecosystems (extent and condition) and changes in socioeconomic systems or the spatial relationship between ecosystems and socioeconomic systems. In addition, using biophysical spatial models in ecosystem services accounts contributes to the development of policy measures targeting the enhancement of natural capital, ecosystem services and the benefits they provide.

To calculate the value of the actual flow of nature-based recreation, the zonal travel cost method (zonal TCM) was used. The zonal TCM is a valuation technique in the family of 'revealed preference techniques', whereby consumers' preferences are disclosed by their purchasing habits. This valuation technique is SEEA-compliant, permitting consistent comparison with valuation reflected in the SNA (UN et al. 2014a). For zonal TCM, consumers' purchasing habits are estimated based on the number of trips that they make at different travel costs. This method was chosen instead of other valuation techniques that might have been more suitable for this purpose, such as hedonic pricing (Liebelt et al. 2018), because of the lack of EU-level data on house prices. Therefore, the travel cost was the most suitable proxy to estimate the exchange value of visits generated at different distances, even when assessing walking/biking trips.

The travel cost method is useful in certain circumstances, but has flaws from both an economist's and an environmentalist's perspective, as Graves (2013) points out. The central theoretical flaw in the travel cost method is that it can only capture use values, shedding no light on non-use (or passive) values, which could be much larger, at least in principle. Moreover, there are additional flaws that have the potential to result in overstatement of use value, further distorting resource allocation against non-use outcomes. To give an example: the travel cost method assumes that a trip is for a single purpose, while that often is not the case. Other potential flaws would result in understatement in use value; for example, the travel cost method does not take into account that people that value nature-based recreation highly may choose to live close to the nature site. While their travel costs will be relatively modest, they may face higher housing prices (Graves 2013).

Valuation of ecosystem assets
Just as produced assets, ecosystem assets produce services and may change in value as a consequence of this production. However, because ecosystem assets have the potential to regenerate and provide multiple services and have varying degrees of use over time, standard valuation of ecosystem assets is more complex than that of produced assets (UN 2021).

With respect to asset valuation, the value can be calculated as the Written Down Replacement Cost (WDRC) of the environmental asset, which is the current replacement cost of the asset minus accumulated depreciation. The WDRC of an asset varies over time, depending on how the asset is depreciated, and what maintenance and renewal works are carried out on the asset. In that sense, it is comparable with valuation in produced assets. Alternatively, the value of ecosystem assets can be computed as the Net Present Value (NPV) of future ecosystem services. These two methods may give different results (UN 2021). In most cases, the last method is the most obvious one, even though it is quite complicated, because the NPV depends on scenario, asset life (with regeneration as a complication), interdependencies between different services now and in the future and the discounting rate (UN 2021).

For correcting national accounts for ecosystem degradation (or regeneration), it is not necessary to know the total value of the ecosystem asset, but only the change of this value. One approach that is sometimes suggested is to use restoration cost as a method to value ecosystem degradation. The restoration cost is the cost to bring the ecosystem assets back to their value at the start of a certain period. However, the step of restoring an asset is a separate decision, i.e. an investment decision, and it is not automatically optimal to do that (Obst et al. 2016, p. 15). Therefore, while the information on the cost to restore can be useful to analyse cost-effectiveness of environmental protection expenditures, it is not in line with the logic of the SNA (Obst et al. 2016, p. 16). Instead, one should focus on the reduction of the net present value of the future ecosystem services (UN 2021).

As a final remark, one must be careful not to double account the value of ecosystem assets and services. If some ecosystem services or assets are for example included in the rent or price of land, then the value of these ecosystem services should not be counted again.

5.4.5 Issues on valuation of ecosystem services in SEEA

SNA values at exchange values, i.e., the price at which goods and services are or could be exchanged. SNA is not about welfare. It is stated that based on the index number theory one can derive welfare change conclusions from market-based values, but this is only valid for small changes. The market price gives information on marginal value and marginal costs. When market prices are not available, valuation may be based on similar goods after adjustment for quality and other differences. If this is not possible, one may use hedonic pricing or avoided damage cost, which does not include consumer surplus. A new approach is the Simulated Exchange Value approach (Caparros et al. 2017).

Caparros et al. (2017) use demand and cost functions to estimate an equilibrium price and quantity to be used in the national accounts. For example, in the application of recreation, they assume zero marginal cost, and profit maximisation, implying that the price is set a revenue maximising value. The data for the demand functions are based on stated preference studies with different estimation techniques. The advantage of the method is that the value created is relatively independent of the estimation technique, which is not the case if the total consumer surplus would be estimated. On top of that, it is more consistent with the use of market prices, which is customary in national accounting. However, a problem with the method is that the price is not paid, and therefore it has budgetary consequences and would normally influence demand for other goods, although obviously the total production value is not changing. Another problem is that the number of the physical accounts is different from the simulated number of visitors, although this may be solved by setting a price of zero for the other visitors. Moreover, the demand for one recreational area depends on the prices in other recreational areas, implying that these interactions have to be modelled. Perhaps this can be solved by conducting questionnaires for all natural parks together.

SEEA discussion paper on valuation

A SEEA discussion paper on valuation (Barton et al. 2019) provides interesting insights. In SNA, the value of non-market output is estimated as the sum of production costs. Some national accountants see ecosystem services as a form of non-market output and therefore estimate it as the cost of supplying them.

The objective of SNA is to 'compile measures of economic activity in accordance with strict accounting conventions based on economic principles'. SNA uses exchange prices, which implies that externalities are not included, and that price discrimination is ruled out, i.e., a seller charges customers different prices for the same product or service based on what the seller thinks they can get the customer to agree to. Household production is not included in SNA, but it can be added in a satellite account.

In the current regime, many ecosystem services are provided free of charge, and in calculating simulated exchange values, one may ask what the most plausible institutional regime would be. The purpose of ecosystem accounting is to make ecosystem services visible. There are ecosystem services provided to firms, to households, services that are co-jointly produced, and services that are not accounted in any value of good or service.

Some regulating services are indirectly reflected in SNA, for example because workers become more productive when they are healthier. In some cases, output in SNA is lower with ecosystem services, for example for flood protection. As a result, ecosystem accounting can both reduce and increase the value of SNA.

In philosophy, environmental ethics states that the living environment has some inalienable legal rights to live and flourish. Social norms and rules are key for regulating ecosystem services (Ostrom 2015). In ecology, the key aspects are diversity, distinctiveness, vital habitat, naturalness and representativeness. It is related to the concept of high conservation value, that is normally defined by experts for management and planning. FSC Australia (2013) defines six criteria: biodiversity concentration, landscape values, threatened ecosystems, natural in critical situation providing biodiversity services, forests fundamental for basic needs of local communities, and forests critical to local communities' traditional cultural identity.

Mazzucato (2018) emphasises that the concept of value changed during history, and that at the moment in neoclassical economics there is a tendency to forget the difference between value creation and value capture. For example, financial institutions claim value creation, while they are only intermediaries in appropriation and distribution of value created by and within other sectors of the economy. There is an emphasis on well-being experienced by the individual, and not groups or societies.

A frequently used approach to valuation is the Total Economic Value (TEV) framework. However, this has the following problems: it does not include entities that have not been identified as having an economic value, it does not distinguish between stock and flow values, and it is inherently static for direct use values without changes in stocks (fish, timber, soil). There is a potential for double counting between use and indirect use values. Moreover, bequest and option values reflect only values of current generations, while these long-term values are especially relevant for future generations. Different estimation methods can make it inconsistent. In short, the TEV framework is not suitable for understanding the total and marginal value of environmental assets. IPBES suggests that decision-making would benefit from plural approaches to value biodiversity and ecosystem services. Valuing ecosystem services through exchange values reframes them as anthropocentric, instrumental, quantifiable and monetised.

Another issue specific for the national accounts is that values in the national accounts are based on transaction values. Transactions will only occur if the willingness to pay for the buyer is larger than the willingness to accept for the seller. The system of national accounts abandoned the goal of being a welfare measure for goods traded in the market. However, Weitzman (1976) and Harberger (1971) showed that small changes in national domestic product in the national accounts can be an indicator of changes in welfare as long as no externalities are involved.

> 'To summarise the discussion above, focusing on welfare measures for ecosystem services would be inconsistent with the level measurements provided by national accounts for goods traded in markets. This would not allow comparing levels and would not allow determining the contribution of ecosystems to economic activity. Furthermore, in level terms, many ecosystem services would have a welfare value that tends to infinity, while neither their exchange value nor their simulated exchange value would tend to infinity. On the other hand, variations in welfare are approximated well by an ecosystem accounting measure based on welfare, but by a measure based on (simulated) exchange values. For these reasons, in this discussion paper, we assume that the goal is maintaining consistency with SNA and, hence, estimating values obtained multiplying prices times quantities for ecosystem services' (Barton et al. 2019).

This idea behind ecosystem service and asset valuation is essential for understanding the use of the values created in case these values would be used for policy evaluation or evaluation of the impacts of firms or consumers on ecosystems.

5.4.6 Summary

Ecosystem accounting provides ample opportunities to relate economic activities to ecosystem changes. However, this requires a careful analysis of the relationships involved and a cost-efficient use of information. Accounts of pressure factors, biodiversity and other factors can be integrated to get insights into causal relationships. Moreover, information from a limited number of ecosystem or biodiversity characteristics can be used to get insight into the whole system. Furthermore, the spatial level of information needed for ecosystem and biodiversity accounting may not be consistent with the information available in standard national accounts. Ecosystems must be understood as systems, which requires spatial aggregations that are consistent with the spatial boundaries of the systems. It is a challenge to relate the changes in ecosystem systems to information on accounting areas used in economic accounting in SNA, and to aggregate the information about ecosystems to levels that are useful from an economic accounting perspective.

5.5 The SEEA Experimental Biodiversity Accounting (SEEA BA)

5.5.1 Introduction

In the final version of SEEA EA biodiversity is included. There is a discussion on the relationship between ecosystem services and biodiversity (see Section 7.3). Biodiversity is not regarded as an ecosystem service in itself, although it has a direct link with ecosystem services and ecosystem and species appreciation. However, there are characteristics of ecosystems that influence the capacity of ecosystems to supply ecosystem services. For instance, the use of ecosystem services such as production of crops may even reduce biodiversity.

5.5.2 Goals of biodiversity accounting

Biodiversity is one of the main characteristics of ecosystem assets. As discussed in chapter 2, biodiversity is the variety among living organisms. Biodiversity accounts measure changes in biodiversity and its causes in a systematic manner.

Biodiversity accounting is meant for different purposes. According to UN et al. (2014a) there are possibilities to link biodiversity accounts to the landcover, land-use and environmental protection expenditure accounts of the UN et al. (2014b) at both national and subnational scales. This linkage can facilitate the analysis of the cost-effectiveness of expenditures on habitat and species conservation and the assessment of returns on investment. Moreover, the UN claims that: 'Using the links to economic accounting in the SEEA, it may be possible to link key drivers of and pressures contributing to biodiversity loss, for example, in terms of measures of energy use, carbon emissions and sinks, built-up land and infrastructure, extraction of fish and timber, agricultural expansion and intensity, climate change, fragmentation and nitrogen deposition and loads. In this context, land-use, land-use intensity and land-cover accounts provide information on the extent of ecosystem types and the area lost through conversion. These kinds of integrated analysis will be facilitated if relevant units (e.g., major land-cover types such as forests, grasslands) can be directly linked to economic units.'

> 'Biodiversity accounts may be relevant in the analysis of ecosystem services, particularly in terms of assessing expected ecosystem services flows. For provisioning services, species are harvested directly for food, fibre, timber or energy. Changes in the abundance of species due to human extractive activities would be reflected in species abundance and status. Harvesting in excess of a species' capacity to regenerate (i.e., unsustainable harvesting) would result in lower yields and reduced economic profit and a higher risk of extinction, and would be reflected in a movement towards higher-risk categories in an account focused on species status.' (UN et al. 2014a, p. 91)

5.5.3 The biodiversity concepts and measurement

Measurement of biodiversity is intertwined with measurement of ecosystem conditions and boundaries (UN et al. 2014a, p. 23) for three reasons:

1. Biodiversity measurements reflect the condition of the ecosystem assets (declining biodiversity corresponds with declining ecosystem condition). But biodiversity is also linked to a final ecosystem service, for example iconic species provide cultural services (UN et al. 2014a, p. 44).
2. Changes in the composition of ecosystem assets and their distribution over land cover types reflect changes in biodiversity.
3. Changes in biodiversity at ecosystem level provide information for biodiversity at the level of species.

In each approach of biodiversity accounting, there are specific assumptions made on biodiversity dynamics. Biodiversity loss is in many approaches related with homogenisation, i.e., different ecosystems get more and more the same composition of species. For specific ecosystems, a change may start with the introduction of invasive species that are able to survive in the environment, while the original species gradually decrease in number and finally may even become extinct (when it happens in all comparable ecosystems everywhere in the world). This is a reason to measure species richness and compare it with the original state of an ecosystem, as is done in the MSA indicator and the PDF indicator. The advantage is that if one assumes that MSA and PDF provide an indication for homogenisation, these measures can be aggregated over space. However, one must be aware that these measures only measure part of the relevant biodiversity issues.

Aggregation of biodiversity over space is complex, but it is possible, although it depends on the indicator chosen. In many approaches, a biodiversity indicator is measured relative to a benchmark. For example, in the WFD, the measurement of threatened species uses the pre-industrial period as a benchmark UN et al. 2014a, p. 86). However, these measurements do not differentiate between different types of ecosystems or different levels of natural biodiversity, the scarcity of ecosystem types or the scarcity of species in the ecosystems.

As said, MSA and PDF are not capturing all relevant aspects of biodiversity. In 2003, the CBD agreed upon four indicators including the MSA (UN et al. 2014a, p. 92):

- Trends in extent of selected ecosystems
- Trends in abundance and distribution of selected species (MSA)
- Trends in status of threatened species and
- Change in genetic biodiversity.

On a larger scale, information from for example the IUCN Red List of Threatened Species may be added UN et al. 2014a, p. 94).

Preferably, species abundance must be measured for all groups of species such as animals, fungi, Protista and plants and at the BSUs or EAs levels. In practice, however, information is usually available at higher levels, and only for a selection of species (UN et al. 2014a, p. 95). Therefore, the challenge is to use available information as efficient as possible and to guide investments in new measurement in such a manner that as much relevant information as possible is assembled at the lowest cost, see Section 2.3.

5.5.4 Some examples of tables for biodiversity accounting

There have been published a few number of examples on practical approaches to biodiversity accounting, such as i) an ecosystem extent account, ii) species composition in an Ecosystem Asset (EA), iii) biodiversity health accounting, and iv) Biodiversity accounting according to abundance.

Table 5.6 provides an example of an ecosystem extent account. The final biodiversity indicator is calculated as habitat area multiplied by a headline indicator of biodiversity based on species in regions. Although the calculation of biodiversity as a linear function of area makes relating biodiversity to pressure factors relatively easy, this does not automatically mean that one can aggregate biodiversity indicators in this manner. In such an accounting system, one would like to have the causes of the increases and decreases of biodiversity to be included. To get a step in that direction, the rows with additions and reductions may be split over different causes. However, in order the whole causal chain from the inventory of environmental interactions of the economy to midpoint and endpoint indicators must be followed, as discussed in Section 4.2.4 on LCA.

Table 5.6 *Species account for Ecosystem Extent Account for the E–01-- HABITAT CLASS (e.g., Plains Woodland)*

	Habitat Area*	**Species Indicators**				**Headline Species Indicator (HI)**	**Stock of Biodiversity (HI * habitat area)**
		SI1	**SI2**	**SI3**	**SI4**		
		Input	**Input**	**Input**	**Input**	**Output**	
	indicator weightI	**W1**	**W2**	**W3**	**W4**	**=W1*SI1 +W2*SI2 +W3*SI3 +W4*SI4**	
Open							
Additions	*						
Reductions	*						
Close							
Net Change							
Reference							N/A

Source: UN et al. (2017, Table 4).

Table 5.7 summarises species composition in an Ecosystem Asset (EA), that indicates whether or not a native species is being protected or whether the species are rare species. The table is specified by a classification system of species. On top of that, one may have a specific account for categories of Red List species. One must be aware that changes in the number of species in each Red List category may be caused by different statistical reasons.

Figure 5.18 shows a categorisation scheme for this. It makes a distinction between changes in biodiversity because of extinction and reclassification, and several other reasons. Obviously, more steps must be included to trace and evaluate the question to what extent the change is negative or positive for biodiversity in general.

Table 5.7 *Burdekin NRM species status in 2000*

	Introduced species	Native species				Total species
		Unprotected	Protected	Total	Rare and endangered	
Animals	32	69	825	894	88	926
Mammals	15	2	112	114	20	129
Birds	10	0	458	458	33	468
Reptiles	2	0	202	202	26	204
Amphibians	1	0	51	51	9	52
Bony fish	4	56	0	56	0	60
Cartilaginous fish	NA	NA	NA	NA	NA	NA
Insects	0	11	2	13	0	13
Plants	376	5	3,239	3,244	91	6,320
Fungi	0	0	68	68	0	68
Protista	0	0	148	148	0	148
Total	408	74	4,280	4,354	179	4,762

Source: Bond et al. (2013).

Table 5.8 *Account of changes in number of species of different Red List categories*

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

Table 5.9 gives another aspect of ecosystems: biodiversity health. Additions and reductions of diseases and toxins are registered, and other health measures are included. Again, the next step would be to explain the changes.

Table 5.9 *Biodiversity health accounting (EU01/EUA01)*

	Animal health measures		Microorganism health measures	Plant health measures
	Disease	Toxins		
Opening level				
Additions				
Reductions				
Closing level				
Net change				
Reference level				

The next step is to measure both the number of species, and the number of individuals of each species. Thus, instead of the number of species that are available, one may include the number of individuals per species. This can be done for all species together, but it would be more useful to have it for all species separately. Again, here arise aggregation issues. By relating the current number of species to a reference point, one may aggregate the species richness.

Table 5.10 *Biodiversity accounting according to abundance*

	Animals					Plants	Headline indicator (output indicator)
	Mammals	Birds	Reptiles	Fish	Invertebrates (e.g., bees and butterflies)		
Relative measures							
Opening population as a proportion of the reference population							
Closing population as a proportion of the reference population							
Net change							

The sketch above in Table 5.10 gives a first indication of how biodiversity accounting could be put into practice. This table has to be developed much further.

Although there is still little experience with this new kind of biodiversity accounts, Bogaart et al. (2020) compiled the first biodiversity accounts for the Netherlands using the SEEA EA framework as a guidance. The Dutch Biodiversity Account is mostly based on official biodiversity indicators, published by the Environmental Data Compendium (CLO). For this account, indicators were collected on two hierarchical levels: ecosystems and species. These levels are tightly coupled, since many ecosystem indicators are constructed from species abundance and/or distribution information. The Red List Indicator (status, length, colour) and Living Planet Index are the most relevant indicators. This report on Dutch Biodiversity Account is valuable because of its elaboration of biodiversity accounting with underlying biodiversity indicators.

It is a challenge to account for biodiversity change from different perspectives. Moreover, many steps are needed to make it possible to derive relations between economic developments and biodiversity changes explicitly. One may question to what extent causality should be completely includes in the biodiversity accounts, or to what extent the accounts should be descriptive only.

5.6 Example of marine ecosystem and biodiversity accounting in the EU

5.6.1 Introduction

In this section, an overview is provided for the current knowledge and developments in marine ecosystem and biodiversity accounting in the following four subsections: marine biodiversity state and trends, impact of fishing on marine ecosystem accounts, physical SEEA EA accounts for the marine ecosystem, and ecosystem services studies, such as for Marine Strategy Framework Directive (MSFD) (2008/56/EC).

5.6.2 Marine biodiversity state and trends

In preparation for the of the MSFD, Bos et al. (2011) explored how ecological data could be used to provide marine biodiversity information on the three different levels (species, habitat and ecosystem) that are proposed in the 2010 Commission Decision on the criteria and methodological standards for Good Environmental Sta–us (GE–) - descriptor 1. Each level is divided in sublevels. For the species, level information is required on species distribution (1.1), population size (1.2), and population condition (1.3). For the habitat level, information is asked for habitat distribution (1.4), extent (1.5) and condition (1.6). For the ecosystem level, information on ecosystem structure is required (1.7). Bos et al. (2011) defined specifically for the Netherlands a set of 13 metrics of biodiversity, covering the width of the Commission Decision criteria (2008/56/EC). Table 5.11 provides an overview how the 13 marine biodiversity metrics relate to species groups and to the criteria of the GES descriptor 1. Bos et al. (2011) did not include a metric for the information on the ecosystem structure in their list.

Table 5.11 *Overview of the 13 marine biodiversity metrics of Bos et al. (2011) and the criteria of the GES descriptor-1 a)*

		Groups					**Criteria of GES descriptor 1**					
	Metrics of biodiversity developed by Bos et al. (2011)	**Benthos**	**birds**	**fish**	**mammals**	**habitat**	**1.1**	**1.2**	**1.3**	**1.4**	**1.5**	**1.6**
1	Distribution	V	V	V	V	V	C			C	C	C
2	Density	V	V	X	V			C				
3	Biomass	V		X				C				
4	Resilience (vulnerability)	V	V	X	V	X			C			
5	Dependence on the marine environment (birds)		V						C			
6	breeding in the Netherlands		V						C			
7	importance of the Dutch Continental Shelf (DCS) for the species	X	V	X	V				C			C
8	trends	X	X	V	X				C			
9	rarity	V	V	V	V	V	C		C	CV		
10	large specimens within populations	V		V			C		C			C
11	(potentially) large species	X		V					C			C
12	species richness	V	V	V	V							C
13	species evenness	V	X	V								C

a) V = available information of taxon (groups), X = unavailable information of taxon (groups) and C = correspondence criteria. Empty spaces indicate irrelevance.

Source: Bos et al. (2011, Table 4, p. 19)

Not all metrics were applicable to all datasets. They constructed maps per biodiversity metric and per taxonomical group and concluded that separate maps of biodiversity metrics are most informative, allowing to draw conclusions. Biodiversity patterns have to be assessed at the relevant scale. Bos et al. (2011) showed that biodiversity hotspots can be identified and used as a starting point for spatial management, and they provided an overview of biodiversity characteristics per area within the Dutch part of the North Sea, and of the Natura 2000 status of the areas is given by Bos et al. (2011).

Tamis et al. (2019) contributed to a nature balance for the Dutch Continent Shelf (DCS) which is part of the North Sea. It is a useful overview of indicators related with activities, pressures, ecosystem (state) and policies of the marine sector according to the DPSIR framework and applying the cumulative impact assessment (CIA) method (see Section 4.6). Some attention is paid to human impacts by showing the ecosystem services on production (abiotic material, energy nutrients, biomass for energy and other purposes, food), cultural services (recreation, spiritual and religious values for structures and sorts of terrain) and regulating services (physical and chemical regulation for carbon and water, currents such as erosion, protection of the shore, and waste water purification). The state of the ecosystem is evaluated by evaluation of the state for the ecological components: marine mammals, seabirds, fish, seafloor and water column. The pressure factors are pollution, litter, noise, extraction of species, change in hydrography, migration barriers, disturbance and physical loss, introduction of non-indigenous species, eutrophication. The activities are shipping, fishing, activities on land, dredging, windfarms, oil and gas activities, tourism and recreation, shore defence and new land, organisation of rivers, defence, harbours, aquaculture and sand extraction. For nature structure and functioning and human activities and pressures on the one hand and corresponding MSFD descriptors on the other hand, three situations of the North Sea were evaluated: current state and trend, pressures and activities, environmental targets and policy measures.

5.6.3 Impact of fishing on marine ecosystem accounts

Additional to the indicators used by the MSFD to track the ecosystem state in relation to the GES, there was the need for another class of 'surveillance' indicators. Shephard et al. (2015) proposed surveillance indicators which monitor key aspects of the ecosystem for which there is insufficient evidence to define targets and support formal state assessment; and/or where links to anthropogenic pressures are either weak or not sufficiently well understood to underpin specific management advice. Surveillance indicators are not only expected to directly track state in relation to GES, but to provide complementary information (including warning signals) that presents a broader and more holistic picture of state, and inform and support science, policy, and management.

The 'Wild Seafood' Provisioning Service (WSPS), on which commercial fisheries rely, is probably one of the best studied marine ecosystem services, due to its economic relevance and because extensive information sources exist for assessment purposes. Yet, the indicators often proposed are not suitable to describe the capacity of the ecosystem to deliver the WSPS. Therefore, Piet et al. (2017b) proposed surplus production (SP), a well-established concept in fisheries science, as the basis to calculate the capacity of marine ecosystems to provide the WSPS. SP is defined as the difference between stock production (through recruitment and body growth) and losses through natural mortality. This is, therefore, the production of the stock that could be harvested sustainably without decreasing the biomass. To assess the sustainability of the exploitation of the WSPS, they developed an indicator for this based on surplus production and compared it with existing fisheries management indicators. When both SP-based indicators showed a decreasing trend, contrasting with an increasing trend in the existing fisheries management indicators, the calculation of the SP-based indicators was scrutinised, revealing that the weighting of the stocks into an aggregated indicator, strongly determines the indicator values, even up to the point that the trend is reversed. The aggregated indicators based on SP-weighted stocks can be considered complementary to existing fisheries management indicators, as the former accurately reflect the capacity of the commercial fish to provide the WSPS and the sustainability of the exploitation of this service. In contrast, the existing fisheries management indicators primarily reflect the performance of management towards achieving fisheries-specific policy goals (Piet et al. 2017b).

Piet and Gelabert (2019) developed a pilot European Seafloor account from a state-pressure-impact perspective, although the impact part is not very well developed. Ecosystem health is measured as the biomass of the biota that make up the seabed habitat, i.e., benthic invertebrate community, relative to an undisturbed situation. This is considered an adequate first approximation for biodiversity in relation to the capacity to supply many ecosystem services. To better assess impact, the benthic invertebrate biomass is subdivided in four longevity classes, i.e., the average time that a benthic invertebrate species lives, to reflect their sensitivity to additional human-induced mortality. The assumption is that a disturbance by fishery will have a longer effect for species that live longer, so bottom trawling shifts species composition from long lived to short lived taxa. The change in the biomass is calculated as the natural growth based on a logistic growth model which is habitat specific and the reduction due to gear-specific fishing intensity. The causal relationship is between fishery intensity and sea floor integrity, based on a recent global meta-analysis (ICES 2016). For fishing, intensity information from Vessel Monitoring through Satellite (VMS) is combined with fisheries logbook data, to get per grid cell insight into the area of the seabed that is in contact with the fishing gear as fraction of the area of the grid cell, the so-called Swept Area Ratios. For the pilot seafloor integrity account (SIA), ICES (2018) is used in combination with the EU SeaMap habitat map.[4] The result is

[4] www.Emodnet-seabedhabitats.eu/access-data/download-data

an accounting table that shows the development of SIA (% relative to the undisturbed situation) for four longevity classes, and the inflows and outflows. The new situation is just the old situation plus inflow minus outflow, calculated based on an impact assessment model, habitat maps and fishing intensity data. In the discussion it is concluded that most of the relevant area is included, and that fishing is the main activity among human activities, including man-made structures. It should be realised that the focus was on the benthic invertebrate community and the ecosystem services supplied by it. Further improvements would need to include detail on the composition of the benthic community in terms of functional traits.

5.6.4 Physical SEEA EA accounts for the marine ecosystem

Graveland et al. (2017) have examined whether and how natural capital (for ecosystem services and ecosystem condition) could be compiled and implemented for the DCS. Globally, there is still little experience with the account’s compilation for the marine environment. Ecosystem accounts for marine areas such as the DCS are thus still very experimental, and much still has to be developed, tested and learned. Graveland et al. (2017) conducted a preliminary study, utilising the relatively many different data sources that are available for the North Sea as well as the experience from a parallel study on the compilation of natural capital accounts for the terrestrial part of the Netherlands. This resulted in a first pilot compilation of marine natural capital accounts for the DCS. Graveland et al. (2017) recommend as a possible next step to initiate a pilot project with a small set of accounts which have a limited scope with regard to the number of condition indicators and the number of ecosystem services and to extend that over many years to keep the process manageable with respect to budget and required capacity.

In 2020, Horlings et al. (2020) have produced the first ecosystem service supply and use accounts and ecosystem asset accounts for the Netherlands, based on the SEEA-EA framework (see Section 5.4). The authors state that the results are not complete, because they do not represent the total or ‘true’ value of nature. Notable omissions are marine and freshwater services, flood control and coastal protection, and fishing. Physical estimates of fishing and other marine ecosystem services have yet to be developed.

The condition account presents indicators for the general condition or state of an ecosystem and indicators for pressure that can affect ecosystem functioning. State indicators reflect the state or condition of vegetation, biodiversity (or nature value), soil, water and air. The biodiversity indicator ‘Living Planet Index’ shows that in several ecosystem types, including marine ecosystems, biodiversity has decreased since 1990 (Lof et al. 2019).

5.6.5 Ecosystem services studies

It is well known that protecting the full range of ecosystem services is best achieved by maintaining biodiversity overall (Worm et al. 2006).

Broszeit et al. (2017) considered whether a selection of GES indicators related to biological descriptors, D1 Biodiversity, D2 Non-indigenous species, D4 Food webs and D6 Seafloor integrity, may provide information relevant to ecosystem services, potentially allowing use of collected environmental data for more than one purpose. Published lists of indicators for seven selected marine ecosystem services were compared to 296 MSFD-relevant biodiversity-related indicators taken from the DEVOTOOL catalogue, which is regretfully not available anymore. The authors concluded that 64 of these biodiversity indicators are directly comparable to the ecosystem service indicators under consideration and 247 biodiversity indicators were identified as potentially useful ecosystem service indicators (Broszeit et al. 2017). In other words, indicators related to the two fields of interest (biodiversity and ecosystem services) show high similarities (Figure 5.10). Broszeit et al. (2017) suggest that future monitoring effort can be used not only to ensure that GES is attained, but that ecosystem service provision is maximised.

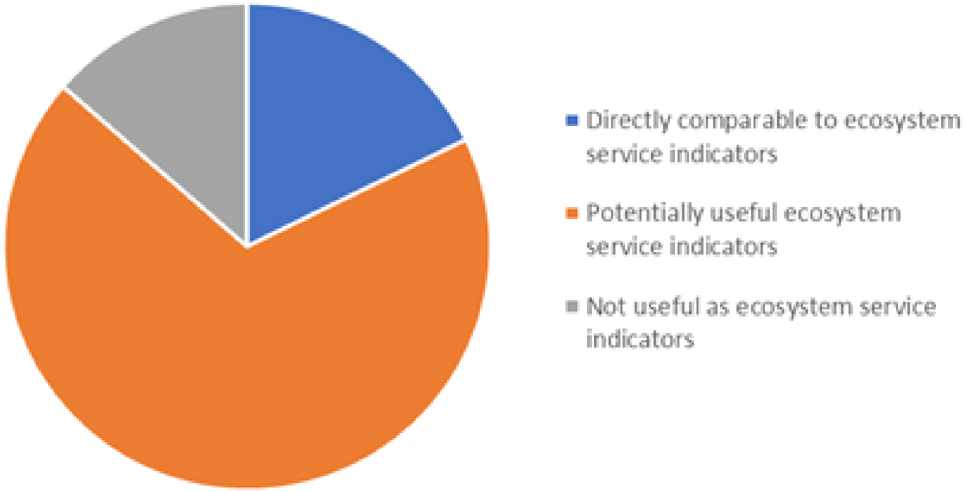

Figure 5.10 *Suitability of MSFD-relevant biodiversity-related indicators as ecosystem service indicator Based on data from Broszeit et al. (2017).*

An interdisciplinary approach to the assessment of natural capital and ecosystem services in marine ecosystems was adopted by Buonocore et al. (2020). In particular, the study aimed at assessing the biophysical value of natural capital stocks in a Mediterranean Marine Protected Area (MPA), through the parallel use of the emergy and eco-exergy accounting methods. Emergy accounting converts the thermodynamic basis of all forms of energy, resources and human services into equivalents of a single form of energy, usually solar. To evaluate a system, a system diagram organises the evaluation and account for energy inputs and outflows. Eco-exergy is defined as the work capacity of an ecosystem compared with the same system at thermodynamic equilibrium (Jørgensen 2006). The assessment focused on four main macro-habitats: sciaphilic hard bottom (coralligenous bioconstructions), photophilic hard bottom, soft bottom, and Posidonia oceanica seagrass beds (Buonocore et al. 2020). In addition, to complement the biophysical assessment with an economic perspective, the emergy values of natural capital stocks were converted into monetary units. The study aimed at identifying a set of ecosystem services generated by Posidonia oceanica seagrass beds and Coralligenous bioconstructions and estimating their economic value. The main steps of the implemented environmental accounting model are summarised in the conceptual diagram (Figure 5.11).

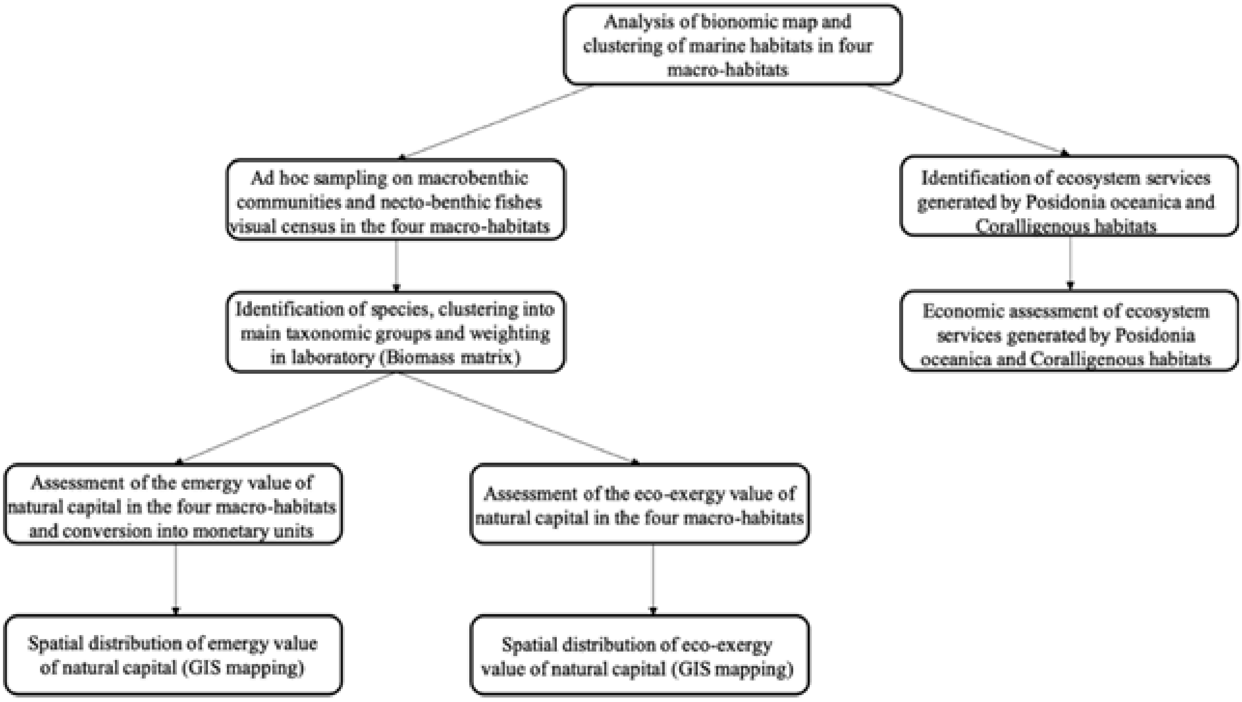

Figure 5.11 *Conceptual model of the environmental accounting model implemented in the study of Buonocore et al. (2020)*

Culhane et al. (2020) assessed the capacity of marine ecosystems to supply services using available policy-reported assessment information on marine biodiversity and ecosystem status. The assessment consists of three key steps:

1. identifying all the instances where a marine ecosystem component can potentially contribute to the supply of a marine ecosystem service
2. developing a critical pathway analysis to identify the major ecosystem component(s) contributing to the supply of a given service and
3. interpreting available information on the state and trends of these major contributing components with knowledge of the ecosystem state-service relationship, to assess the ecosystem's capacity for service supply and its direction of change (Culhane et al. 2020).

Case studies were carried out showing the outcome as a direction of change, see Table 5.12.

Table 5.12 *Summary of the outcomes of service supply capacity assessments of the three MECSA case studies carried out using information from EU and other policy reporting*

Marine ecosystem service	**Critical ecosystem component/SPU**	**Marine (sub) region assessed**	**Reason for choice of marine (sub-) region**	**Assessment outcome [a)]**
Recreation and leisure from whale watching Cultural	Whale and dolphin species relevant for whale watching	Mediterranean Sea	Whale (and other cetacean) watching is a growing commercial industry in this region	Bad current service supply capacity, which has been deteriorating
Nutrient waste removal and storage Regulation and maintenance	Phytoplankton in all water column habitats	Baltic Sea	Eutrophication is a significant pressure across, or in certain parts of, all regions	Bad current service supply capacity Unable to assess: Direction of change of (current) service supply capacity
Seafood from wild fish and shellfish Provisioning	Commercial fish and shellfish (epifauna) species in all habitats where commercial fisheries take place	North Sea	Commercial fisheries are a significant pressure across all marine regions	Good service supply capacity, which has been 'improving'

a) Current service supply capacity and direction of change.

Source: Culhane et al. (2020).

Two of the grand challenges in marine ecosystem ecology as defined by Borja et al. (2020) are explicitly related to biodiversity: 1) Understanding of interaction among biodiversity and ecosystem processes and; 2) Measuring ecosystems shifts, biodiversity and habitat loss.

The most widely recognised framework for linking biodiversity and human well-being is the ecosystem services (ES) cascade framework (Potschin-Young et al. 2018), which works with 'production chains' of ES, starting from the biophysical structures and processes of an ecosystem and ending with the societal benefits (Liquete et al. 2013). However, quantitative relationships between biodiversity, ecosystem functioning and ES, are still poorly understood (Armoškaitė et al. 2020). Armoškaitė et al. (2020) advance these assessments by providing an assessment tool, which links marine ecosystem components, functions and services, and graphically represents the assessment process and its results.

The tool consists of two parts:

1. a matrix following the ecosystem services cascade structure for quantifying the contribution of ecosystem components in the provision of ecosystem services;
2. and a linkage diagram for visualising the interactions between the elements.

With the aid of the Common International Classification of Ecosystem Services (CICES), the tool was used to assess the relative contribution of a wide range of marine ecosystem components in the supply of ecosystem services in the Latvian marine waters. Results indicate that the tool can be used to assess the impacts of environmental degradation in terms of ecosystem service supply and subsequent valuation in socioeconomic terms.

Teixeira et al. (2019) applied expert evaluation to identify relevant flow linkages from biodiversity to ecosystem services supply for eight case studies across European aquatic ecosystems covering freshwater, transitional, coastal and marine waters realms. They used ecosystem components such as habitats and biota as proxies for biodiversity and as the focal point for linkage identification. In agreement with the finding of Culhane et al. (2018), statistical analysis revealed the importance of considering mobile biota in the spatial assessment of habitats.

The aim to combine socioeconomic demands with biodiversity conservation is not yet standard practice in spatial planning. Therefore, Van der Biest et al. (2020) recommend to establish more ecologically sensible objectives that include ecosystem processes, besides species and habitats, as a more effective way of environmental conservation. They elaborated on the development of a method that integrates two key principles to advance the integration of ecosystem services with biodiversity conservation in planning practice: (1) consider the variety of ecosystem processes, biotic as well as abiotic, that support biodiversity and ecosystem services, and (2) link the ecosystem processes to biodiversity and to socioeconomic benefits to identify the common ground between seemingly conflicting objectives. They developed a methodology for a stepwise approach which is applied in a case study for the Belgian North Sea, based on an extensive review of available knowledge on ecosystem functioning, expert consultation and stakeholder involvement. They demonstrate how including ecological and anthropogenic processes opens opportunities to align biodiversity and ecosystem services and how this increases chances to provide long-term benefits for biodiversity and human well-being.

Von Thenen et al. (2020) argued that the ecosystem cascade can be used to structure the stock-taking and future scenario analysis in marine spatial planning (MSP). They applied a consistent approach to sorting indicators for measuring ES into the cascade. The indicators can be filtered based on the cascade steps, several quality criteria, and themes and linked to ES sections (provisioning services, regulating services, cultural services). The indicator pool allows to assess (1) the ecosystem components generating the services; and (2) the impacts on ES and their beneficiaries when changes occur in the provision of the services due to planning or management decisions.

5.6.6 Summary

For the marine ecosystems the SEEA EA accounts are gradually developing and have even more data problems than the terrestrial ecosystem accounts. Even more than in terrestrial ecosystems the quantitative causal relationships between biodiversity, ecosystem services and ecosystem services are poorly understood. To understand this relationship ecosystem processes and their relationships with ecosystem services and biodiversity needs to be included. Moreover, the extent to which there are interfaces between biodiversity conservation and the provision of ecosystem services needs to be explored.

5.7 Conclusion

Basically, SEEA EA is a systematic manner to register the changes in the ecosystems and to describe ecosystem services delivery. For the last part, these are directly related to the SNA. However, the relation with driving forces requires extra steps to be taken. The ecosystem quality may be characterised by pressure factors such as climate, water quality and air quality, and other characteristics of the ecosystem, of which biodiversity is one. In the end, the purpose of this accounting is to relate the changes in the ecosystems and biodiversity to driving forces, i.e., developments in the socioeconomic system. This requires explicit steps between socioeconomic activities through emissions and resource use to pressure factors and then to ecosystems. In some tables, it is suggested to make explicit what the cause of changes is by at least making a difference between causes by humans and nature, and by normal changes and catastrophic changes. However, this type of accounting requires much further development.

In biodiversity accounting, changes in biodiversity are related to pressure factors to predict the levels of biodiversity at each point in space. The quality of these derivations strongly depends on the quality of the models. To validate and calibrate the models, one needs observations, and this relates to discussions on cost-effective measurement of biodiversity. However, to relate pressure factors to biodiversity changes, one needs good measurement of the pressure factors as well. Pressure factors may be modelled, but these models must be calibrated and validated, where spatial differences may have consequences for the development of biodiversity.

To take it a step further, one makes economic drivers specific with respect to space. At the moment, this task is almost impossible to accomplish, as the biodiversity assessments approaches presented in Chapter 4 are not able to do this (except for rudimentary attempts for land use through a land allocation model such as CLUE or IMAGE). Perhaps there will be more opportunities in the future with new techniques, such as with remote sensing that creates more opportunities to get spatially explicit emission data. However, it may be difficult to actually acquire information on production activities in a location specific manner, because data are protected by privacy laws for statistical data. Therefore, it is not easy to get spatial data on location of production activities, even though the data exist. This implies that the sources of emissions will remain unclear, and that averages must be used in some way or another.

To make the accounting system policy relevant, one has to add causal relationships to the system. For example, in general equilibrium modelling, input-output tables are used to calculate the whole system in a systematic manner, where instead of the fixed average parameters, variable parameters for marginal changes may be used. For environmental consequences, one may calculate for example the greenhouse gas effects of changes in the economic system by using the average emissions per production type, where these average parameters can be calculated based on the UN study (UN et al. 2014b). However, in this case it may be that these parameters are not fixed and that the marginal and average parameters are not equal, for example because new technologies give less emissions than the average technology used.

A next step in making causal investigations is to investigate the spreading of for example nitrogen emissions over space into an ecosystem condition factor. Data from the accounting system may be useful to estimate or validate such a relationship, but they are far from sufficient. The data on for example acidification in the accounting system may be related to biodiversity effects and may be used to understand the dynamics of the system. Biodiversity is an indicator that may be used for this. In the accounting framework, it is tried to make a distinction between changes in biodiversity or pressure factors that are caused by normal human activities, those that are the consequence of normal natural processes and those that are caused by disasters such as fires, which may be caused by human and natural causes. And all these causes of real changes must be distinguished from changes caused by new information and other accounting reasons. Such accounting requires a type of modelling, and therefore will not be easy to accomplish.

The advantage of SEEA including experimental ecosystem and biodiversity accounting is its consistency with the national accounts. SNA is focused on economic transactions and is helpful to analyse input-output relationships and their consequences for the functioning of the system. By relating emissions and resource use to economic activities, it is possible to relate changes in resource use to changes in the economic system. And by relating emissions and resource use to ecosystem extent and condition and relating these ecosystem extent and condition to the delivery of ecosystem services, one may derive insights in the indirect effects of the production system through the natural environment. By valuing these ecosystem services, one may make explicit what the effect of nature (with biodiversity as a key component) on the economic system is and one may include direct deliveries of ecosystem services by nature, as well as disservices, and aggregating them in ecosystem asset values one may compare changes in natural assets with changes in produced assets and may correct national income and other aggregates for this. By ecosystem accounting, one may get better insights into the real value creation of the social-economic system.

However, bear in mind that the biodiversity accounting approaches have a number of shortcomings, partly the same as for the national accounts. First, only market transactions are registered and it is not meant as a method to calculate and register welfare. Second, if you derive indicators based on the system per unit of output, these are average parameters. To the extent that the system is used to investigate the effects of different scenarios and policies on the environment, one has to be aware that the national accounting system and thus SEEA includes only average relationships. Therefore, it is a consistent system to account for economic and environmental flows, but it does not tell directly what the change in flows will be if the system is changed. In the last case, parameters for marginal flows are needed, while the SEEA has average flows. Third, there may be an aggregation problem. When using the system for analysing scenarios and policies, there is a problem that sectors and commodities are aggregated in the SNA and that this may result in incorrect allocation of inputs, regions, emissions and natural resource use when an individual product, sector or company is analysed. This is a structural problem in the SNA.

The ecosystem accounting tries to monetise environmental assets. Valuation has ethical and cultural considerations, and one may see the valuation in monetary terms as inappropriate or potentially misleading (UN et al. 2014a, p.105). Furthermore, in the literature, there is criticism on commoditising ecosystems and nature (Obst et al. 2016; Kosoy and Corbera 2010; Soma 2006). When monetising, one must be very aware of what the purpose of monetisation is and be very careful to what extent the monetisation gives the right information, in particular if it is used to inform policy decisions (UN et al. 2014a, p.105).

Valuation of ecosystem services is different from other valuation approaches. With respect to the monetary accounts, valuation is based on market prices. This implies that the non-monetary values of flows are not included while these can be very relevant. Externalities will normally not be monetised if there are no market prices, but in principle it is possible to give a monetary value to it as much as possible according to transaction prices if there would be a market. This is, however, fundamentally different from valuing externalities according to damage costs, as is done in cost-benefit analysis.

As for produced assets, valuation of ecosystem assets is difficult. Ecosystem assets are valued based on the net present value of expected returns. This implies that future benefits are valued less than current benefits and this implies a value judgement. Furthermore, as far as biodiversity accounting allocates ecosystem benefits to sectors based on value added, this implies that the current marginal benefits of the ecosystem services are included, while consequences for future scarcity are not included.

6 Overview of biodiversity accounting approaches in practice

6.1 Introduction

This chapter provides a short overview of approaches in biodiversity accounting for business. There have been attempts of biodiversity accounting by companies themselves. Section 6.2 provides an overview of these examples by companies. In Section 6.3, the Product Biodiversity Footprint is introduced, which is an LCA-based approach which explicitly includes biodiversity. In Section 6.4 a number of approaches of measuring biodiversity for assessment are introduced, which might be applied by companies as well. In Section 6.5 biodiversity for companies in the financial sector is discussed. Finally, Section 6.6 concludes.

6.2 Biodiversity accounting for business

Lammerant et al. (2019) provided an overview of biodiversity accounting approaches for business. They concluded that generally accepted methodologies are currently lacking. Accounting approaches suitable for business must be practical and pragmatic as well as meaningful and relevant. Criteria were relevance, rigor, replicability and consistency, while additionally sector coverage, user friendliness and information needs may be relevant as criteria. They mentioned another list of criteria: meaningful, measurable and comparable, possible to aggregate, practical, replicable and credible, context-based and responsive. The goal of the study of Lammerant et al. (2019) was to provide an overview and comparison of methodological approaches and an overview of obstacles and gaps that still exist.

The study provides a table with an overview of different studies by characterising them by the organisational focus areas including corporate applications and purposes such as assessing biodiversity risks and opportunities, comparing options, going for no net loss or biodiversity net gain perspective, or internal and external communication. Table 6.1 presents the biodiversity measurement approaches evaluated by Lammerant et al. (2019).

Table 6.1 *Biodiversity measurement approaches for companies with their scope and their use*

Biodiversity measurement approach	Acronym	Scope a)			Corporate use b)	
		Habitat/species	Ecosystem services	Genes	Current	Future
Biodiversity Footprint Financial Institutions	BFFI	V			2	
Biodiversity Indicators for Site-based Impacts	BISI	V			1	
Biodiversity Impact Metric	BIM	V			2	1
Global Biodiversity Score	GBS	V			3	1
LIFE Methodology	LIFE	V			2	2
Product Biodiversity Footprint	PBF	V	X			
Species Threat Abatement and Restoration metric	STAR	V			2	2
Biodiversity Footprint Methodology and Calculator	BFMC	V			3	
Corporate Biodiversity Footprint	CBF	V			2	1
Biodiversity Net Gain Calculator	BNGF	V				
BIRS and ES assessment	BIRS	V	V			
ReCiPe2016	ReCiPe	V				
Agrobiodiversity Index	ABDi	V	X		3	
Biological Diversity Protocol	BDP	V			2	2
Biodiversity Performance Tool for Food sector	BPT	V				
Biodiversity Monitoring System for the Food Sector	BMS	V			2	
Environmental Profit & Loss	EPL	V	V		3	3

a) V = quantitative approach, and X= qualitative approach; b) 1 = potential, 2=remote and 3 = matured approach for corporate applications of current and future assessment of biodiversity.

None of the approaches include genes in their assessment, most approaches are only focused on habitat or species.

In the case of corporate use in current circumstances, there are three approaches which have fully matured: BFMC, ABDi and EPL. Only the latter approach can be used to assess future circumstances of biodiversity.

Barker (2019) focused on the reasons for corporations to use natural capital accounting. One reason is just to fulfilment of current or expected legal requirements, another one is to create shareholder value by being better perceived by consumers, NGOs and others, which may imply greenwashing. This can be approached by the lens of legitimacy theory and the idea that firms need a license to operate. A third reason may be conventional, where institutional rules, beliefs and norms developed within an organisation through peer group socialisation create socially constructed behavioural constraints. Friedman (1970) referred already in the 1970s to the fact that firms are guided by norms embodied in ethical custom. These norms may stay for a long time and then have a sudden disruption. Since 2019, most big companies integrate financial and non-financial data in their financial reports, but every company does this in its own way.

Financial accounting has standards that provide relatively reliable, comparable, relevant and complete information in a standardised manner. By contrast, the natural capital protocol is more a take-your-pick document than an accounting standard, where measurement is only partial and a lot of the valuation of assets is based on the future.

Financial accounting is mainly meant for shareholders of companies, while natural capital accounting may serve a broader audience. There is a distinction between accounting and reporting, accounting is based upon registered observations and thus relatively certain and focused on the past, whereas reporting involves the future. A purpose of accounting is to show the vulnerability of corporations for adverse environmental impacts, either directly or indirectly, for example through a risk of increased regulation and taxation being imposed or changes in consumer behaviour.

6.3 Product Biodiversity Footprint

An interesting example of how LCA methods are used in practice by businesses is the Product Biodiversity Footprint (PBF) approach. It combines state-of-the-art knowledge on LCA and ecology to better inform businesses on biodiversity. The PBF is the results of public-private research and development partnership initiated in 2017 by I-Care & Consult and co-developed by Sayari. This research is funded by the French Environmental Protection Agency (ADEME), and three private companies: L'Oreal, Groupe Avril and Keering.

The PBF method consists of three modules. The first module computes the LCIA of a product using LC-IMPACT (Verones et al. 2020). It has 7 impact categories: land (stress) occupation, land (stress) transformation, water stress, photochemical ozone formation, terrestrial acidification, freshwater eutrophication and climate change. Except for the climate change impact category, which is a global impact, these impact categories can account for spatialisation.

For impact categories related to habitat change (land stress occupation, land stress transformation, and water stress), LC-IMPACT characterisation factors (CFs) account for the vulnerability of species in the native areas (ecoregions or watersheds) based on Chaudhary et al. (2015). To calculate the impact on biodiversity at global level, native resolution CFs are aggregated at the country level, including vulnerability scores from 0 to 1.

Impact categories related to pollution (photochemical ozone, terrestrial acidification and freshwater eutrophication) and climate change are assessed with a 'regional perspective' (i.e., the perspective of the native area to one species lost). Based on this, the impact on biodiversity is calculated at the regional level.

In LCA methods, the concrete impacts from different production practices on biodiversity in a specific area are not taken into account. For this reason, in the second module of the PBF method the LCA results are refined using specific information on practices. Chaudhary and Brooks (2018) present first attempts to address different land management intensity levels (minimal, light and intensive). This however raises practical questions on how to relate a given practice (e.g., type of tillage) to an intensity level. In LCIA models, generally six land categories are considered: primary forest, secondary forest, permanent crops, annual crops, grassland and urban land. Within each category, wide variability in management practices exist, which is not captured in LCA. PBF uses ecological data to differentiate between practices impacting biodiversity, which is used in adjusting the CFs for land occupation and land transformation. For example, to evaluate the impact of palm oil cultivation on the environment, non-certified cultivation can be compared with a certified one, i.e., palm oil that was certified by the Roundtable on Sustainable Palm Oil (RSPO). To assess the impact, a so-called Practice Adjustment Coefficient (PAC) is computed, based on quantified ecological data and literature, linking biodiversity richness and type of land occupation or transformation.

LCA methods do not cover the impact of species overexploitation and the presence of invasive species in biodiversity. Therefore, in the third module, two semi-quantitative indicators are provided to assess invasive species and overexploitation of species, as no recognised methodology exists for quantifying those aspects. The Species Management indicator is used for assessing overexploitation and the Invasive Alien Species indicator for assessing impact of invasive species. The actions undertaken to diminish the impact or risk of impact are taken into account. Both the level of impact and level of action is given a score and is multiplied to provide an overall score.

In summary, the main output of the PBF is the LCA results for seven impact categories and the results from the two semi-quantitatively assessed indicators. Therefore, PBF is just a type of extended LCA, without integrating them in one footprint category such as land use or biodiversity loss.

6.4 Weighted biodiversity metrics

The most common approaches in biodiversity impact assessment are unweighted indicators of intactness of original species composition such as MSA or PDF. However, not every natural ecosystem might have the same value in decision-making. Therefore, practical methods have been developed that put a weight on each hectare to reflect the importance of biodiversity in that area. In this section, three approaches for weighted biodiversity metrics are discussed: Biodiversity Metric 2.0, the Nature Points Methodology and Healthy Ecosystem Metric Framework. These are fairly straightforward metrics that aim to determine the change in biodiversity after an intervention. The metrics could be used to assess how much compensation is needed, for instance.

6.4.1 Biodiversity Metric 2.0

Biodiversity Metric 2.0 (Crosher et al. 2019) is a tool developed in the UK to evaluate changes in biodiversity by changes in habitat to measure the biodiversity benefits of land management, to understand how temporary works may impact biodiversity and to calculate how much compensation is needed when habitats are permanently lost. The tool is based on points, called biodiversity units, and consists of multiplications of factors. The planning philosophy behind the metric is straightforward sequence of steps:
- first one tries to avoid changes,
- second one tries to minimise the damage, and
- third one tries to remediate and in last instance to compensate.

Furthermore, the idea is that if biodiversity is destroyed it should be common practice not to compensate this with biodiversity of lower quality.

The starting point of the metric is the habitat area. The total area is defined by areas of different habitats, such as grassland or modified grassland, and each type of habitat has a distinctiveness score. Second a score is determined for condition on a 5-point scale, which is described in a technical document. Then a score is made for the strategic significance based on the status of the area as being not significant, relevant, and formally relevant for nature. Finally, a score for connectivity tells the proximity of the habitat to similar areas of habitats. The multiplication of all numbers is the end score.

The same calculation can be repeated for the post-intervention situation, and the change in biodiversity units is the metric of the change in biodiversity. However, when things change the endpoint is not certain. Therefore, risk factors are multiplied with the change: difficulty of creating or restoring habitat, the time needed to establish the new habitat type, and if as a compensation elsewhere biodiversity measures are accomplished, there may be risk related with that. Because for offsite compensation the same metric can be used, the tool is useful for post-measurement effects. Figure 6.1 summarises the steps involved with a numerical example.

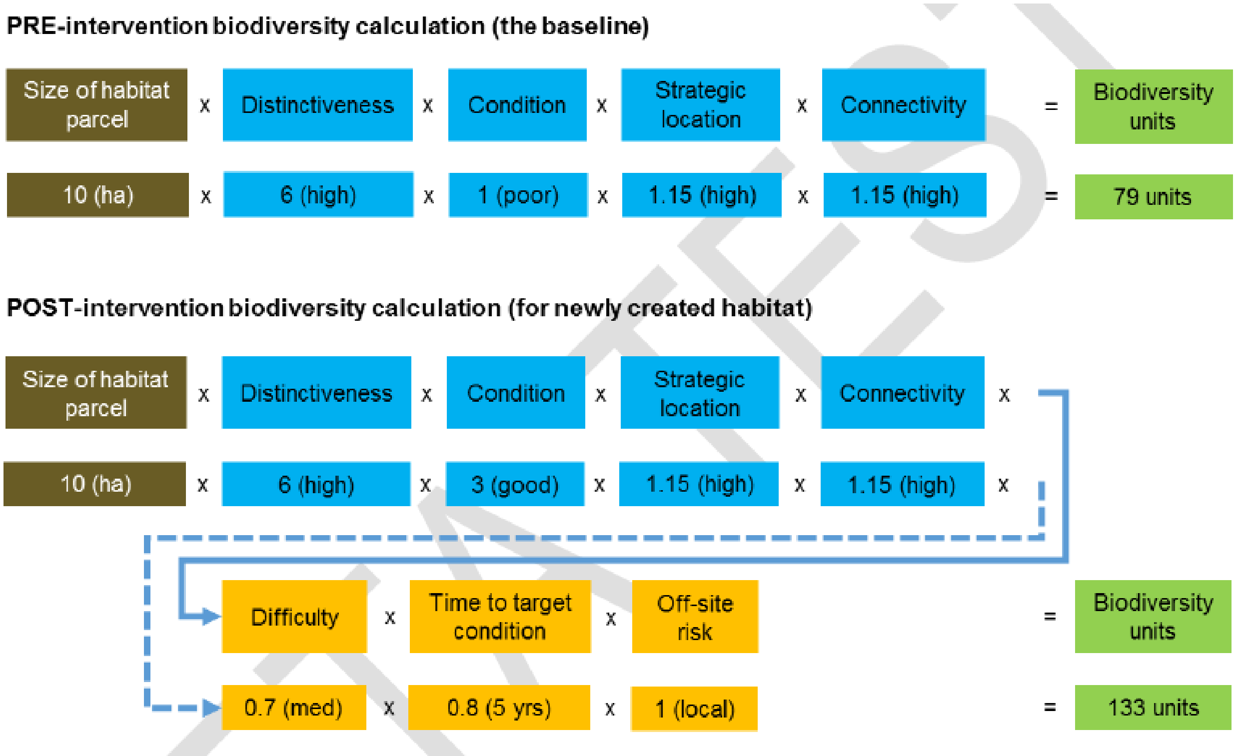

Figure 6.1 *Calculation scheme Metric 2.0*
Source: Crosher et al. (2019), Box 2-2.

The change is between pre- and postintervention can be differentiated into area lost, area retained, area enhanced, and new areas created.

With respect to data, one may start with readily available information, such as a habitat inventory and Sites of Special Scientific Interests (SSSI) boundary information (MAGIC database in the UK) and search for information on species distributions (NBN atlas in the UK). Then it is wise to have a walkover in the area to get insights in the really existing parcels of habitats. A habitat classification system is needed to help to categorise the habitats and the surveyor must evaluate the quality of the habitats per parcel. For linear habitats such as hedgerows, lines of trees such as in urban areas, rivers and streams, separate calculation schemes are available, because it is difficult to compare these in points.

Biodiversity metric 2.0 is meant to help decision-making with respect to biodiversity, but it is obvious that additional information is needed, for example on protected and local species and the question to what extent irreplaceable habitats may be destroyed.

From an economic point of view, one may estimate the price of biodiversity changes by calculating the cost of compensatory measures, using the pointing scheme. This gives roughly a cost per biodiversity-unit, although one must be aware that by far not everything is included, and that the compensation measures are location specific. Moreover, prices should perhaps be location and habitat type specific. In such a way changes in biodiversity that legally do not require compensation can be included in the business accounting or project accounting framework.

In summary, the method consists of a number of weights that are semi-quantitative, and indicator based, and therefore the scientific foundation is loose. Furthermore, it requires much local information, and therefore the method is not suitable for calculations in complete value chains.

6.4.2 Nature Points Methodology

The nature points methodology has been developed by PBL Netherlands Environmental Assessment Agency in 2009 (Sijtsma et al. 2009; Gaalen et al. 2014). This methodology has been developed specifically for the

Dutch situation. The calculation of nature points uses the typology of the so-called 'Nature target types' from the Manual for Nature target types (Bal et al. 2001). All nature within the area of influence of an intervention (project or policy) is subdivided into these nature target types, to which reference lists of characteristic species of this nature type are linked. On this basis, the following calculation principle is used (Jaspers et al. 2016):

*Nature points = quality * weighting factor * surface area*

Hereby, the factors are determined as follows (Jaspers et al. 2016; Arcadis and CE Delft 2018):
- Quality: percentage of the number of reference list species present in the current situation or expected in the future situation, on average over at least 3 characteristic species groups
- Weighing factor: factor based on, or derived from, the PBL scale for weighing of nature types based on (inter)national rarity and trend of species. It is about the (relative) threat to the ecosystem or natural type. Endangered systems have a higher weight than common and non-threatened systems. The weighing factor is determined (or searched for) specifically for the types of nature present (zero alternative) or expected (policy alternatives) types of nature. The weighting factor has been determined on the basis of (inter)national rarity and trend of the species characteristic of the nature type. The weighing factor is determined beforehand on the basis of systematic, objective information and expert knowledge.
- Surface area: measured or calculated area of the nature type unit in hectares.

The total number of nature points per intervention is the sum of the nature points of the different nature types in the present or future situation. This enables a comparison of the biodiversity between the reference situation and policy alternatives.

The following six steps are used to calculate the nature points (Jaspers et al. 2016; Arcadis & CE Delft 2018):
1. Selection of nature types (reference and expected situation in the policy alternatives).
2. Determination of the areas per nature type (reference and expected situation in the policy alternatives).
3. Selection of species groups per nature type. At least three species groups are selected from the species lists. Typically, plants are a good basis for nature quality, and birds are a good measure of disturbance. The third group of species can be defined specifically for the nature type.
4. Determination of the observed species per nature type in the reference situation. This can be done on the basis of for example the National Database Flora and Fauna (NDFF) and field inventories.
5. Determination of the expected species per nature type in the policy alternatives. This can be done on the basis of expert judgement on the basis of four suitability criteria (site conditions, minimum area, accessibility, disturbance).
6. Calculate the nature points (reference and the policy alternatives) through the following sub-steps:
   a. calculation of the fraction of the number of observed or expected species in relation to the total number of species in the reference list by nature type;
   b. calculation of the arithmetic mean quality score over the 3 species groups per nature type;
   c. calculation of the actual nature points; and
   d. summing up the nature points for the nature types in the current and future situation.

It should be noted that the nature points methodology requires detailed ecological knowledge and expert judgment about what species groups are relevant and what can be expected in the policy alternatives. Moreover, the importance of monitoring species should be stressed, as species observations are crucial to assess the current situation.

The calculation of nature points is based on species richness, which is the number of different species represented within a particular surface area. The number of species is counted and compared to the expected number of species; a species can either be present or not. The methodology does not take into account the abundance of the species, and in that respect, it is similar to the PDF (as opposed to MSA, the mean species abundance).

It Is noteworthy that the nature point methodology has not been drawn up with the valuation of nature and biodiversity in mind. Nature points measure changes in the quality and quantity of biodiversity and not welfare effects. As De Blaeij and Verburg (2011) pointed out, there is no direct link between nature points on the one hand and ecosystem services and people's preferences on the other hand. To connect nature points to welfare, it is necessary to know how people value nature points, and how people value changes in nature types or ecosystems.

However, as many ecosystem services depend on biodiversity, either directly or indirectly, it makes sense to assess to what extent ecosystem services are affected by changes in nature points. Although these changes often cannot be quantified directly with the current state of knowledge, it may be possible to make a statement at a qualitative level about the welfare effects resulting from changes in biodiversity (Arcadis and CE Delft 2018).

The nature point methodology is fairly similar to the Biodiversity Metric 2.0 methodology described in Section 6.3. However, Nature points measures the quality based on a selection of species, while Biodiversity Metric 2.0 has a broader interpretation of habitat quality. To measure the biodiversity benefits of land management, the latter evaluates changes in habitat, whereas the nature point methodology assesses changes in species. Both methodologies are in fact multiplications of factors that result in an overall score; this score is either called biodiversity units (Biodiversity Metric 2.0) or nature points. In Biodiversity Metric 2.0, there are more factors though: habitat type (distinctiveness score), habitat condition, strategic significance, and a score for connectivity.

6.4.3 Healthy Ecosystem Metric Framework

The University of Cambridge Institute for Sustainability Leadership CISL) developed a metric to evaluate the impacts of business for biodiversity and to improve their decision-making (CISL et al. 2016; Di Fonzo and Cranston 2017; CISL 2020). Their approach is inspired by the Environmental Profit and Loss (EP&L) methodology developed by Kering (Danish Environmental Protection Agency 2014). According to the Natural Capital Coalition, the Healthy Ecosystem Metric will help companies with the following:
- Measure: the impact from sourcing raw materials on biodiversity, soil and water in global supply chains
- Identify: high-risk locations where a company is most likely to experience biodiversity, soil and water risks or create negative impacts
- Drive: the collection of credible impact data that companies can use to develop targets or KPIs
- Inform: response strategies to safeguard natural capital and drive improved business performance

The metric must be meaningful in that it can drive decision-making and the methodology can be clearly understood, it must be measurable and comparable over time and space, it must be easy to aggregate, must be practical in the sense that required data are easily accessible and that when improved information comes available it can be easily brought in, it must be replicable and credible, i.e. based on scientific methods, it must be context based in considering local conditions, and it must be responsive to changes and improvements in company activities, both in the long and short term.

To make it easy to aggregate land area influenced is the basic unit. The basic equation is:

impact = land area x impact on biodiversity x impact on soil x impact on water.

The logic of the further development is very simple and consists of five steps. First, it starts defining land use types, and then uses land impact coefficients from peer-reviewed studies. Second, the companies' supply chain location is needed, and ecosystem context datasets are used to get biodiversity values on quality. The calculation scheme is as follows. First, the amount of raw material is determined and as far as possible the location of land where it is produced. Second, the yield for the expected location of land use is taken from databases (from FAO or other sources), and this determines the number of hectares. Third, the land use type is described and the land use intensity and this combined determines through a look up table the percentage loss in biodiversity. Fourth, the impact on quality is determined based on context information. Fifth, the outcome is a multiplication of everything, which is the loss in biodiversity in hectare equivalents.

Companies may influence their biodiversity impact by changing land area needed, changing land use type and changing ecosystem context by changing the location. The framework is summarised in Table 6.2.

Table 6.2 *Healthy Ecosystem metric framework summary*

Impact on biodiversity (ha-eq.)	**Land area**	**Impact on biodiversity**		
	Area required for supply chain/ operations	Quantity of biodiversity		Quality of biodiversity
Measurement	Hectares (ha= tons/yield)	Land use type	Land use intensity	Ecosystem context
Data sources	Company data (or external data if unknown) - Tons purchased - Yield	Company data (or external data if unknown) - Land use type - Land use intensity		Company data (or external data if unknown)
		External data Coefficients relating land use type and intensity to impact on biodiversity		External data Conversion factor to account for the importance of a location for biodiversity (e.g., a location's biodiversity richness or rarity)

Source: CISL et al. (2016).

For measuring biodiversity, the Biodiversity Intactness Index is used, where it is argued that the MSA is not suitable because it is less practical, but especially because it is not responsive to changes in business activity (CISL et al. 2016, Appendix D). However, the logic of MSA and the Biodiversity Intactness Index (BII) is more or less the same, with different reference points. The II can just be seen as an alternative to MSA and PDF. Just as the Healthy Biodiversity Metric and the Nature Points, the Healthy Ecosystem Metric results in some standardised hectares change by weighting the different ecosystems.

6.5 Biodiversity accounting for financial institutions

6.5.1 Introduction

ASN and CDC Biodiversité developed a methodology for biodiversity accounting. A study compares the two approaches and searches for common grounds (Berger et al. 2018). The purpose of biodiversity accounting for financial institutions is to manage investment risks and to develop broader strategies around SDGs and other strategies of the banks. Biodiversity indicators may be used to engage in the policies of firms on this topic and to be selective in the investment portfolio (Berger et al. 2018).

ASN has as a policy to reach a net positive biodiversity impact in 2030 (ASN 2023). Net biodiversity gain can be reached through averted loss and/or degradation of biodiversity or improving the protection status through positive management actions (restoration, enhancement). Therefore, the definition is extremely broad and uses a baseline as a reference situation (CREM and Pré Sustainability 2019). This is in line with the Principles for Positive Impact Finance, the Platform Carbon Accounting Financials and the IRIS+ methodology of the Global Impact Investing Network. Investment in enhancement of existing biodiversity or restoration of biodiversity obviously has a direct positive effect on biodiversity. However, most investments have more indirect effects and have only positive effects compared with a baseline. Examples are investments that address one or more drivers of biodiversity loss, investments in the production of energy resources that replace energy resources with a higher impact on biodiversity, investments in alternative livelihoods that generate less unsustainable resource extraction and biodiversity loss than the original livelihoods, and investments to avert known future risks to biodiversity. The use of investment criteria and stimulating companies to fulfil certain criteria, such as those of the FRC (Financial Reporting Council), work even more indirect. For transparency in biodiversity reporting, net biodiversity impacts in actual increases in biodiversity, reduction of the negative impact and avoidance of a negative impact on biodiversity need to be distinguished.

The method used for the ASN biodiversity footprint uses the Exiobase database to get the LCI and the ReCiPe2016 method to calculate the biodiversity effects. The positive effects of wind energy are caused by replacing the average current energy, implying that when the current energy mix becomes more sustainable, the benefit of investing in wind and solar energy is reduced. However, direct biodiversity impacts of wind energy is not considered. This is meant as an artefact of the method (Pre Sustainability and CREM 2019), but it seems a fundamental issue.

There are other issues with the methodology used. First, it uses averages as a comparison (such as the average energy mix), while the marginal effects are relevant. Second, a positive biodiversity footprint does not mean that the investment portfolio is positive for biodiversity, but only that it is less negative for biodiversity than an average portfolio. It assumes that investment is determined by the financing decisions of companies. On the one hand, if a bank just invests in a project that would otherwise be financed by another bank, then the bank does not have a replacement effect. On the other hand, if the bank would invest in activities with high biodiversity losses, this obviously stimulates negative tendencies.

Third, there is an issue with portfolio choice. It is suggested to improve biodiversity performance by changing the investment portfolio. However, to what extent is this a solution for a societal problem and not just window dressing? For example, if building activities would be reduced, but society still needs these activities, someone else has to finance them. This is roughly the same substitution problem.

Fourth, the reference situation is unclear. To prevent picking out the activities with a net positive impact, leaving the bad ones for other banks, one must correct for the fact that some things have to be produced. For example, there is no option to stop producing food altogether, so one must compare producing food in the current situation with alternative ways of consuming and producing food. The question is to what extent the financing decision influences the outcome. Perhaps not financing unsustainable practices is a good starting point. However, for picking the ones that are consistent with a sustainable transition, the choice is much more difficult.

The assessment of biodiversity for companies in the financial sector has a number of aspects to consider. First, there is the choice of biodiversity indicators (Section 6.5.2). Second, there are a number of steps in the analysis to consider (Section 6.5.3).

6.5.2 Biodiversity indicators

Biodiversity is important both as a part of the life support system and because of its intrinsic value. One must not only focus on rare species or biodiversity services, but on the intactness of biodiversity in general. Biodiversity is important for the resilience of ecosystems and the firms that depend on these ecosystem services, while biodiversity has an option value for future services.

The banks use respectively the MSA and PDF as metrics, where both tell roughly the same story, as has been already discussed in Chapter 2. Both have a time and a space dimension, implying that the size of the metric grows proportionally with both space involved and duration of the disturbance. The idea is that if a land area is disturbed, it may be restored after use, and for example greenhouse gasses do not stay in the air forever. For example, CH_4 becomes CO_2 with less harm after some decades. This type of mechanism must be taken into account in the analysis.

6.5.3 Steps in the analysis

Berger et al. (2018) distinguish six steps in the biodiversity footprint approach for financial institutions:
1. Analysis of the focus of investments
2. Assessment of the pressures generated by the activities invested in
3. Assessment of the impact of the pressures on biodiversity through pressure-impact relations. Both GLOBIO and ReCiPe have pressure-impact relations
4. Interpretation of the results
5. Comparison with reference and
6. Action, including influencing policy.

Figure 6.2 summarises the approach.

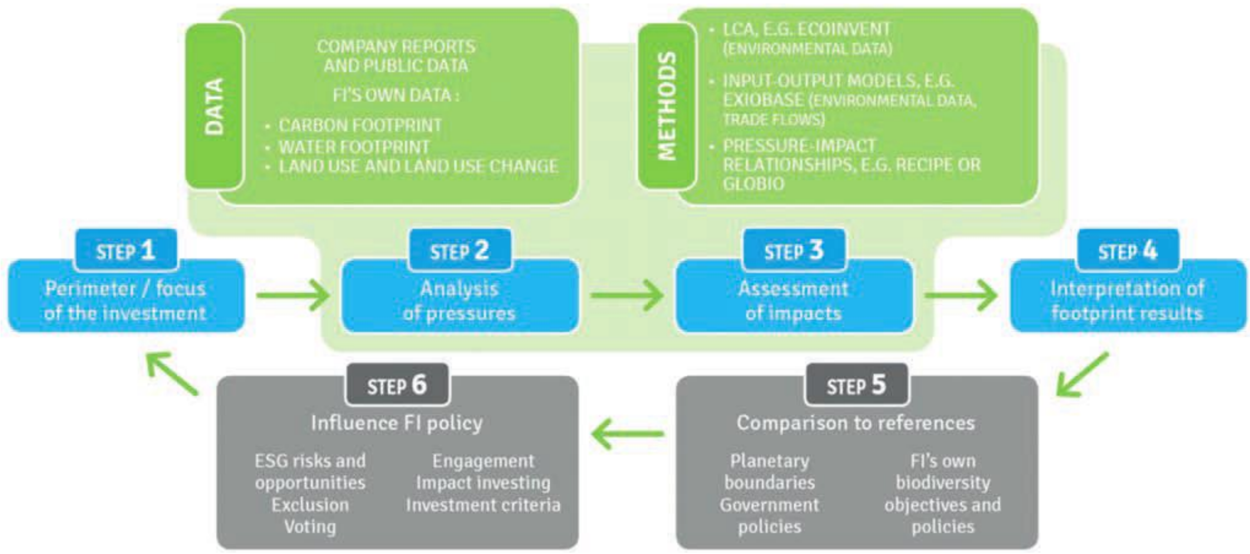

Figure 6.2 *Six step for biodiversity accounting approach*
Source: Berger et al. (2018, p. 17).

A number of requirements and desired characteristics of the approach have been developed:

- Requirements
  Relevance, responsive to change, transparency, fit for purpose (data should be compatible and consistent with objective), compatibility, consistent over time and between datasets and robust.
- Desired characteristics
  Biodiversity as a whole, cross-sectoral, global and including the whole value chain. In addition, consensus, i.e., data sets and methodologies public and peer reviewed or broadly accepted, and a wide community of stakeholders is involved in reviewing.
- Relevance
  The biodiversity footprint should include the most important pressures: habitat change, overexploitation, invasive alien species, pollution and climate change, based on Millennium Ecosystem Assessment (2005).
- Responsiveness to change
  Problem is that extended input-output tables such as Exiobase have an aggregation level that is too high, implying that averages determine the outcomes. To a lesser extent, this is the case with LCA databases such as Ecoinvent. Collecting data from the source is expensive. Depending on importance and easiness of collecting data, one may determine the level of detail.
- Transparency
  With respect to carbon footprints, ACTIAM publishes its methodology explicitly online, and describes explicitly scope, measures, data, calculations and assumptions (attribution rules) of its footprints.

6.5.4 Scopes and attribution

In carbon accounting, three scopes are distinguished, see Figure 6.3:

- Directly influenced by the decision maker on the terrain of the decision maker,
- Influenced by the decision maker by buying inputs directly used by the decision maker (for example electricity), and
- Influenced by upstream and downstream decision-makers.

On top of that, a scope 0 impact can be distinguished for land use, i.e., by using land, it cannot regenerate towards its natural state, and this implies reduced greenhouse gas emissions and lower biodiversity than without the activity.

One may go even further than Berger et al. (2018) by extending the scope with indirect land use (Searchinger et al. 2018). One can reason that when land would not be used by the current activity, another activity could take place on the land, and the current activity would occupy land somewhere else, that may be in a high biodiversity area.

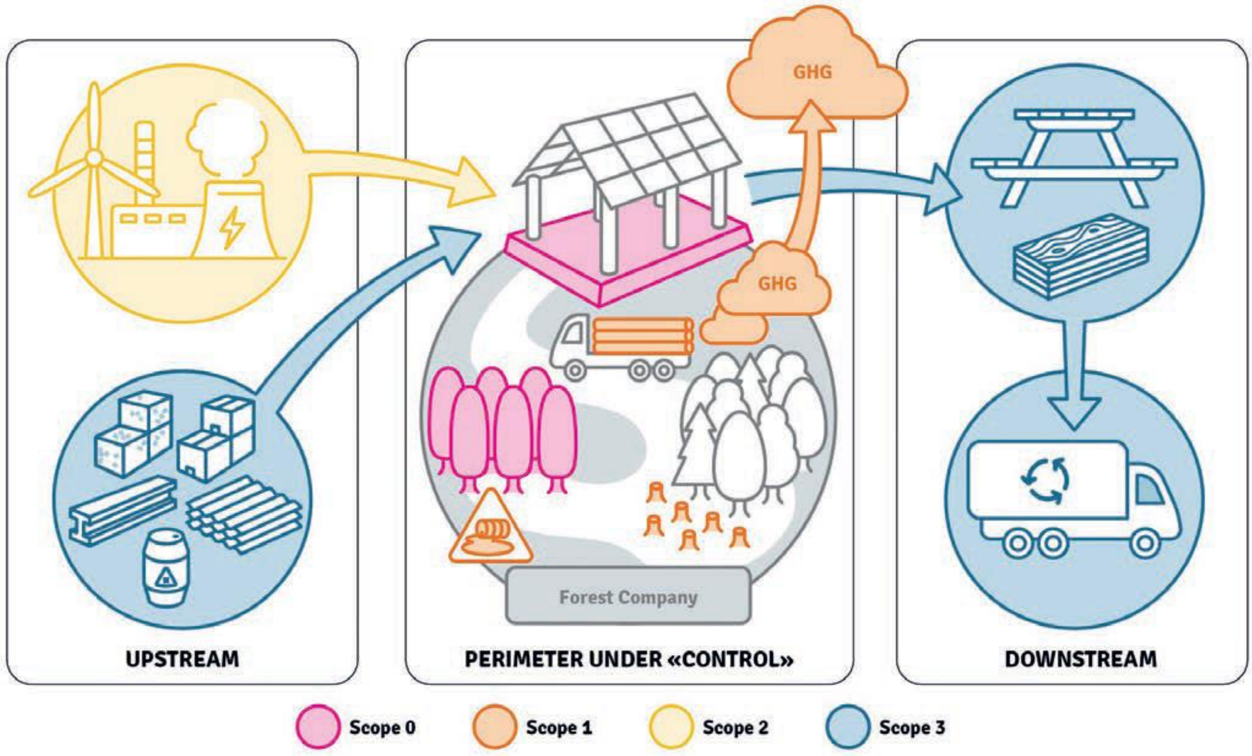

Figure 6.3 *Graphical presentation for the four scopes*
Source: Berger et al. (2018, p. 20).

Which part of the biodiversity impacts of different activities should be attributed to a firm or a financing institution. It depends on how much control such a decision maker has: with complete financial control for example by having a majority of shares or full operational responsibility, 100% of the biodiversity consequences should be included. With a minority of shares, it may be according to its shares, but what about loans from banks to companies, or banks managing money for a company? One may argue that even investments in state bonds make the bank co-responsible for biodiversity effects of state activities. The attribution of responsibility of a firm is a fundamental issue to be developed further.

Although not mentioned explicitly, the discussion on positive impacts (Berger et al. 2018, p. 27) suggests a comparison with an alternative scenario. Is the alternative the case where the product would not be made, or is it the scenario where the product would be produced according to traditional methods? In the first case the effect mostly will be negative, while in the second case all measures taken will have a positive influence.

6.6 Conclusion

Stopping biodiversity loss requires that companies take biodiversity values into account in their decision-making. A first step would be that they include these values in their accounting systems. Most big companies integrate financial and non-financial data in their financial reports. Lammerant et al. (2019) provide an overview of business approaches to biodiversity accounting, where it seems that for every purpose, many different methods exist, which implies that outcomes may be inconsistent. While in financial accounting, standards guarantee that relatively reliable, comparable, relevant and complete information is provided, such a standard does not exist for biodiversity accounting. In the current approaches, biodiversity indicators related with the quality of ecosystems, such as PDF and MSA, are most commonly used, but they do not measure all relevant aspects of biodiversity, such as the diversity in ecosystem types and extinction of species. Weighting of the indicators for biodiversity quality may include part of it, but there is no consensus on weighting systems.

Companies conduct biodiversity accounting for various reasons, such as legal requirements, reputation management or anticipating future regulations. These motivations do not guarantee that these accounts provide sufficient information to tackle the current biodiversity challenges.

Biodiversity accounting for financial institutions is not fundamentally different from biodiversity accounting for other businesses. However, two aspects are more important. First, financial institutions may be interested more than other companies in the financial risks of changes in natural capital, because this determines their risk portfolio. Second, the influence of financial institutions on decisions that are relevant for biodiversity is much more indirect than that for most other companies. This complicates allocating the responsibility of financial institutions for the biodiversity consequences of their investment portfolios. Third, whether or not investment decisions have a positive or neutral effect on biodiversity depends very much on the scenario it is compared to. To what extent is the decision to carry out specific activities determined by decisions made by financial institutions?

7 Valuation of biodiversity

7.1 Introduction

Since most decisions in a market economy are based on financial information, it is relevant to make the impact on biodiversity comparable to financial information from companies and other decision makers. Therefore, it is relevant to assign a value to biodiversity, so that its value can be compared to financial accounts or incorporated into decision-making. However, there is no consensus on both the idea of valuing biodiversity (people argue that it is impossible by definition) and the methodology for valuing biodiversity (biodiversity is a very heterogeneous concept). This chapter discusses a number of issues with respect to valuing biodiversity.

One approach is to derive a value for biodiversity from LCA analyses. This approach is, however, not very satisfactory because it is based on ecosystem services, which are only weakly related to biodiversity indicators (Section 7.2). In Section 7.3, the relationship between the value of ecosystem services and biodiversity is further investigated. This is related with the ecosystem service accounting approach discussed in Section 5.4. Since the current ecosystem services approach does not directly take into account biodiversity and ecosystem quality in practice, it is valuable to search for other methods. In compensation and offset systems for infrastructure investment, for instance, some implicit or explicit prices for biodiversity emerge. Therefore, Section 7.4 discusses the extent to which this can provide useful information for consistent monetary biodiversity accounting.

7.2 Valuation of biodiversity

If one would like to make effects on biodiversity comparable with financial accounts of firms, it can be useful to monetise these biodiversity effects, implying that one must have a price per unit of biodiversity (De Bruyn et al. 2018; Amadei et al. 2021). One approach is to value the biodiversity indicator used. As discussed earlier, PDF and MSA are the indicators that are used the most. For the EU, in the Handbook of Environmental Prices for the EU28 published in 2018, De Bruyn et al. (2018) presented values for biodiversity where the ecosystem quality indicator PDF is priced, based on literature studies that relates PDF in plots of land to the value of ecosystem services provided.

In other words, they determined a price per PDF/m^2/yr. De Bruyn et al. (2018, p. 73) presented three statistics to reflect the distribution of values of biodiversity in the EU: lower bound value, median value, and upper bound value, see Table 7.1. The upper bound value (€ 0.649) was based on the highest average value found in a meta-analysis study by Kuik et al. (2008), and the median value was the median value found in the same study. In their study, Kuik et al. (2008) used the value per unit of Ecosystem Damage Potential (EDP), which is roughly the same as PDF (Kuik 2007). The lower bound value (€ 0.024) based on restoration costs as calculated by Ott et al. (2005), which is used by NEEDS (2006). This approach is exemplary for many other methods such as the true pricing approach applied on agri-food products (Galgani et al. 2023; Galgani et al. 2021) and marginal damage costs for loss of ecosystem services from land-use change or ecosystem degradation (Lord 2020) based on the SPIQ-FS dataset of marginal costs by FoodSIVI of Oxford University. Both methods relate an indicator of biodiversity with the value of ecosystem services provided.

Table 7.1 *Values of biodiversity for the EU presented by De Bruyn (2018)*

Value	Value (PDF/m^2/yr	Source
Lower bound	0.024	Kuik et al. (2008)
Median	0.083	Kuik et al. (2008)
Upper bound	0.648	Ott et al. (2005)

The meta-analysis study of Kuik et al. (2008) on biodiversity values collected 160 valuation studies from the literature. Only 24 studies were suitable for the meta-analysis, which yielded 42 observations in terms of a value per PDF/ha/yr. The spread of study location was very broad, although 33 of the 42 observations are from either Europe or North America. With respect to the types of ecosystems, there were 16 observations for forests, 17 for rivers, 2 for coastal areas, and 7 for other ecosystems. There were differences in the valuation methodologies of the 42 observations: 18 studies used contingent valuation, 17 used choice experiments and 7 used other methods. As a result, the spread values of biodiversity found by Kuik et al. (2008) was very large as well. Therefore, the median value of biodiversity seems to be a better indicator to present than the average.

Kuik et al. (2008) explained the value per EDP per hectare by a number of factors using linear regression techniques. The factors included population density, the size of the change and the area of the total ecosystem in hectares, and differentiates with dummies for forest, river and coastal ecosystems.

However, values of biodiversity might change with the explanatory factors. For example, if the population density would increase from 100 to 400 people per hectare, then the value of the PDF would more or less double. The value of forests is about three times as high as of a river, while a coastal area is somewhere in between. If the size of the ecosystem from which land is transformed to land with less biodiversity is doubled, the value per PDF is reduced by 30% (own calculations). Detailed information on the regression analysis can be found in Kuik et al. (2018).

An alternative way of evaluation is to quantify the value of ecosystem services provided. However, Science for Environment Policy (2015) concludes that the relationship between biodiversity and ecosystem services is not very clear yet, although biodiversity has some role in ecosystem service delivery (see Section 7.3).

In the Netherlands, nature points are used in the analysis of consequences for nature (see subsection 6.4.2). Nature points are based on reference species in different ecosystems, roughly calculated as the number of species found compared with this refence number. This is multiplied by area and a weighting factor that is related with the scarcity of the ecosystem that is influenced. In contrast with PDF and MSA, the nature points include a reference to the scarcity of ecosystems. Otherwise, the principle is more or less the same (Arcadis and CE Delft 2018). Nature points have no price at the moment (De Bruyn et al. 2018).

7.3 The tension between the value of biodiversity and the value of ecosystem services

One key shortcoming in many biodiversity valuations is that they are actually focussed on ecosystem services instead of biodiversity. For example, Morelli et al. (2017) estimated the spatial covariance between areas important for ecosystem services and biodiversity in France and found low spatial congruence and in some cases even opposite associations between ecosystem services and each biodiversity component. In the study of Morelli et al. (2017), various diversity and community metrics for common bird communities were calculated: taxonomic diversity (TD), functional diversity (FD), community specialisation index (CSI), trophic index (TI), phylogenetic diversity (PD) and community evolutionary distinctiveness (CED). Morelli et al. (2017) found a negative effect of crop production on bird community evolutionary distinctiveness (CED), which in fact implies that conservation policies focusing solely on the economic value of ecosystem services will fail to protect overall biodiversity.

The focus on ecosystem services actually conceals the complexity of ecosystems and the potential relevance of biodiversity for human well-being. Bartkowski (2017) argues that biodiversity contributes to human well-being in ways additional to the value of those ecosystem services, meaning that the fact that an ecosystem is more or less biodiverse constitutes value-inducing effects additional to the value of ecosystem services. To properly include biodiversity in this picture, the perspective must be broadened by including the temporal and spatial dimensions (Figure 7.1 with biodiversity values in bold terms). This perspective should take into account that (Bartkowski 2017):

1. ecosystems are not static (temporal dimension 1); supply changes result from the dynamic nature of the ecosystems themselves and changes in human activities
2. human preferences (tastes, needs) are not static as well (temporal dimension 2); changes in needs/demand can be triggered by changes in ecosystems
3. the provision of ecosystem goods and services does not take place in a vacuum, but is usually embedded in larger networks of interactions (spatial dimension).

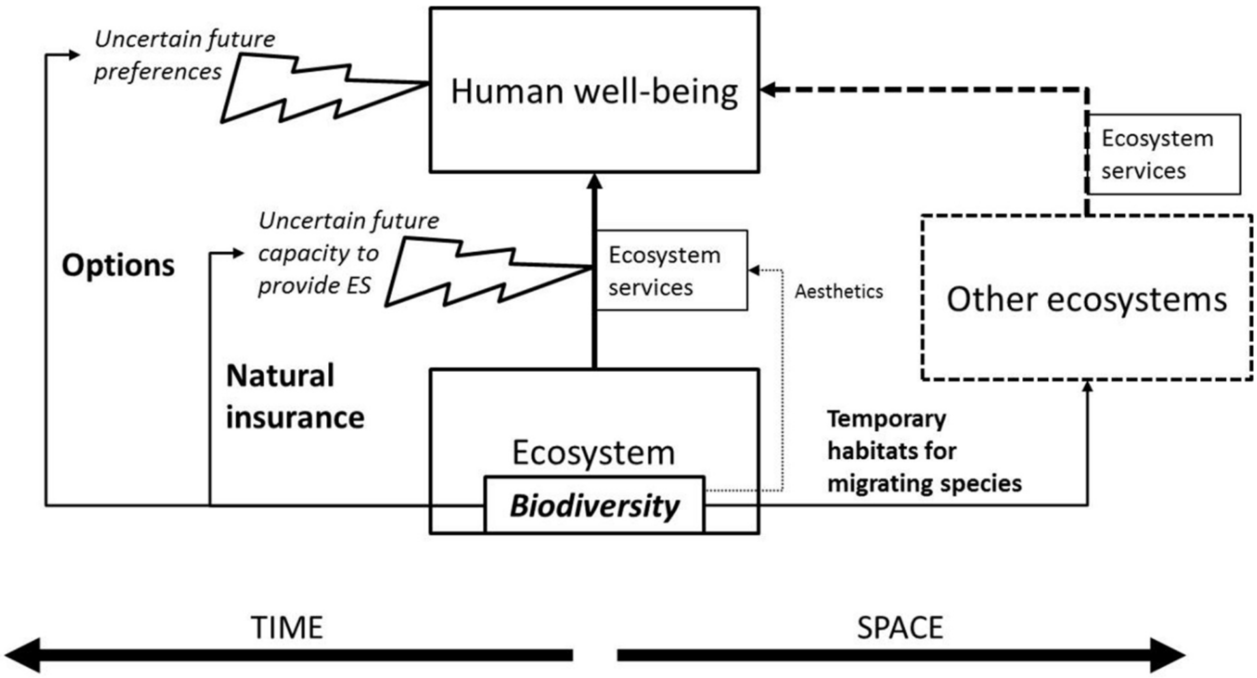

Figure 7.1 *Conceptual framework of biodiversity's economic value*
Source: Bartkowski (2017).

Many valuation studies focus on specific entities, for example a given ecosystem service or a given species, and the scarce studies that have biodiversity as a valuation object, mostly do not capture its complexity. With respect to biodiversity, three categories of economic value can be distinguished (Bartkowski 2017, see Figure 7.2, with biodiversity-related values in darker grey boxes):

- insurance value, which arises when biodiversity can reduce the uncertainty surrounding the provision of ecosystem services to risk-averse stakeholders
- option value, which arises from biodiversity's being a portfolio of options that reduce the uncertainty surrounding future preferences towards ecosystems
- spill-over value, which arises from the role of biodiversity in spatial interactions between ecosystems.

A significant part of biodiversity's economic value is a type of insurance against uncertainties about the future. Biodiversity has the ability to alleviate both the uncertainty of a given ecosystem's capacity to provide ecosystem services in the future, and the uncertainty of which ecosystem services will be demanded in the future. By stabilising the ecosystem, biodiversity provides a 'natural insurance' (Baumgärtner 2007) against fluctuations in the ecosystem's capacity to provide these services; at the same time, it is a pool of options to accommodate future changes in preferences and thus demand for ecosystem services. Another source of its value relates to the fact that ecosystems are not isolated from each other but interlinked in many different ways. These interlinkages result, among other things, in what can be called the spill-over value of biodiversity. Therefore, biodiversity in such ecosystems has spill-over effects on other ecosystems (Bartkowski 2017).

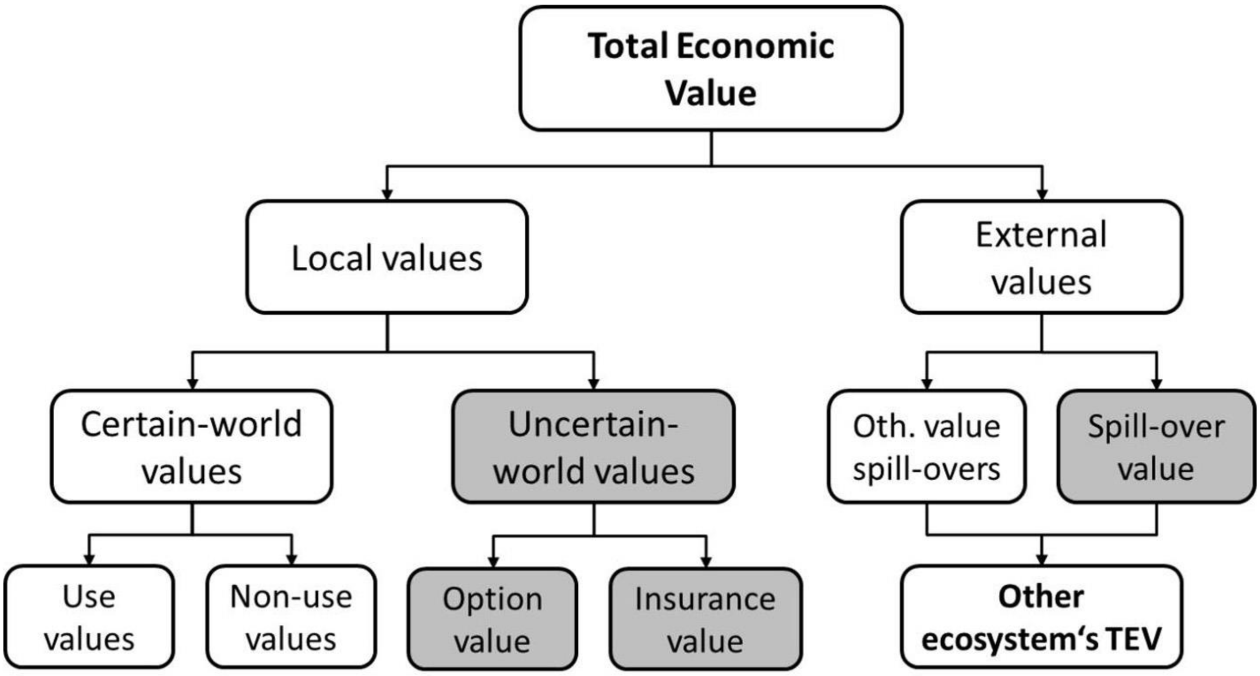

Figure 7.2 *Total Economic Value framework with insurance value of biodiversity*
Source: Bartkowski (2017).

With respect to the valuation of biodiversity, there are a number of specific challenges, namely the non-market nature of components of biodiversity value (especially in the case of insurance value); high levels of uncertainty involved in the relationship between biodiversity and human well-being and related to this, the multidimensionality, abstractness and complexity of the concept. Depending on the type of challenge, either the production function, revealed preferences (such as hedonic pricing) or stated preferences (discrete choice experiments or deliberative monetary valuation) will be the most suitable biodiversity method (Bartkowski 2017). Therefore, it is unclear whether and how biodiversity values could be included in systems of environmental-economic accounting (UNEP-WCMC 2015).

7.4 Biodiversity offset markets as an information source for biodiversity valuation.

An obvious approach to value biodiversity is looking for markets that value biodiversity. This type of markets does exist for some investment projects that require compensation for biodiversity loss, the so-called offsets. In this section it is investigated to what extent these markets currently and potentially can provide the relevant information for biodiversity valuation.

While development projects are essential for progress, they are currently a significant cause of today's unprecedented loss of biodiversity, which is recognised as one of the most critical global issues facing humankind. Biodiversity is lost as natural habitats are destroyed and fragmented, for example by agriculture, fisheries, forestry, oil and gas, mining, transport, tourism and the construction of infrastructure. In the search for sustainable development, governments, companies, financial institutions, and civil society are seeking innovative mechanisms to compensate for unavoidable losses to biodiversity and impacts on human well-being and attract more investment to conservation (Ten Kate and Crowe 2014).

To mitigate or compensate for biodiversity loss, steps to avoid, minimise, rehabilitate, and offset (or failing that, compensate) negative impacts are essential when development is planned, following the mitigation hierarchy (see Figure 7.3). The aim is to achieve at least no net loss (NNL) of biodiversity, and preferably a net gain (NG). To be effective, planning for biodiversity net gain must be included in the very early stages of development projects (BBOP 2009).

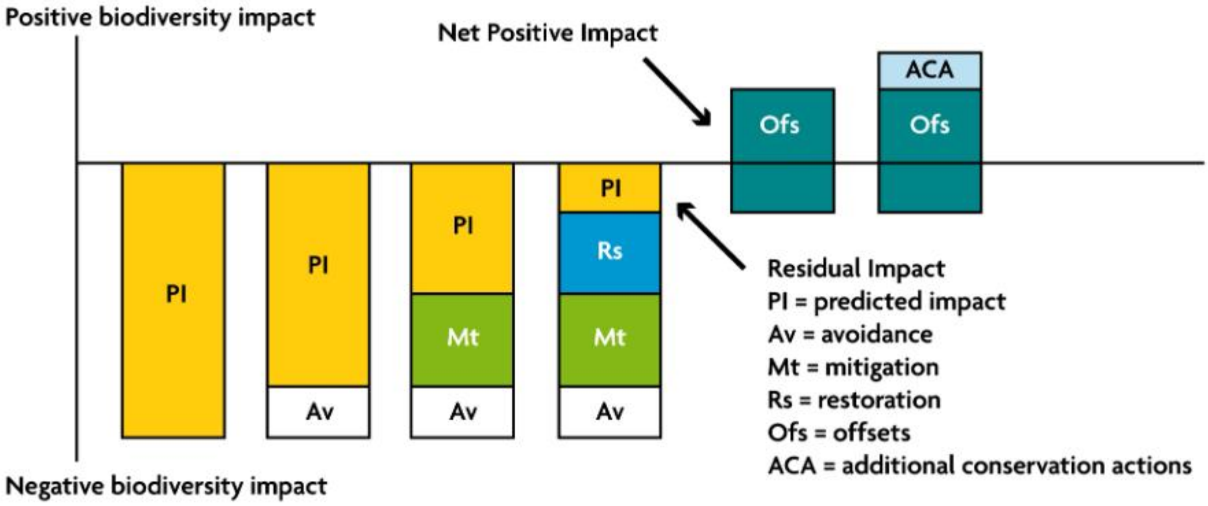

Figure 7.3 *Mitigation hierarchy and offsets*
Source: BBOP (2009).

Figure 7.3 schematically illustrates the process that is followed when the mitigation hierarchy is applied to a development project. Biodiversity value, and losses and gains in biodiversity, are shown on the left-hand axis. The large yellow bar shows the predicted impacts that result in biodiversity loss (BBOP 2009).

Generally, the mitigation hierarchy can be defined as follows (Ten Kate and Crowe 2014):

a. Avoidance: measures taken to avoid creating impacts from the outset, such as careful spatial or temporal placement of elements of infrastructure, to completely avoid impacts on certain components of biodiversity. This results in a change to a 'business as usual' approach.
b. Minimisation: measures taken to reduce the duration, intensity and / or extent of impacts that cannot be completely avoided, as far as is practically feasible.
c. Rehabilitation/restoration: measures taken to rehabilitate degraded ecosystems or restore cleared ecosystems following exposure to impacts that cannot be completely avoided and / or minimised.
d. Compensation or offset: compensation includes measures taken to compensate for any residual significant, adverse impacts that cannot be avoided, minimised and / or rehabilitated or restored. Measures to achieve no net loss or a net gain of biodiversity for at least as long as the project's impacts are biodiversity offsets.

Biodiversity offsets
According to Ten Kate and Crowe (2014), the term 'mitigation measures' refers to the full suite of steps in the mitigation hierarchy, including biodiversity offsets. 'Biodiversity offsets' are measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken. The goal of biodiversity offsets is to achieve no net loss and preferably a net gain of biodiversity on the ground with respect to species composition, habitat structure, ecosystem function and people's use and cultural values associated with biodiversity.

It is good to mention that some policies refer to biodiversity offsets, while others refer to compensation. At the same time, in some languages, such as in Dutch, there is no separate word for 'offset', and the word 'compensation' is always used in this context. Therefore, it seems worthwhile to address the difference between offset and compensation. Compensation is a very flexible term that can mean a number of different things. Dictionary definitions often refer to something, typically money, awarded to an individual as recompense for loss, injury, or suffering. Occasionally, compensation is defined more in terms of 'making good' specific damage, in which case it become closer to the definition of 'offset' above (except that it lacks the specific requirement for achieving 'no net loss'). Specifically, in terms of biodiversity, compensation involves measures to recompense, make good or pay damages for loss of biodiversity caused by a project (Ten Kate and Crowe 2014). While compensation can be in money, offsets are always formulated as biodiversity. Therefore, offsets are the relevant term in this study, and we will use the word compensation as synonym of offset.

Compensation in the Habitats directive
Within the European Habitats Directive (Council Directive 92/43/EEC), requirements for compensation are defined only in a general way. Article 6(4) of the Habitats Directive states:

> 'If, in spite of a negative assessment of the implications for the site and in the absence of alternative solutions, a plan or project must nevertheless be carried out for imperative reasons of overriding public interest, including those of a social or economic nature, the Member State shall take all compensatory measures necessary to ensure that the overall coherence of Natura 2000 is protected. It shall inform the Commission of the compensatory measures adopted.'

The term 'compensatory measures' is not defined in the Habitats Directive (Ten Kate and Crowe 2014). According to the European Commission, experience would suggest that

> 'compensatory measures are, strictly speaking, independent of the project (including any associated mitigation measures). 'They are intended to offset the negative effects of the plan or project so that the overall ecological coherence of the Natura 2000 network is maintained (EC 2007)'.

Requirements for compensation sometimes arise in the context of land-use change and are quantified relative to the loss of particular natural resources. Moreover, in several countries, compensation is often calculated based on certain costs associated with reforestation or restoration, such as obtaining and planting seedlings. Typically, such costs address only a part of the overall losses of biodiversity arising from land-use change (Ten Kate and Crowe 2014).

In the Netherlands, a similar hierarchy scheme is used with regard to nature that is protected by the European Habitats Directive. To begin with, an appropriate assessment has to be conducted to assess whether significant negative effects can be excluded. If the appropriate assessment of a plan or project (or programme) does not provide the required assurance that the natural features of a Natura 2000 site will not be adversely affected, permission for a plan or project can only be granted if (BIJ12 2019):
- There are no alternatives (A),
- There are imperative reasons of overriding public interest (D)
- The necessary compensatory measures are taken to ensure that the overall coherence of Natura 2000 (C).

This so-called ADC-test is referred to by the European Commission's 'last resort'. It is based on Article 6(4) of the Habitats Directive, while the appropriate assessment is based on Article 6(3). In general, the following holds: the more nature values are damaged, the more requirements for justification and consideration in the assessment of alternatives and imperative reasons of overriding public interest are imposed (BIJ12 2019).

Implementation of biodiversity offset
With respect to offsets, two areas are never ecologically identical. Therefore, it is necessary to assess how one can achieve biodiversity benefits at the offset sites that are equivalent to losses at the impact site. This assessment typically must consider how to (OECD 2016):
- Measure biodiversity
- Define acceptable trade-offs between biodiversity of different types and locations
- Manage the risk that biodiversity offsets are not delivered in the future and
- Account for time lags in the delivery of biodiversity offsets.

Biodiversity offsets may generally be implemented using one-off offsets, payments in-lieu or habitat and nature compensation banking. One-off offsets are undertaken by the developer themselves or by a third-party provider on their behalf, often a conservation NGO. Payments in-lieu is an approach to biodiversity offsetting whereby regulatory agencies levy fees on developers for causing adverse impacts to biodiversity. The agency then arranges for the collected fees to be spent on compensatory biodiversity conservation in a subsequent process. The level of payments in-lieu is typically based upon a reasonable cost estimate of the financial resources necessary to adequately compensate society for the biodiversity loss (OECD 2016).

The habitat and nature compensation banking approach relies on pre-existing biodiversity offset projects that can be purchased directly by developers of projects that reduce biodiversity. Habitat and nature compensation banking are a repository of existing offset credits where each credit represents a quantified gain in biodiversity resulting from actions to restore, establish, enhance and preserve biodiversity (OECD 2016).

In the Netherlands, the Dutch government announced the installation of a habitat bank (LNV 2020; Gorissen et al. 2020); this nature bank seems comparable with biobanking. The purpose of this nature bank is to take and record nature compensation and improvement measures in a structured way. The nature bank is primarily focussed on compensating the deterioration of nitrogen-sensitive nature areas, since high levels of nitrogen deposition are a key driver of nature in the Netherlands. The compensation measures will be implemented 'in stock': nitrogen-sensitive habitats will be created and expanded where they already occurred. National government projects and water boards, which serve a major public interest and rely on the aforementioned ADC test for their permits, can use nature from the nature bank as a compensation measure for the adverse effects they cause to nature. Moreover, LNV (2020) pointed out that the approach is that the compensation will always be greater than is strictly necessary based on the possible adverse effects of a project. It remains unclear whether this means that the government aims for a net gain in biodiversity.

There will be €125 million government funds available, that will be spent on setting up, filling and managing the nature bank. Until the bank is fully operational, part of the funds will be available for compensatory measures for national infrastructure and water safety projects. In the near future, the national government will work out the nature bank in more detail together with the provinces and land management organisations, including legal anchoring of the bank (LNV 2020). In 2022, the implementation of the habitat banking in the Netherlands has been stopped (RVO 2022).

Projects of the national government and water boards that represent an imperative reason of overriding public interest can make use of the nature bank. Priority will be given to projects within the framework of national infrastructure and water safety and defence projects that represent a security interest. These projects can call on the nature bank as soon as the compensatory nature has actually been secured. This will take several years. The national government invites provincial authorities, in consultation with municipalities, to join this national initiative or to set up a similar regional nature bank (LNV 2020).

The description above shows that offset markets are still in development but are meant to compensate and will generate prices to be paid for biodiversity loss. This information is useful to derive prices for biodiversity. However, at this moment correct measurement of biodiversity in these markets has not matured yet. There is no uniform solution available for the definition of acceptable trade-offs between biodiversity of different types and locations. Moreover, it is unclear whether:
- the offsets guarantee that the compensation is implemented sustainably and
- the methodology is suitable to account for time lags in the delivery of biodiversity offsets.

Therefore, at the moment the offset markets do not function sufficiently to get information on the value of different types of biodiversity.

7.5 Conclusion

With respect to the valuation of biodiversity, there are a number of specific challenges; namely the non-market nature of components of biodiversity value, high levels of uncertainty involved in the relationship between biodiversity and human well-being and, relatedly, the multidimensionality of the concept, and its abstractness and complexity.

The standard approach to biodiversity valuation is to use the indicators resulting from a biodiversity impact assessment (usually PDF or MSA) and to put a price on it based on the average ecosystem services provided by a high-quality ecosystem. This value is usually derived from a statistical study that relates a biodiversity indicator for a land area with ecosystem services provided in the same land area. The focus on ecosystem services conceals the complexity of ecosystems and fails to address the potential relevance of biodiversity for human well-being.

Another approach is to use information on the costs of compensating biodiversity loss from economic activities at a specific location with biodiversity improvements elsewhere, called biodiversity offsets. The habitat and nature compensation banking approaches, which involve markets for compensatory expenditures, could potentially provide information on the order of magnitude of costs needed to compensate for biodiversity losses. However, the manner in which the offsets are implemented is inconsistent, and many have been criticised for not providing sufficient assurance that the biodiversity damage caused is adequately compensated. Current information on prices of the offsets does not solve the problem, as these approaches require valuation of biodiversity loss to make explicit which biodiversity value to compare with.

8 Conclusions and recommendations

8.1 Conclusion

There is still a long way to go before biodiversity values will be integrated in decision-making on all levels. Including biodiversity values in the decentralised decision-making of a market economy requires consistent accounting of biodiversity, which is comparable with financial accounting systems. There are a large number of biodiversity indicators, of which the ecosystem quality indicators MSA and PDF are the most frequently used, but the relationship among these indicators and between the biodiversity indicators and the value of ecosystems provided is not sufficiently developed yet. This problem is relevant for ecosystem accounting related with national income accounts and input-output accounts, and for calculating and evaluating biodiversity footprints for companies and products. All these systems need analyses of the causal chain from activities to emissions to (midpoint) impact indicators and to (endpoint) indicators that are directly related with human welfare.

Monetary valuation of biodiversity is mostly avoided and when it is done it is mostly based on the value of ecosystem services, which are not clearly related with the biodiversity indicators used. Further improvement of the methods requires both further development of conceptual insights and better methods to monitor, measure and explain biodiversity changes.

While terrestrial ecosystem accounting currently has a focus on explaining a limited set of indicators such as MSA or PDF with quantitative relations, in marine ecosystem accounting the focus is more on describing many policy-related indicators and assembling large numbers of pressure factors. Because the logic of the two systems is the same, the challenge is to introduce the quantitative relations in the marine approach and to broaden the relations standardly investigated in the terrestrial approaches, while reaching a consistent approach for both of them.

8.2 Recommendations

1. Although there are a large number of biodiversity indicators in use, there is no framework that integrates all relevant aspects of biodiversity consistently. Biodiversity accounting systems used so far are mainly based on ecosystem quality indicators such as the Potentially Disappeared Fraction of Species (PDF), or the Mean Species Abundance (MSA). These indicators do not provide a full picture of what is really important in biodiversity, such as the multi-dimensionality of biodiversity. There is need for a coherent framework of biodiversity indicators linking multiple existing (and new) indicators at multiple scales.
2. The monitoring of biodiversity information is usually expensive. However, the use of new developments of information technology, remote sensing and citizen science in combination with improved ecological insights might lead to better and cheaper monitoring of biodiversity. More research and experiments are needed to improve the promising improvements of measuring and monitoring biodiversity and the causal relationships with pressures.
3. Although the logic of the accounting systems is the same, the approaches of terrestrial ecosystem accounting and marine ecosystem accounting differ widely. While terrestrial ecosystem accounting currently has a focus on quantifying a limited set of indicators such as MSA or PDF, marine ecosystem accounting focuses more on describing policy-related indicators and assembling indicators for pressure factors. More interaction between research on both approaches is needed to learn from each other’s approach and enrich their own approaches.
4. More efforts from research and policy are needed to develop a consistent biodiversity accounting system – analogue to the financial accounting systems – for the inclusion of monetary values for biodiversity in the decentralised decision-making of a market economy. Life cycle accounting (LCA) for products, accounting systems for companies and the national accounting systems relating economic activities with

monetary biodiversity values use their own type of causal logic and type of location specific information. There is a need for consistency for all types of analyses of the causal chain from activities to emissions to (midpoint) impact indicators and to (endpoint) indicators that are directly related with human welfare.

5. The biodiversity accounting systems related to the System of Environmental Accounts (SEEA) of the United Nations may potentially provide the local information required for biodiversity accounting. Ecosystem Service accounting from SEEA uses market prices for valuation, while for business application in many cases other valuation approaches are used. In some examples, such as the Nature Points in the Netherlands, the Biodiversity Metric in the UK or and the Healthy Ecosystem Metric in the UK, biodiversity quality indicators such as PDF or MSA are weighted with the location specific areas based values based upon criteria such as extinction risk and policy importance.
6. The current monetary valuations of biodiversity are mainly based on ecosystem services, although there is a limited relation between ecosystem services and biodiversity indicators. Therefore, the challenge is to explicitly integrate the different biodiversity indicators in an expansion of ecosystem service systems definition including the spatial distribution of biodiversity.
7. The way in which biodiversity offset schemes are implemented is inconsistent, because each offset scheme has its own rules and criteria. Moreover, many offset schemes do not provide sufficient guarantee for biodiversity compensation. Therefore, the transparency of offset schemes needs to be improved by developing a more harmonised framework for biodiversity offsets, for instance. Secondly, these frameworks also need to clearly identify who is responsible for guaranteeing that the compensation for biodiversity is realised.

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Appendix 1 Acronyms

Acronym	Description
ABDi	Agrobiodiversity Index
AI	Artificial intelligence
ANLb	Agrarisch Natuur- en Landschapsbeheer (Agricultural Nature and Landscape Management in the Netherlands)
AoC	Area of Concern
AoP	Area of Protection
BBOP	Business and Biodiversity Offsets Programme
BDM	Biodiversity Monitoring Switzerland
BDP	Biological Diversity Protocol
BERI	Bioclimatic Ecosystem Resilience Index
BFFI	Biodiversity Footprint Financial Institutions
BFMC	Biodiversity Footprint Methodology and Calculator
BHI	Biodiversity Habitat Index
BII	Biodiversity Intactness Index
BIJ12	Implementing agency in the Netherlands for the twelve Dutch provinces
BIM	Biodiversity Impact Metric
BIP	Biodiversity Indicators Partnership
BIRS	Biodiversity Indicator and Reporting System
BISI	Biodiversity Indicators for Site-based Impacts
BMS	Biodiversity Monitoring System for the Food Sector
BNGF	Biodiversity Net Gain Calculator
BPT	Biodiversity Performance Tool for Food sector
BSU	Basic Spatial Unit
CBD	Convention for Biological Diversity
CBF	Corporate Biodiversity Footprint
CBS	Statistics Netherlands (Centraal Bureau voor de Statistiek)
CDI	Crop Diversity Index
CED	community evolutionary distinctiveness
CEMP	Conservation Effectiveness Monitoring Program of the Australian Capital Territory government
CF	Characterisation factor
CIA	Cumulative Impact Assessment
CICES	Common International Classification of Ecosystem Services
CISL	Cambridge Institute for Sustainability Leadership
CLO	Compendium voor de Leefomgeving (Environmental Data Compendium)
CML	Centrum voor Milieuwetenschappen Leiden (Institute of Environmental Sciences in the Netherlands)
CPC	Central Product Classification
CSI	community specialisation index
DALY	Disability-Adjusted Life Year
DCS	Dutch Continental Shelf
DNA	Deoxyribonucleic acid
DPSIR	Causal framework describing the interactions between driving forces, pressures, states, impacts, and responses
EPL	Environmental Profit and Loss
EA	Ecosystem Accounting
EAA	Ecosystem Accounting Area
EASAC	European Academies Science Advisory Council
EBV	Essential Biodiversity Variables

EC	European Commission
ECN	Energy research Centre of the Netherlands
eDNA	Environmental DNA
EDP	Ecosystem Damage Potential
EEA	European Environment Agency
EHS	Ecologische Hoofdstructuur (National Ecological Network in the Netherlands)
EPL	Environmental Profit & Loss
ePNOF	Empirical Potentially Not Occurring Fraction of Species
EU	European Union
EUNIS	European Nature Information System
FAO	Food and Agriculture Organization of the United Nations
FCS	Favourable Conservation Status
FD	functional diversity
FRS	Fraction of Remaining Species
FSS	Farm Structure Survey
GBO	Global Biodiversity Outlook
GBS	Global Biodiversity Score
GDP	Gross Domestic Product
GD-PAME	Global Database for Protected Area Management Effectiveness
GEO-BON	Group of Earth Observations Biodiversity Observation Network
GEOS	Goddard Earth Observing System
GES	Good Environmental Status
GHG	Greenhouse gas
GIASIP	Global Invasive Alien Species Information Partnership
GIS	Geographic Information System
GRIIS	IUCN Global Register of Introduced and Invasive Species database
GRIS	IUCN Global Register of Invasive Species database
GTP	Global Temperature Potentials
GWP	Global warming potential
HDI	Human Development Index
HELCOM	Baltic Marine Environment Protection Commission (Helsinki Commission)
IAGTP	Integrated absolute global temperature change potential
ICES	International Council for the Exploration of the Sea
ICT	Information and Communications Technology
ILCD	International Reference Life Cycle Data System
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IPO	Interprovinciaal Overleg (Association of the Provinces of the Netherlands)
ISIC	International Standard of Industrial Classification
ISO	International Organization for Standardization
ISSG	IUCN SSC Invasive Species Specialist Group
IUCN	International Union for Conservation of Nature and Natural Resources
JRC	Joint Research Centre
LAI	Leaf Area Index
LCA	Life Cycle Assessment/Analysis
LCI	Life cycle inventory analysis
LCIA	Life Cycle Impact Assessment/Analysis
LEAP	Livestock Environmental Assessment and Performance
LFI	Large Fish Index
LIDAR	Light Detection And Ranging / Laser Imaging Detection And Ranging
LNV	Ministerie van Landbouw, Natuur en Voedselkwaliteit (ministry of Agriculture, Nature and Food Quality in the Netherlands)
LPI	Living Planet Index
LULUCF	Land Use, Land-use Change and Forestry
MBM	Mammalian body mass

ML	Machine learning
MOL	Map Of Life
MPA	Marine Protected Area
MSA	Mean Species Abundance
MSC	Marine Stewardship Council
MSFD	Marine Strategy Framework Directive
MSP	Marine Spatial Planning
MSY	Maximum sustainable yield
MTI	Marine Trophic Index
NBN	national biodiversity network in the UK
NDFF	National Database Flora and Fauna in the Netherlands
NDP	Net Domestic Product
NEM	Netwerk Ecologische Monitoring (Ecological Monitoring Network in the Netherlands)
NGO	Non-Governmental Organisations
NNN	Natuurnetwerk Nederland (National Ecological Network in the Netherlands)
NOAA	National Oceanic and Atmospheric Administration
NPP	Net Primary Productivity
NPV	Net Present Value
OECD	Organisation for Economic Co-operation and Development
OEF	Organisation Environmental Footprint
OSPAR	Convention for the Protection of the Marine Environment of the North-East Atlantic (Oslo-Paris)
PAC	Practice Adjustment Coefficient
PAME	Protected Area Management Effectiveness
PAS	Programmatic Approach for Nitrogen emissions (Programmatische Aanpak Stikstof)
PBF	Product Biodiversity Footprint
PBL	Planbureau voor de Leefomgeving (Netherlands Environmental Assessment Agency)
PD	Phylogenetic diversity
PDF	Potentially Disappeared Fraction of Species
PEF	Product Environmental Footprint
PEF	product environmental footprint
PEFCR	Product Environmental Footprint Category Rule
PMP	Provinciaal Monitoringsplan (Provincial Monitoring Programme in the Netherlands)
PREDICTS	Projecting Responses of Ecological Diversity In Changing Terrestrial Systems (database)
RCP	Representative Concentration Pathway
RIVM	Rijksinstituut voor Volksgezondheid en Milieu
RLI	IUCN Red List Index
RNN	Rekenmodule Natuurkwaliteit NNN (Calculation Module for Natura Quality of the NNN in the Netherlands)
RSPO	Roundtable on Sustainable Palm Oil
RVO	Netherlands Enterprise Agency (Rijksdienst voor Ondernemend Nederland)
SAR	Species-Area Relationship
SDG	Sustainable Development Goal
SEEA	System of Environmental Accounts
SEEA EA	SEEA Ecosystem Accounting
SEEA EBA	SEEA Experimental Biodiversity Accounting
SHI	Species Habitat Index
SIA	Seafloor Integrity Account
SNA	System of National Accounts
SNL	Subsidiestelsel Natuur & Landschap (Grant Scheme for Nature and Landscape in the Netherlands)

SP	Surplus Production
SRE	Stockholm Resilience Centre
SSSI	Sites of Special Scientific Interest in the UK
STAR	Species Threat Abatement and Restoration
TCM	Travel Cost Method
TD	taxonomic diversity
TEV	Total Economic Value
TI	trophic index
TRACI	Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts (model)
UN	United Nations
UNEP	UN Environment Programme
UNEP-MAP	UNEP’s Mediterranean Action Plan
UNEP-SETAC	UNEP’s Society of Environmental Toxicology and Chemistry
UNEP-WCMC	UNEP’s World Conservation Monitoring Centre
UNSC	UN Statistical Commission
USGS	United States Geological Survey
VF	Variation Factor
VMS	Vessel Monitoring through Satellite
WBI	Wild Bird Index
WDRC	Written Down Replacement Cost
WFD	Water Framework Directive
WGCEAM	ICES Working Group on the use of Cumulative Impact Assessments for Management
WSI	Water Stress Indicator
WSPS	Wild Seafood’ Provisioning Service
WTA	Withdrawal-to-availability
WWF	World Wildlife Fund for Nature
ZSL	Zoological Society of London

Appendix 2 Databases and models

A2.1 Databases

PREDICTS database
The PREDICTS project—Projecting Responses of Ecological Diversity In Changing Terrestrial Systems—has collated from published studies a large, reasonably representative database of comparable samples of biodiversity from multiple sites that differ in the nature or intensity of human impacts relating to land use. The 2016 release of the database contains more than 3.2 million records sample data over 26,000 locations and representing over 47,000 species. The database can help in answering a range of questions in ecology and conservation biology. To our knowledge, this is the largest and most geographically and taxonomically representative database of spatial comparisons of biodiversity.

Source: Hudson et al. (2017).

GRIIS database
The Global Register of Invasive Species (GRIS) was developed as a concept and prototype by the IUCN SSC Invasive Species Specialist Group (ISSG) in 2006 as part of a project undertaken for the Defenders of Wildlife on the Regulation of Live Animal Imports into the United States. The concept was revisited and expanded by the ISSG to address Aichi Biodiversity Target 9 and support its achievement- with the development of the Global Register of Introduced and Invasive Species (GRIIS). GRIIS hosted by the ISSG compiles annotated and verified country-wise inventories of introduced and invasive species. Development and population of the GRIIS was undertaken by the ISSG within the framework of activities of the Information Synthesis and Assessment Working Group of the Global Invasive Alien Species Information Partnership (GIASIP).

Link: https://griis.org/. Visited on July 20, 2023.

GD-GAME
a Global Database for Protected Area Management Effectiveness (GD-PAME)
The Global Database on Protected Area Management Effectiveness (GD-PAME) is a comprehensive database of protected area management effectiveness (PAME) information. It was originally developed at the University of Queensland. It is a joint effort of IUCN World Commission on Protected Areas (WCPA) and UN Environment, managed by UNEP-WCMC, and aims to compile PAME evaluations for all countries in the world from governments and other authoritative organisations, which are referred to as data providers. The GD-PAME is hosted on the Protected Planet website, along with the World Database on Protected Areas (WDPA) at www.protectedplanet.net, where the PA effectiveness data in the GD-PAME database can be viewed and requested.

Link: www.protectedplanet.net. Visited on July 20, 2023.

Red List database
The Red List is a database maintained by the International Union for Conservation of Nature (IUCN), that provides information on the global conservation status of species. Established in 1964, The IUCN's Red List of Threatened Species has evolved to become the world's most comprehensive information source on the global conservation status of animal, fungi and plant species. The IUCN Red List is a critical indicator of the health of the world's biodiversity. Far more than a list of species and their status, it is a powerful tool to inform and catalyse action for biodiversity conservation and policy change, critical to protecting the natural resources we need to survive. It provides information about range, population size, habitat and ecology, use and/or trade, threats, and conservation actions that will help inform necessary conservation decisions.

Link: https://www.iucnredlist.org/. Visited on July 20, 2023.

Collector App
The Collector App for ArcGIS enables organisations to use maps to gather data in the field and to synchronise the results with their enterprise GIS data. With this app, observers can update data in the field, log their location, and put the data captured back into a central GIS database directly from their phone or mobile device. This increases accuracy and helps eliminate recording errors. Fieldworkers are much more efficient and accurate, reducing error and time (Esri Insider 2016).

NDFF
The 'Nationale Database voor Flora en Founa' (NDFF) or the Netherlands Database for Flora and Fauna bundles data from more than 100 individual databases with the use of a pre-determined set of codes and taxonomies. All induvial databases store their data in the NDFF. NDFF is daily updated, and observations are controlled and validated. The Nature managers can choose to carry out the monitoring themselves in the area they manage, or they can leave it to the province. If managers carry out the monitoring themselves, they can apply for a subsidy from SNL. The provinces are supported in their monitoring tasks by the 'Nature Information and Nature Management' unit organisation BIJ12 through the development of ICT facilities and by taking on responsibility for national coordination. BIJ12 is a collaboration of the Dutch provinces, and a part of the collaboration of provinces, the so-called Interprovinciaal Overleg (IPO).

Link: https://www.ndff.nl/overdendff/data/. Visited on July 20, 2023.

BDM
Complementing other environmental information, BDM data serve to underpin nature conservation policy and other political decisions with a large impact on biodiversity, such as those made regarding agriculture and forestry. Biodiversity Monitoring Switzerland surveys the long-term development of species diversity in select plant and animal species. By keeping an eye on common and widespread species, BDM focuses on trends and developments in Switzerland's normal landscape. Inspired by Swiss Range Statistics and the Swiss National Forest Inventory, BDM chose a systematic sampling grid consisting of three distinct nationwide networks. The GLOBIO3 databases and a national biodiversity monitoring database of Switzerland, which has much more detail (BDM).

Link: https://www.biodiversitymonitoring.ch/index.php/en/home-footer. Visited on July 20, 2023.

Ecoinvent database
The Ecoinvent Database is a Life Cycle Inventory (LCI) database that supports various types of sustainability assessments. It enables users to gain a deeper understanding of the environmental impacts of their products and services. It is a repository covering a diverse range of sectors on global and regional level. It currently contains more than 18,000 activities, otherwise referred to as 'datasets', modelling human activities or processes. Ecoinvent datasets contain information on the industrial or agricultural process they model, measuring the natural resources withdrawn from the environment, the emissions released to the water, soil and air, the products demanded from other processes (electricity), and of course, the products, co-products and wastes produced.

Link: https://ecoinvent.org/the-ecoinvent-database/. Visited on July 20, 2023.

WWF Wildfinder database
The WildFinder database contains presence/absence data for the world's terrestrial amphibians, reptiles, birds, and mammals, by terrestrial ecoregion. Ecoregions are defined as 'relatively large units of land that contain a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of the natural communities prior to major land use change' (Olson et al. 2001). The 825 terrestrial ecoregions are nested within fourteen biomes with similar major vegetation features (e.g., tropical moist forests, temperate grasslands) and within eight biogeographic realms with similar geography, fauna, and flora (e.g., Neotropical, Nearctic). A full GIS coverage and data set for these ecoregions can be downloaded at:

Link: http://www.worldwildlife.org/publications/wildfinder-database. Visited on July 20, 2023.

LADA database
LADA is a scientifically-based approach to assessing and mapping land degradation at different spatial scales – small to large – and at various levels – local to global. LADA's main objective is to identify and understand the causes of land degradation and the impacts of land use, to enable adequate and sustainable land management solutions to be devised. A key principle of LADA is that land use is the main driver of land degradation, rather than soil, terrain or climate. Initiated in drylands, the LADA methods and tools have been adapted for application in other environments as well. LADA produced s specific assessment products and methods at three scales of operation: global, national and local.

Link: https://www.fao.org/land-water/land/land-governance/land-resources-planning-toolbox/category/details/en/c/1036360/. Visited on July 20, 2023.

GLOBIO4 database
The GLOBIO4 model produces spatial datasets with scenario outcomes for land use/cover and mean species abundance (MSA) for plants, warm blooded vertebrates (birds and mammals) and an overall MSA. For selected papers and contributions to global assessments these datasets are made available for download here. The GLOBIO4 scenario data is provided under a Creative Commons License (CC-BY 4.0) and is free to use. All global model output datasets are in GeoTif raster format and use the WGS84 coordinate system on a 10 arcsecond spatial resolution, this roughly equals 300 x 300 meter at the equator. In all databases used characteristics of parcels of land are related to the abundance of species. For each pressure factor, the abundance of species in disturbed parcels of land is compared with the abundance of species in an undisturbed situation for different natural conditions.

Link: https://www.globio.info/globio-data-downloads. Visited on July 20, 2023.

Brightway2
In Brightway2, a database is the object used to organise a set of nodes and edges in a life cycle inventory graph of the industrial supply chain and natural world. For example, a specific version of Ecoinvent could be a database, but so would a set of biosphere flows, as biosphere flows are also nodes in our inventory graph. Databases can be big, such as Ecoinvent, or as small as a single dataset. You can have as many databases as you like, and databases can have links into other databases. You can also have databases that each depend on each other. SimaPro differentiates between what it calls projects and libraries, but both would be a database in Brightway2.

Link: https://documentation.brightway.dev/en/latest/content/introduction/introduction.html. Visited on July 20, 2023.

Phyllis database
Database containing information on the composition of biomass and waste. Each data record with a unique ID-number shows information (if available) on:
- classification codes
- ultimate analysis: carbon, hydrogen, oxygen, nitrogen, sulphur, chlorine, fluorine and bromine
- proximate analysis: ash content, water content, volatile matter content and fixed carbon content
- biochemical composition
- calorific value (MJ/kg)
- (alkali) metal content
- composition of the ash
- remarks (specific information)

For each data record the source (reference) is indicated. In the database three types of weight units are used:
- as received (ar): weight percentage from the material in its original form (including ash and moisture)
- dry: weight percentage from the dry material (including ash)
- dry and ash free (daf): weight percentage from the dry and ash free material

From Phyllis you can obtain analysis data of individual biomass or waste materials or average values for a group of materials. You can get answers to questions such as: - What is the average sulphur content of wood? - What is the ash content of willow? - What is the average caloric value of chicken manure? Carbon conversion factors for products are used from the Dutch emission inventory and the Phyllis database, developed by ECN

Link: https://www.ecn.nl/phyllis2/. Visited on July 20, 2023.

MAGIC database
The MAGIC website provides authoritative geographic information about the natural environment from across government. The information covers rural, urban, coastal, and marine environments across Great Britain. Those are presented in interactive maps which can be explored using various mapping tools that are included. Natural England manages the service under the direction of a Steering Group who represent the MAGIC partnership organisations. With respect to data, one may start with easily available information, such as a habitat inventory and SSSI boundary information (MAGIC database in the UK) and search for information on species distributions (NBN atlas in the UK).

Link: https://magic.defra.gov.uk/. Visited on July 20, 2023.

Exiobase database
EXIOBASE is a global, detailed Environmentally Extended Multi-Region Input Output (EE-MRIO) table including a supply-use table. It was developed by harmonising and detailing supply-use tables for a large number of countries, estimating emissions and resource extractions by industry. Subsequently the country supply-use tables were linked via trade creating a multi-region supply-use tables and input-output tables from this. The EE-MRIO that can be used for the analysis of the environmental impacts associated with the final consumption of product groups.

EXIOBASE3 is one of the most extensive Environmentally Extended Multi-Region Input Output (EE-MRIO) systems available worldwide. EXIOBASE 3 builds upon the previous versions of EXIOBASE by using rectangular supply-use tables in a 163 industry by 200 products classification as the main building blocks.

Link: https://www.exiobase.eu/. Visited on July 20, 2023.

A2.2 Models

AWARE
AWARE is to be used as a water use midpoint indicator representing the relative Available Water REmaining per area in a watershed, after the demand of humans and aquatic ecosystems has been met. It assesses the potential of water deprivation, to either humans or ecosystems, building on the assumption that the less water remaining available per area, the more likely another user will be deprived.

Link: https://wulca-waterlca.org/aware/what-is-aware/. Visited on January 21, 2024.

GEOS-Chem
GEOS-Chem is a global 3-D model of atmospheric chemistry driven by meteorological input from the Goddard Earth Observing System (GEOS) of the NASA Global Modeling and Assimilation Office. It is applied by research groups around the world to a wide range of atmospheric composition problems. Scientific direction of the model is provided by the international GEOS-Chem Steering Committee and by User Working Groups. The model is managed by the GEOS-Chem Support Team, based at Harvard University and Washington University with support from the US NASA Earth Science Division, the Canadian National and Engineering Research Council, and the Nanjing University of Information Sciences and Technology.

Link: http://geos-chem.org. Visited on January 21, 2024.

LANCA
LANCA enables the calculation of characterised indicators that describe the effects of processes on the performance of various environmental systems. The LANCA® calculations are based on geo-ecological classification systems and make use of site-specific input data. The ecosystem functions of erosion resistance, mechanical filtration, physicochemical filtration, formation of new groundwater, and biotic production potential can be taken into account by this method within a Life Cycle Assessment.

Link: https://www.ibp.fraunhofer.de/en/expertise/life-cycle-engineering/applied-methods/lanca.html. Visited on January 21, 2024.

USEtox
USEtox is a model based on scientific consensus providing midpoint and endpoint characterisation factors for human toxicological and freshwater ecotoxicological impacts of chemical emissions in life cycle assessment, developed under the auspieces of the United Nations Environment Program (UNEP) and the Society for Environmental Toxicology and Chemistry, (SETAC) Life Cycle Initiative. USEtox represents best application practice as an interface between ever advancing science and a need for stability, parsimony, transparency, and reliability.

Link: https://usetox.org/model. Visited on January 21, 2024.

CLUE

CLUE model
The CLUE model is a dynamic, spatially explicit, land use and land cover change model. The different versions of the CLUE model (CLUE, CLUE-s, Dyna-CLUE and CLUE-Scanner) are among the most frequently used land use models globally. Applications range from small regions to entire continents. The CLUE model is a flexible, generic land use modelling framework which allows scale and context specific specification for regional applications. The model is distributed as freeware and can be downloaded together with tutorials through the link on this web-site. It is not possible to provide support on the application unless in the context of collaborative projects.

Link: https://sysdyn.org/info/spasial-dinamik/clue-s/. Visited on January 22, 2024.

IMAGE
IMAGE (Integrated Model to Assess the Global Environment) is an integrated assessment model that simulates the environmental consequences of human activities worldwide. It represents interactions between society, the biosphere and the climate system to assess sustainability issues such as climate change, biodiversity and human well-being. The objective of the IMAGE model is to explore the long-term dynamics and impacts of global changes that result from interacting socio-economic and environmental factors. The model is often used to develop scenarios supporting environmental assessments and provide insights into the consequences of different response strategies.

Link: https://models.pbl.nl/image/index.php/Welcome_to_IMAGE_3.2_Documentation. Visited on January 22, 2024.

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The mission of Wageningen University & Research is “To explore the potential of nature to improve the quality of life”. Under the banner Wageningen University & Research, Wageningen University and the specialised research institutes of the Wageningen Research Foundation have joined forces in contributing to finding solutions to important questions in the domain of healthy food and living environment. With its roughly 30 branches, 7,600 employees (6,700 fte) and 13,100 students and over 150,000 participants to WUR’s Life Long Learning, Wageningen University & Research is one of the leading organisations in its domain. The unique Wageningen approach lies in its integrated approach to issues and the collaboration between different disciplines.