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Handbook of Soil-Related Impact Assessment

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Abstract

This handbook is designed to support researchers in conducting state of the art assessments for evaluating impacts of agricultural soil management. Furthermore, it is addressed to a wider audience to aid understanding and interpretation of published impact assessments results. The handbook has been produced in response to a growing awareness of the fundamental role of soils and of the importance of sustainable soil management for coping with societal challenges such as food security, climate change or biodiversity loss.

The handbook is structured to follow the process of conducting impact assessments, starting with a short theoretical background, the setting of system boundaries and the definition of purpose and decision making level, and ending with methods to integrate findings for multiple impact areas into a single assessment. Readers are welcome to either follow this process and go through the handbook from start to finish, or to browse and look up individual chapters that are relevant for them.

While the handbook provides a methodical guideline for structuring soil related impact assessments, it does not discuss the various indicators available to measure each impact area. This complex aspect will be addressed in detail in a separate publication.

The information presented in the handbook is also available online through the BonaRes Assessment Platform (www.bonares.de/sustainability). This book is the first edition of a living document. Further editions will be published at irregular intervals to accommodate updates and new research generated within the BonaRes project.

Keywords Soil Function, Resource Use Efficiency, Ecosystem Service, Guideline, BonaRes

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1. Foreword

1 Welcome to the Handbook of Soil-Related Impact Assessment

This handbook is designed to support researchers in conducting state of the art assessments for evaluating impacts of agricultural soil management. Furthermore, it is addressed to a wider audience to aid understanding and interpretation of published impact assessments results. The handbook contains the information also provided online by the *BonaRes Impact Assessment Platform*[1]. It is the first edition of a living document and further editions will be published at irregular intervals to allow for updates, improvements and the inclusion of additional content[2].

The handbook is structured to follow the process of conducting impact assessments, starting with a short theoretical background, the setting of system boundaries and the definition of purpose and decision making level, and concluding with methods to integrate findings for multiple impact areas into a single assessment. Readers are welcome to either follow this process and go through the handbook from start to finish, or to browse and look up individual chapters that are relevant for them. While the handbook provides a methodical guideline for structuring soil related impact assessments, it does not discuss the various indicators available to measure each impact area. This complex aspect will be addressed in detail in a separate publication.

The length of the chapters is inhomogeneous, which to some degree is a reflection of project-related work schedules, but which is also partly on purpose. A strong emphasis is given to the structured selection of impact areas, because this is the stage with the highest risk of introducing bias. A second focus is on factors that are difficult to include in impact assessments and that are therefore often not considered by studies, such as leakage effects, indirect land uses changes or rebound effects. With regard to the latter, we were able to draw on new research findings generated within the BonaRes project. Finally, we put a focus on soil related ecosystem services within the European Environment Agency's CICES classification, because we perceive a lack of standardization in current research on ecosystem services to be an impairment to systematic reviews and to a transfer of research findings.

We hope you find this handbook and the BonaRes Impact Assessment Platform interesting and helpful, and we welcome any feedback that helps us to further improve them.

[1] www.bonares.de/sustainability

[2] The latest version of this handbook will always be available as free download from: www.bonares.de/services

2. Theoretical Background: Impact Assessment

2. Theoretical Background

2.1 Impact Assessment

Info-Box

Impact assessment is a set of logical steps that prepares information for decision-makers about the potential positive and negative consequences of policy or management options. In the context of soil management, impact assessment is crucial to recognising possible consequences of management decisions on social and natural systems across spatial and temporal scales.

The assessment process involves the evaluation and comparison of the consequences of different soil management options according to six steps: identification of problems, definition of objectives, development of alternative options, analysis of impacts, comparison of options, and recommendation for evaluation, monitoring, and implementation.

What is impact assessment?

Impact assessment is a method "*to structure the analysis of human-environment interactions*" (Helming and Pérez-Soba, 2011). It synthesises scientific knowledge to inform policy and management (Carpenter et al., 2006). The knowledge gained by impact assessment supports stakeholders in various areas of decision-making, such as soil management (de Olde et al., 2016), governance and policy formulation (Podhora et al., 2013), or research design for sustainable development (Bond & Poppe, 2012).

Impact assessment prepares evidence on the advantages and disadvantages of possible management or policy options by assessing their potential impacts on intended and unintended, short-term and long-term issues relevant for society (SEC, 2009; Helming et al. 2013). Impact assessment in the context of natural resource use requires the definition of the activities and the environmental and societal system components that are evaluated.

2. Theoretical Background: Impact Assessment

Why apply impact assessment to soil management?

Impact assessment in the context of soil management is crucial in order to recognise the potential consequences of management decisions on social and natural systems across spatial and temporal scales. The assessment evaluates and compares the consequences of different soil management options. This includes management options suggested from a scientific perspective (e.g. fertilisation schemes, tillage technologies, bio-control and microorganism utilization, crop rotations and catch cropping, application of sensor technologies, irrigation), those stemming from bioeconomic innovations (e.g. new cultivars and harvesting technologies, on-site harvest processing) and those derived from policy implementation at local, national and international levels (e.g. greening measures, carbon certificates). The latter category is particularly important because agricultural policies defined at the European level (CAP) and international conventions, such as those for combating desertification (UNCCD), have implications for domestic production (e.g. reduction of protein imports). Likewise, domestic decisions on land management (e.g. renewable energy strategies) may have sustainability implications in other world regions due to global market adjustments (discussed in chapter: *Indirect Land Use Changes*).

Assessments include issues of social acceptance, risk perception, economic costs and benefits, as well as environmental impacts beyond the soil system such as the interaction with water, air, climate and biodiversity. In order to structure different assessment approaches and utilize them for evaluating options in the context of creating a sustainable bioeconomy, we developed the *BonaRes Assessment Framework*, the *BonaRes Assessment Platform* (www.bonares.de/sustainability) and the *Handbook of Soil-Related Impact Assessment*. These products provide a systematic approach to assessing impacts of soil management on societal targets.

Assessment results can be used to inform stakeholders and facilitate a better alignment of human-nature interactions with societal goals, such as efficient use of natural resources or sustainable development (Helming and Pérez-Soba, 2011). Stakeholders include:

a) **Policy makers** that require assessments of current and future states of all soil functions and services, as well as tools to anticipate future driving forces and trends in soil management.

b) **Biomass producers (farmers)** that require site-specific information to develop optimal management solutions for tillage, fertilisation, pest management, crop rotations, cultivar selection, and soil conservation. Information needs of this group are at the highest spatial (field) and temporal (up to days) resolution.

c) **The biomass processing sector** that requires information on biomass quantities and qualities that can be made available at specific locations and points in time, including agricultural commodities and side products such as straw and residues.

d) **The wider research community** that requires information on soil related impacts and interdisciplinary linkages for their research (e.g. agriculture, hydrology, biodiversity, ecosystem services, climate modelling).

e) **Civil Society** and their institutions, which are concerned with effects of soil management on the environment and on human well-being.

2. Theoretical Background: Impact Assessment

How to apply impact assessment to soil management?

For soil related impact assessment, we use a **set of six steps** based on the **DPSIR framework** (see info-box in the following chapter). These steps can be seen as a dynamic process studying the societal reaction towards pressures on natural and human systems. It investigates a continuous loop of human-nature-human interactions. We put particular emphasis on the analytical step 4, the assessment step in the narrow sense, which evaluates the consequences of human activities on social and ecological systems from the perspective of societal targets, thereby taking an anthropocentric viewpoint.

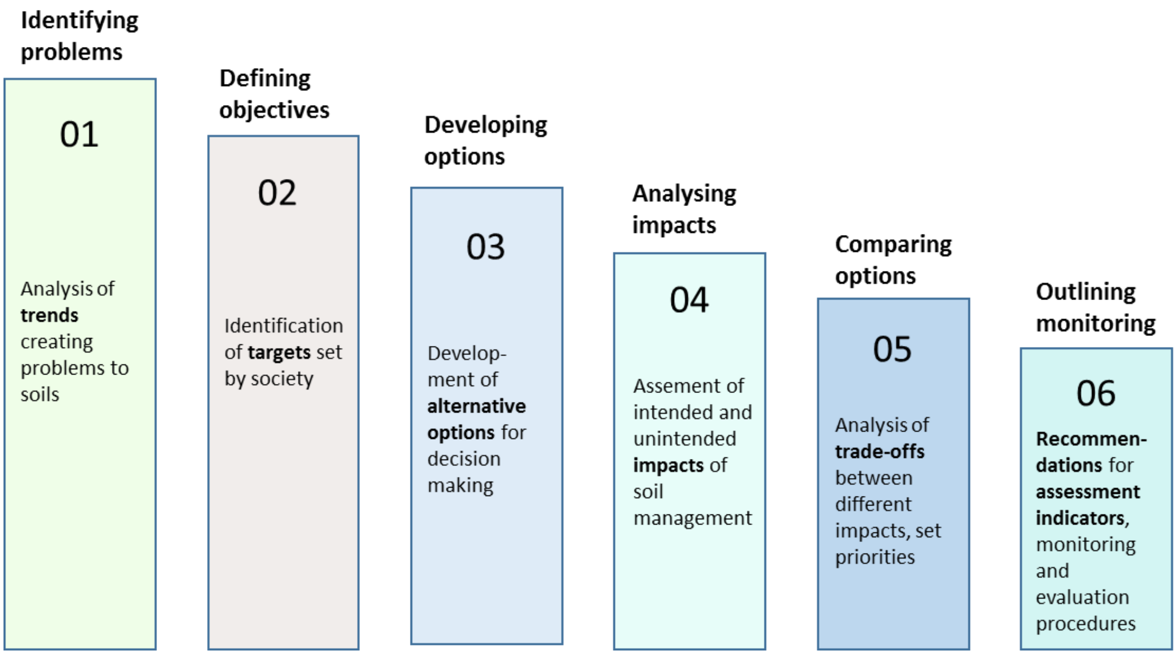

Figure 1: Steps of impact assessment (SEC, 2009; Helming et al.*, 2013) applied to soil management (Helming* et al.*, 2018)*

The six steps of impact assessment link the socio-economic system of societal target setting and decision-making with the natural system of biological, physical, and chemical process interactions (Figure 1). Adapted to soil research, the steps include:

(1) Analysis of future trends and ***driving forces*** for soil management options and identification of problems.

(2) Definition of human activities and options regarding soil management practices exerting ***pressures*** on soil systems.

(3) Analysis of the effects of human activities on the ***state*** of soil processes and soil functions. This analytical step concerns the soil system and depicts how soil processes are affected by soil management, and how this in turn affects the ensemble of soil functions.

(4) Assessment and valuation of direct and indirect ***impacts*** of soil management in the context of social, economic and environmental targets.

(5) Comparing the impact of different options, analysing co-benefits and trade-offs. At this stage, case-specific priorities can be assigned to the different impact areas, enabling a ranking of the analysed options.

(6) Recommendations for assessment indicators, monitoring procedures and the evaluation of policy implementation (Helming et al. 2013; Helming et al., 2018).

Impact assessment of soil management practices under the perspectives of resource use efficiency and maintenance of ecosystem services is an emerging field of research that will be highlighted on the in this handbook and on the BonaRes Assessment Platform. The limited number of examples found in current literature have a predominantly economic focus (Speiser et al., 2013; Peters et al., 2011). Assessments considering societal decision-making, e.g., by farmers, represent a substantial research gap with the exception of a few studies e.g., in Europe (Giupponi & Rosato, 1999) and in developing countries (Tittonell et al., 2015; Wanyama et al., 2010).

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2. Theoretical Background: Impact Assessment

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2. Theoretical Background: The DPSIR Framework

2.2 The DPSIR Framework

Info-Box

The DPSIR framework helps to structure complex human-environmental systems into a sequence from **Driver, Pressure, State** to **Impact and Response** (DPSIR). Due to its generic character, it can be used to investigate impacts of soil management on environmental and societal systems, to summarise existing knowledge of soil management research, and to detect research gaps. The DPSIR framework analytically links soil management (pressure) to soil functions (state) and its services (impact) that are assessed from the perspective of societal goals such as resource use efficiency or ecosystem services. Informed by the assessment results, society may react, governance may be adjusted (response) and steer soil management into a new direction (creating new drivers and restarting the sequence).

What is the DPSIR framework?

The DPSIR framework describes a **five-step causal** relationship of human-nature interactions. It has been developed for the assessment of relations between human activities and the environment (Gabrielsen & Bosch, 2003) and builds on the older three-step "PSI" framework originally developed by the OECD (1993). The five analytical stages help to structure complex human-environmental systems in a sequence from **D**river, **P**ressure, **S**tate to **I**mpact and **R**esponse (DPSIR). For example, driving forces, such as human needs for specific nutrition or improved technological facilities, lead to particular human activities in soil management and biomass production. These activities exert pressures on the environment, e.g., soil, which might change its properties and processes ("state") and corresponding functions. The potential environmental system changes have impacts on the societal system which can be evaluated by impact assessments. Informed by the valuation results, a societal response may take place that alters drivers and resets the causal chain of pressures, states, and impacts.

In this way, the DPSIR framework captures change and dynamic processes in human-nature interactions. It helps to integrate knowledge from different disciplines via adequate indicators in a cycle of impact assessment (Tscherning et al., 2012; Helming et al., 2013). One key strength of the concept lies in its adaptability to various impact areas, objectives and scales of analysis (Tscherning et al., 2012). Potential enhancements of the framework could be designed to also consider interconnections between stages beyond the connection from preceding to following stage (Niemeijer & De Groot, 2008).

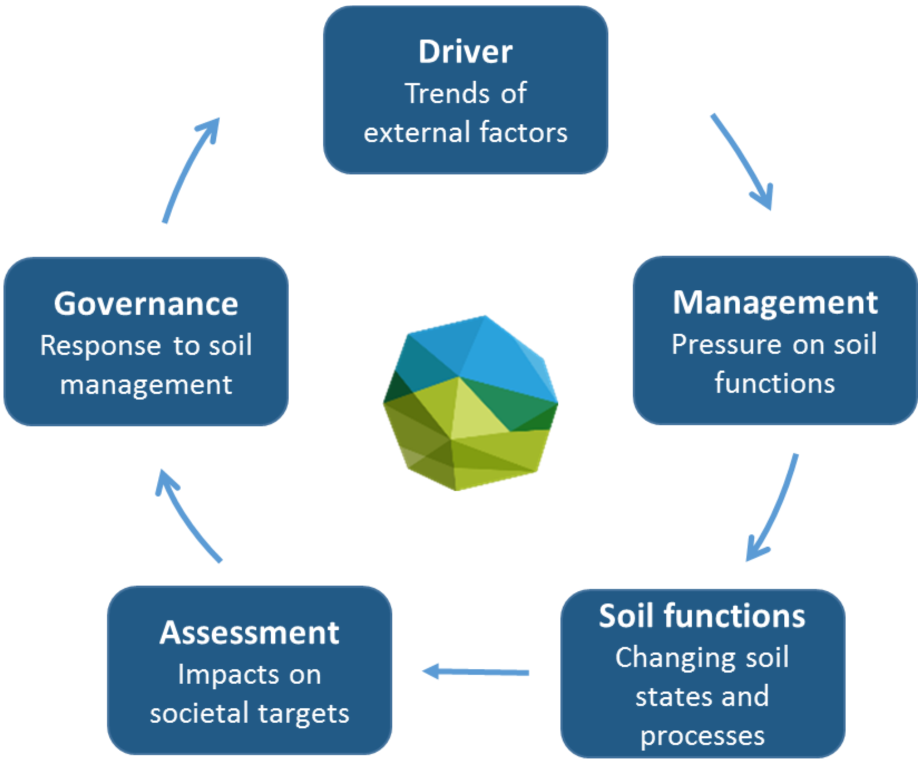

Figure 2: The DPSIR framework applied to soil functions in the socio-economic context, modified from Gabrielsen & Bosch (2003)

Why is the DPSIR framework relevant for soil related impact assessment?

The DPSIR framework analytically links soil management to soil functions and impacts on societal goals and value systems such as **resource use efficiency** or **ecosystem services**. Its perspective on human-nature interactions makes it possible to integrate scientific evidence from different disciplines into one joint framework, helping to develop comprehensive strategies for soil management that both sustain and improve soil functions. Finally, the integrated knowledge helps to provide scientific evidence for a multitude of societal groups and stakeholders involved in soil management (politicians, farmers, planners etc.) in order to support decision-making.

2. Theoretical Background: The DPSIR Framework

How to apply the DPSIR framework to soil management?

Due to its generic character, the DPSIR framework can easily be applied to very different settings of soil research. A number of studies already use the framework to investigate impacts of soil management on environmental and societal systems. For example, DPSIR has been applied to analyse changes in agricultural management such as precision agriculture and manure input on different scales (Bouma et al., 2008), to illustrate how land degradation control could be more effective (Gislanddottir & Stocking, 2005), to analyse how salinity, acidity and erosion threaten the ecosystem services of food production and the regulation of water quality in Australia (Holland et al., 2015), and to detect co-evolution of soil and water conservation policy with human-environment linkages (Wang et al., 2015). Furthermore, studies that focus on particular stages of the framework or on the relationships between stages can be integrated into a full application of the DPSIR framework and a broader cause-effect relationship. For example, Smaling & Dixon (2006) investigate the state, impact and response of soil fertility and nutrient management in global farming systems.

Overall, DPSIR framework can be used to summarize existing knowledge of soil management research and to detect research gaps. A particular example that reviews and synthesises scientific evidence for all single analytical steps of the DPSIR framework is provided by Schjønning et al. (2015). Their thematic focus is the potential compaction of sub soil due to mechanical stress by agricultural machinery. In detail, they identify studies describing external and internal drivers of particular field traffic practices, e.g., technical improvement and saving of labour costs cause the use of bigger and heavier machinery. They cite further studies that show how the pressure by field traffic on soils increases and has consequences for soil processes, such as higher bulk density, less pore volume size and reduced root system. This change of the soil state again has impacts on selected soil functions such as reduced biomass production, less storing and filtering capacities for water and nutrients. Furthermore, Schjønning et al. (2015) present studies investigating the impact on the social system such as the calculation of the relation between yield loss and cost savings in machine operations (cost-effectiveness of alleviating or avoiding soil compaction). They present potential responses to desirable or undesirable impacts, ranging from provision of subsidies in the political system to farmers' changes in soil management (e.g., increased application of nutrients or planting of taproot plants) to technical improvements towards smaller machinery.

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2. Theoretical Background: The BonaRes Assessment Framework

2.3 The BonaRes Assessment Framework

Info-Box

The BonaRes Assessment Framework has been developed to study impacts of soil management practices on societal targets. It is designed for agricultural systems in industrialised countries where both yield gap and agriculture's contribution to gross domestic product (GDP) are low.

The BonaRes Assessment Framework is based on the DPSIR framework and the six steps of impact assessment. It links driving forces and management decisions to soil reactions, changes in soil functions and their impacts on societal targets. Impacts within the categories of resource use efficiency and ecosystem services are analysed. These two assessment perspectives are considered complementary and strongly correspond with requirements of the German bioeconomy strategies. Furthermore, they are related to a number of targets under the United Nations' Sustainable Development Goals.

Application of the BonaRes Assessment Frameworks helps to structure research, to define impact areas and indicators, and to identify research gaps.

What is the BonaRes Assessment Framework?

The BonaRes Assessment Framework (Helming et al., 2018) is an analytical structure to guide research of impacts of soil management and soil function changes on societal targets. It links the DPSIR framework (Gabrielsen & Bosch, 2003) with the six steps of impact assessment. The framework has been developed to study impacts of soil management practices in industrialised agricultural systems characterised by low yield gaps and representing a low share of their countries GDP (gross domestic product), as is the case in most industrialized countries. Societal goals for soil management in these systems are often characterised by an **ecosystem services** perspective, i.e., increasing biomass production while maintaining the contribution of soil functions to the other ecosystem services, and by a perspective of **resource use efficiency**, i.e., an optimal return from invested resources. It is important to note that for other agricultural systems (e.g., small holder farming in developing countries) additional and/or different assessment perspectives are appropriate, e.g., motivated by societal targets of nutrition, poverty alleviation or rural development.

The BonaRes Framework (Figure 3) allows two modes of interaction between soil management pressures and impacts: soil-born via changes in soil processes and soil functions (solid arrow) and management induced irrespective of changes in the soil system (thin arrows).

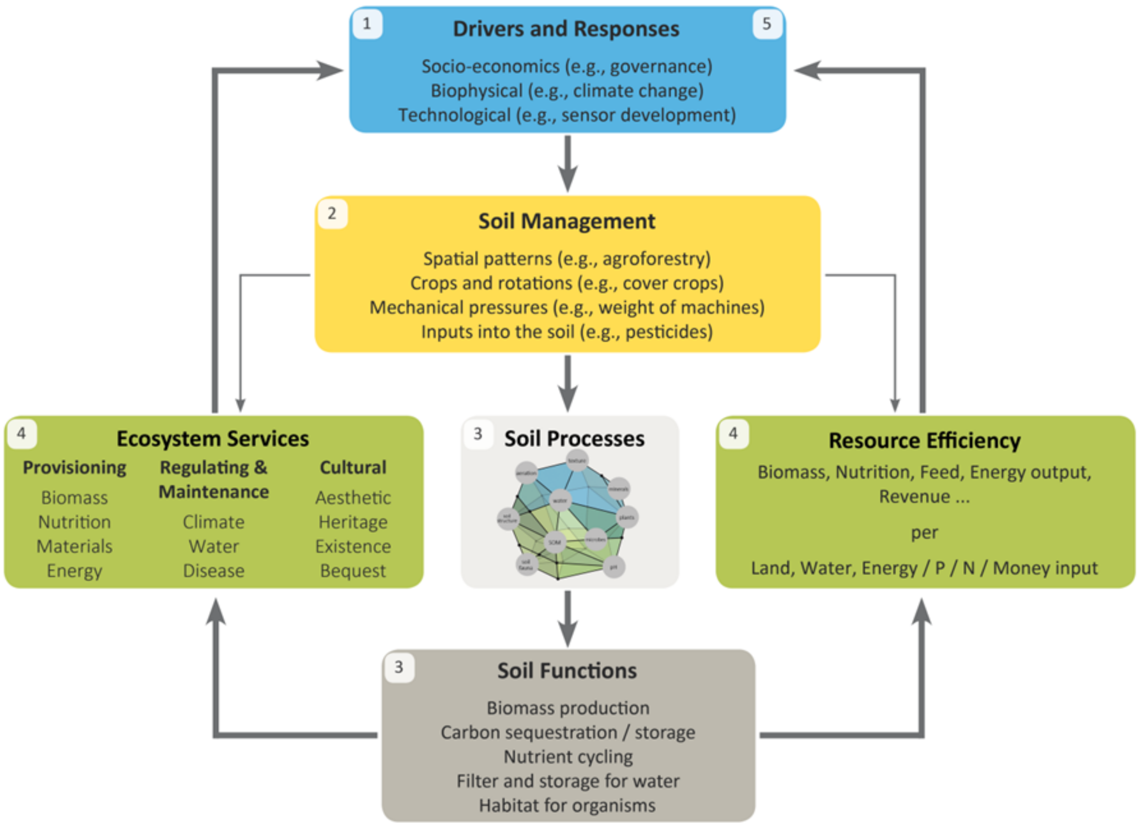

Figure 3: Analytical Framework for impact assessment of soil management and soil functions in BonaRes. Numbers refer to the five steps of the DPSIR framework: Drivers (1), Pressures (2), States (3), Impacts (4), and Responses (5).

Why apply the BonaRes Assessment Framework to soil management?

The linkage between impacts of soil management and societal targets has only recently gained attention in scientific and policy debates. For achieving a sustainable bioeconomy, assessing impacts of different soil management options is a necessity. However, to the best of our knowledge, no systematic approach to an impact assessment of soil management and soil functions on societal targets has yet been developed.

The BonaRes Assessment Framework links driving forces and management decisions to soil reactions, soil functions changes and their impacts to societal targets. Because the multitude of such targets makes it impossible to assess all potential impacts, it is necessary to limit assessments to specific

assessment perspectives. In the BonaRes Assessment Framework, we analyse impacts within the categories resource use efficiency and ecosystem services. These two perspectives are considered complementary, and they strongly correspond with requirements of the German bioeconomy strategies. Furthermore, they are related to a number of targets under the United Nations' Sustainable Development Goals.

Ecosystem service perspective: The concept of ecosystem services aims to demonstrate the value of nature to human societies and specifically refers to the 'final' outputs of ecological systems, i.e., the goods and services directly consumed or used by people. The importance of managing soil functions to support ecosystem services is widely acknowledged. However, the operationalisation of linkages between soil management, soil functions, and ecosystem services remains a challenge, e.g., where studies use un-linked single indicators.

Resource use efficiency perspective: Resource use efficiency can generally be defined as the ratio of benefits (generated by a product or process) divided by the amount of (scarce) resources used for that purpose (di Maio et al., 2017). Within the context of research for sustainable bioeconomies, it is necessary to determine how soil functions affect resource use efficiencies and additionally, to what degree soil management can increase efficiencies independent from soil functions.

Currently, only few assessments documented in the literature explicitly address the role of soils, although assessment results often implicitly reflect changes in soil functions due to the fundamental role of soils for crop growth. In contrast, the focus of the BonaRes Assessment Framework is explicitly on soils. Its application will help to systemically assess and compare opportunities and threats of (current and novel) soil management practices at different spatial and temporal scales. Insights gained in this way will help to strengthen the science-policy interface. They can be applied in stakeholder decision-making processes and used to inform the design of governance instruments aimed at sustainable soil management within a bioeconomy.

How can the BonaRes Assessment Framework be applied to soil management?

The BonaRes Assessment Framework forms the basis of the **BonaRes Impact Assessment Platform**. In particular, it is used to guide the development of a toolbox based on state-of-the-art research that provides indicators, methods and standards for impact assessment of soil management.

Systemising and test case application will reveal comprehensiveness and complementarity of the developed tools and identify further research needs of impact assessment.

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2. Theoretical Background: Soil Functions

Biomass production

Carbon sequestration

Storage and recycling of nutrients

Habitat for biological activity

Filtering and storage of water

2.4 Soil Functions

Info box

Soil functions are dynamic soil properties that emerge from complex interacting biogeophysical processes within the soil and that are relevant for human well-being. They are the basis for soil related ecosystem services and resource use efficiency. The concept of soil functions highlights the relevance of soils for human societies and facilitates their structured scientific assessment.

Soil functions first gained prominence through the proposed European Soil Thematic Strategy (EU 2006). For agricultural soils, five functions are considered particularly relevant (Schulte et al., 2014; Helming et al., 2018; Vogel et al., 2018):

- production of plant biomass,
- storing and filtering of water,
- storing and recycling of nutrients,
- habitat provision,
- and carbon storage

What are soil functions?

According to the cascade model of ecosystem services (Haines-Young & Potschin, 2010), soil functions are bundles of soil-based, biophysical-chemical structures and processes with the capacity to provide services to human societies (de Groot et al., 2010). They can be seen as the basis for soil related ecosystem services, but also for soil related resource-use efficiency. Soil functions first gained prominence through the proposed European Soil Thematic Strategy (EU, 2006) where they represented soils' contribution to: food, fibre and biomass provision, filtering, storing and accumulating of water nutrients and carbon, providing habitats and preserving gene pools, providing a platform for human activities, and acting as a historical archive. In an agricultural context, the functions as a platform for activities or as a historical archive are usually not considered relevant, whereas the functions pertaining to water, nutrients and carbon are of the highest importance.

2. Theoretical Background: Soil Functions

Consequently, Vogel et al. (2018) define the five (agricultural) soil functions listed below. The BonaRes project uses their definition and describes the functions on their website (https://www.bonares.de/soil-functions) in the following way:

Biomass production: "*Soils are indispensable for plant growth and herewith for the production of biomass, which can be used as food supply for humans and animals but also as raw material or energy source. The optimization of crop yields under strongly varying external conditions (temperature, precipitation frequency and intensity, etc.) is one of the biggest challenges with respect to biomass production. Soil fertility is linked to its non-rigid porous structure allowing for roots to penetrate, to store water while air and oxygen is also available and to retain nutrients to be used by plants and recycled by an enormous diversity of soil biota.*
Deficiencies in one or more of these conditions might be compensated by agricultural measures – e.g. compaction by tillage, missing nutrients by fertilization – on the long run, however, sustainable biomass production needs to rely on the natural capacity of soil to provide these essential features."

Storage and recycling of nutrients: "*The cycling of macro- and micronutrients is an essential function of soils. Mineral weathering generates a slow but constant source of elements made available for biochemical cycles. Soils are biochemical reactors, where numerous organic and inorganic compounds are decomposed and transformed so that the nutrients become again available for soil biota and biomass production. The engine driving this reactor is the myriad of organisms living in soil and taking profit from the energy stored in plant residues. Some part of it, however, is protected from decomposition for quite a long time so that soils act as a substantial carbon pool. An important feature is the ability of soil to retain nutrient within the root zone and to avoid leaching towards groundwater or emission to the atmosphere where the same nutrients might be harmful (e.g. NO_3 in groundwater or N_2O in the atmosphere)."*

Carbon sequestration: "*Globally, there is more carbon in soils than in the atmosphere and the aboveground biomass together. Plants extract carbon dioxide from the atmosphere by photosynthesis and incorporate it in their cell structures. This carbon then enters the soil as root exudates from living plants, which are utilized by soil organisms, or by decomposition of plant litter. These processes release carbon dioxide back to the atmosphere. They are more or less rapid depending on climate, soil type, land use, and management practices. Since carbon dioxide is a greenhouse gas, increasing soil organic matter stocks in soil is considered as a possible solution to mitigate climate change. Soil organic matter has also beneficial effects on soil fertility, water retention, susceptibility to erosion, and diversity and activity of soil organisms.*"

Habitat for biological activity: "*There are more species living below ground than above. Soils provide the living space for an enormous diversity of organisms, but in turn, are also formed by soil biota. The activity and diversity of soil organisms are largely affected by biotic and abiotic factors (e.g., climate, soil properties), but also by land use and management practices. Soil organisms play important roles in a number of soil processes such as nutrient cycling, mixing of soil material and development of soil structure. Due to the high diversity of soil organisms, many species are thought to be redundant, which means that they can be replaced by other species fulfilling the same function. However, certain species may play an irreplaceable role in soil functions. Soil structure provides a multitude of niches where different species can coexist with reduced competition for food and reduced stress of being caught by predators. This is a fundamental difference to the living space above ground.*"

Filtering and storage of water: "*Depending on soil properties, precipitation water is partitioned into storage, groundwater recharge, evapotranspiration, and surface runoff. Due to its porous structure, water can infiltrate into soil and is retained there via capillary forces just like in a sponge, so that it is available for plant growth and the development of other organisms. This reduces droughts and the generation of floods. Solutes and particles moving through soil have to pass the extensive inner surfaces of soil constituents where they might be sorbed or transformed. Hence, soils are also important to filter harmful substances, originating from industrial and agricultural production, municipal activities, or atmospheric deposits.*"

Please note: Each of these functions is a collective term with numerous sub-functions. For example, the function "storing and recycling of nutrients" comprises individual functions for the storage of all macro- and micro-nutrients, as well as functions relevant for the transformation of the different molecules containing these nutrients.

Why are soil functions relevant for the assessment of agricultural soil management?

The importance of managing soil functions to support ecosystem services is widely acknowledged (Glaesner et al., 2014; Schulte et al., 2014; Breure et al., 2012). They represent the biogeophysical reality that is affected by agricultural management. Sustainable intensification, beyond a simple increase or reduction of management intensity, can only be achieved if management can be adjusted to soil internal processes in order to optimize the provision of soil functions. Higher provision of soil functions would then in turn lead to a higher provision of ecosystem services and a higher resource use efficiency.

How can soil functions be considered in the assessment of agricultural soil management?

Soil functions are part of the overall category of ecosystem functions and as such, closely related to ecosystem services. Ecosystem services, the "direct and indirect contributions of ecosystems to human well-being" (Kumar, 2010), are based on the use or appropriation of ecosystem functions (Boyd & Banzhaf, 2007). The underlying terminology builds on the cascade model by Potschin & Haines-Young (2011), where biophysical function or structure, ecosystem function, ecosystem service, benefit and (economic) value are connected elements along a cascade-chain (TEEB, 2010; Braat & de Groot, 2012; Haines-Young & Potschin, 2010; Potschin-Young et al., 2017). By definition, impacts analysed in an impact assessment occur at intersection between the biophysical and the human sphere. Because soil functions fully belong to the biophysical sphere, they are themselves not considered to be impacts. Rather, they are the biophysical basis for impacts such as ecosystem services and resource use efficiency.

However, the operationalisation of linkages between soil management, soil functions, and ecosystem services (Helming et al., 2018, Stavi et al., 2016; Schwilch et al., 2016) and resource use efficiency (Helming et al., 2018) remains a challenge. To address these linkages, a clearly defined terminology is imperative. Unfortunately, the hierarchical relationship between soil functions and ecosystem services is not as straight-forward as it should be, and there is an overlap in definitions as discussed by Baveye et al. (2016). Consequently soil function and soil related ecosystem service are often not clearly distinguished in current literature (an exception is e.g. Pulleman et al. (2012) investigating soil biodiversity). The overlap in definitions is particularly evident for the production of biomass (a soil function) and the provision of plant-based food, fibres and energy (an ecosystem services).

In this handbook and on the BonaRes Assessment Platform, we will focus on soil-based ecosystem services and resource use in order to make the benefits of soils for human well-being visible and to inform sustainable soil management and policy. This also facilitates the integration of soil related impact assessments into a wider societal context. For the assessments we draw on biogeophysical information provided by the soil functions and their respective sub-functions. The individual linkages will be highlighted in the chapters on *Ecosystem Services* and *Resource Use Efficiency*.

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3. Designing an Impact Assessment: System Boundaries

3. Designing an Impact Assessment

3.1 System Boundaries

The definition of system boundaries is an obligatory first step in every impact assessment. System boundaries represent the thematic and spatio-temporal frame within which an assessment is conducted. Impacts that occur outside of this frame are not considered.

The reason for setting system boundaries is threefold: First, potential users of the assessment results should be able to quickly find the information relevant to them. Having to search for this information amidst large amounts of other data would reduce the assessments usefulness. Second and more important, the number of effects and impacts that could possibly be investigated is too high to be handled in a single assessment. Only by restricting an impact assessment to a predefined frame, the method becomes operable. Third, the choice of relevant impact categories varies with the system boundaries. Adequate impact categories and respective indicators can only be identified with clearly defined boundaries.

Setting system boundaries should be done with great deliberation and properly documented. Decisions taken at this step will strongly affect the assessment's results. In particular, decisions are required on

a) **Purpose of the assessment: (decision making level)** ⟹ **Who** is the assessment for?
b) **Spatial and temporal frame** ⟹ **Where & when** are impacts assessed?
c) **Impact areas** to be assessed ⟹ **What** type of impacts are assessed?

3. Designing an Impact Assessment: System Boundaries

Purpose of the assessment (decision making level):

Generally speaking, assessments should be designed to be relevant for an intended target group, e.g., by focussing on the impacts of different options available to a group of decision makers. Farmers, for example, may require a different set of information from an impact assessment than policy makers or members of the local administration (for more information see chapter: *Purpose & Decision-Making Level*).

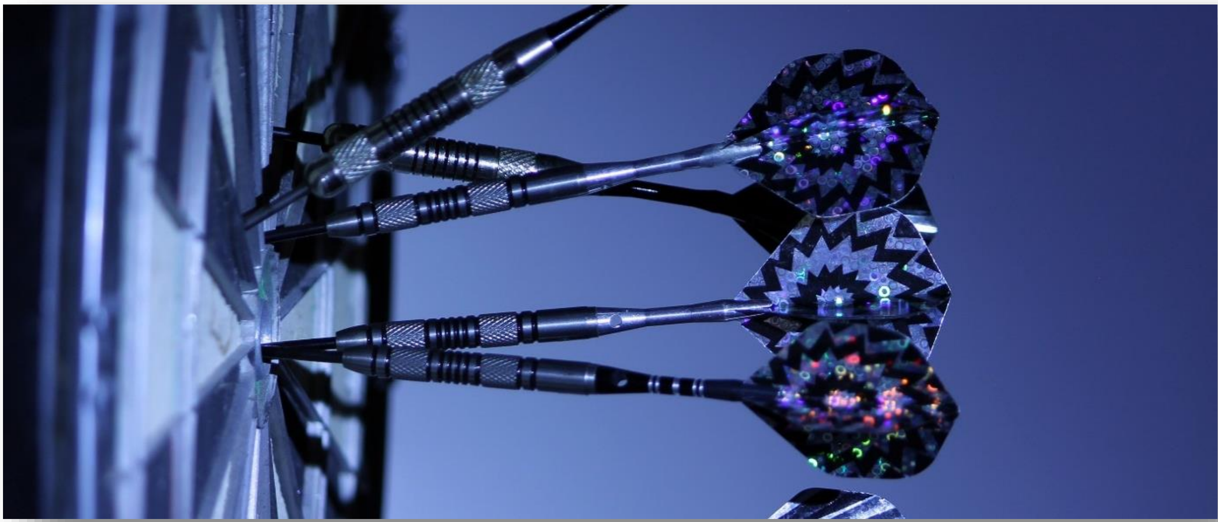
Spatial and temporal frame:

The Purpose of the assessment also influences the setting of the spatial and temporal frame. Another influencing factor are the impact areas that are analysed. Generally speaking, the frame must be set wide enough include all relevant impacts, but narrow enough to focus research efforts and avoid bloated data requirements (for more information see chapter: *Spatio-Temporal Scales*).

Impact area:

Finally, the impact areas for which effects are to be analysed and the methods and indicators by which they are assessed need to be chosen. The BonaRes Assessment Platform provides information on more than one hundred impact areas and on the methods and indicators to assess them. However, in any given assessment only a small selection of these impact areas **can and should** be used. The impact areas supported by the platform correspond to the assessment perspectives *Ecosystem Services, Resource Use Efficiency* and *Sustainable Development Goals* (for more information see chapter: *Impact Area Selection*).

3. Designing an Impact Assessment: Purpose and Decision-Making Level



3.2 Purpose and Decision-Making Level

At the start of an assessment, the assessment's purpose must be defined in order to properly select the topic, set the system boundaries and choose impact areas and indicators. If the purpose is purely scientific with no practical application intended, these variables are all determined by the specific research question. However, if the assessment is intended to inform decision makers, it is useful to align its topic with the so called **decision making levels**, i.e. the scales at which the intended groups operate and at which different options are available to them.

Agricultural management for any given site is influenced by the decisions of multiple actors. These are farmers, but also retailers, consumers, administration and policy makers. For these groups, assessments are particularly useful if they compare impacts of different options available to them, thus facilitating informed choices. An assessment for farmers, for example, could compare the impacts of different systems for slurry application while an assessment for consumers could analyse the impacts of different dietary choices. Focussing on specific actor groups *sets the spatial frame with regard to the assessments topic, i.e. defines the scale at which options are analysed* (see Table 1 below). The assessment of *impacts, however, should not be restricted to this scale* and relevant effects that occur on another scale must not be ignored. For example, an impact assessment of farmers' options for increasing soil carbon sequestration will analyse effects of different management practices at the local scale, but will also assess climatic impacts at the global scale.

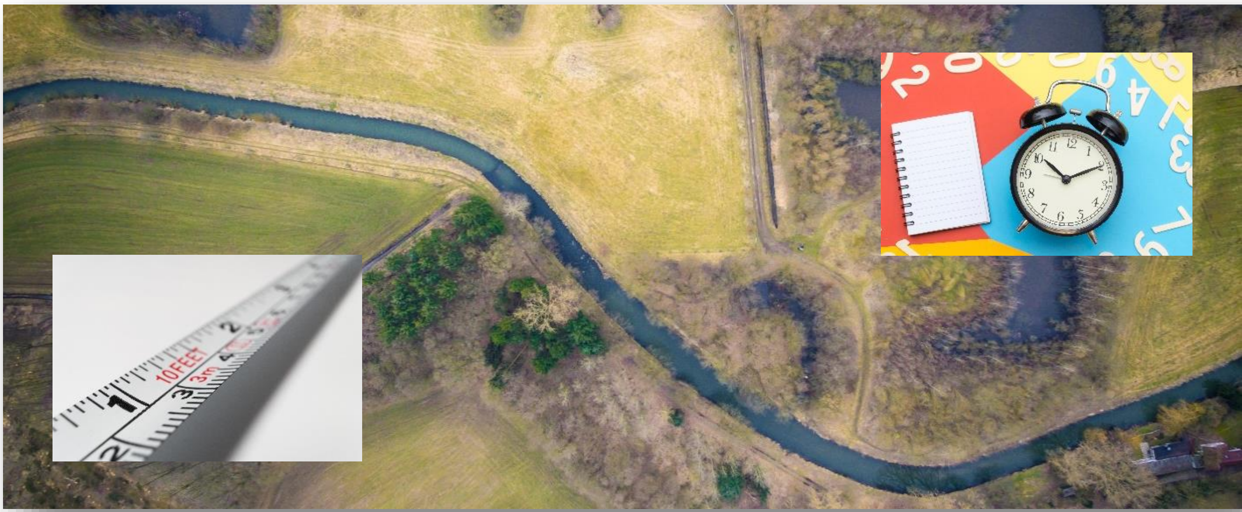
Below we present a selection of decision makers that influence agricultural soil management and provide an overview of the spatial scales they operate on.

3. Designing an Impact Assessment: Purpose and Decision-Making Level

Table 1: Decision makers that influence agricultural soil management and the spatial scales at which they mainly operate.

Decision makers	Spatial scale
Farmer	Farm – Landscape
Retailers	Regional – National - Global
Consumers	Local – Regional – National - Global
Local Administration	Local – Regional
Landscape Planning	Landscape
Regional Administration	Regional – National
National Government	National – Global
Multinational Institution	International – Global

3. Designing an Impact Assessment: Spatio-temporal Scales



3.3 Spatio-temporal Scales

Info-Box

In impact assessment, the spatial and temporal scales are part of the system boundaries, that is the frame within which impacts are analysed. While boundaries that are drawn too wide will result in bloated data requirements, loss of focus and may obscure relevant findings amongst non-relevant data, boundaries that are drawn too narrow will cause misleading results *because relevant impacts are not considered. Adequate definition of both **spatial and temporal system boundaries** is therefore essential.* **Leakage effects** and **indirect land use changes** pose a particular challenge because they require a very wide boundary setting to be detected. A structured way to include effects at large spatial and temporal scales is **life cycle assessment (LCA).** This approach is data intensive though, thereby limiting the number of impact categories that can be analysed**.**

What are spatio-temporal scales?

In impact assessment the spatial and temporal scales are part of the system boundaries, i.e. part of the frame within which effects are analysed. More precisely, define **where** and **when** impacts need to occur in order to be assessed. It is possible for assessments to conduct analyses at multiple spatial or temporal scales, for example analysing effects at field and at regional scale or effects within the current growing season and after ten years.

The selected spatio-temporal scales determine the set of available indicators and have a strong influence on the overall assessment results, because different processes occur at different spatial and temporal scales. The scales should therefore be defined at the start of an assessment and choices should be properly documented.

3. Designing an Impact Assessment: Spatio-temporal Scales

Why is the definition of spatio-temporal scales relevant for soil related impact assessments?

As with all research projects, the resources available for conducting an impact assessments are limited. This results in a trade-off between thematic, temporal and spatial width of the assessment. The wider the spatial and temporal scales are set, i.e. the more spatial scales from field to global and the more points in time are assessed, the lower is the number of impact areas that can be covered.

A definition of both spatial and temporal scales that fits the purpose of the assessment is essential. If spatio-temporal scales are set too wide, it will result in bloated data requirements, loss of focus and may even obscure relevant findings amongst non-relevant data. If on the other hand they are set too narrow, assessment results may be misleading because relevant impacts are not addressed. For example, effects of agricultural management on the biodiversity of farmland birds may not be visible at the field scale but manifest at the landscape scale. Management for increasing soil organic carbon will hardly show an effect after a single year but may have measureable impacts after 20 years.

With regard to defining spatio-temporal scales, leakage effects and indirect land use changes pose a particular challenge because they require a very wide boundary setting to be detected (Lambin & Meyfroidt, 2011).

How can spatio-temporal scales be defined in soil related impact assessments?

The spatial and temporal scales need to be chosen in a way that ensures that all relevant, potential impacts are addressed. This choice depends on the specific topic of the assessment, the purpose of the assessment and on the selected impact areas. For example, an impact assessment with the purpose to analyse policy options needs to cover impacts at the national or regional scale, while a farming system assessment would normally focus on the farm scale. However, if impact areas like habitats, aesthetic value or climate change mitigation are assessed, it may be necessary to expand the system boundaries to the landscape or even global scale.

Due to the complexity involved, expert based pre-assessment and potentially additional research is required before the start of the actual impact assessment.

Inclusion of effects up to a global scale may be required to account for effects like leakage or indirect land use change, where local management decisions result in effects emerging in other world regions. At the temporal scale, the definitions must be sufficiently wide to capture relevant improvements or deteriorations of soil functions or ecosystem service provision, which often only emerge after considerable time lags (Fremier et al., 2013). In the context of agriculture, consideration of multiple years is highly recommended in order to account for long term management effects, crop rotations, pre-crop effects and inter-annual yield variability (Preissel et al., 2015; Zhang et al., 2017).

A structured way to include effects at large spatial and temporal scales is **life cycle assessment (LCA)**. LCA are usually conducted to analyse the use of resources (e.g. water, land, energy) and the occurrence of emissions (e.g. greenhouse gasses, acidification, etc.) along a production process or product life cycle. However, the large spatio-temporal scales covered in LCAs lead to high data requirements and a high workload for assessments. This limits the number of impact areas that can be covered and results in a trade-off between spatio-temporal and thematic detail. Where a high number of impact areas needs to be assessed and the analysis of trade-offs between impact areas is in the focus of the study, LCA may therefore not be the method of choice and standard **multi criteria analysis (MCA)**, for which spatial and temporal scales are not formalised and can be freely adapted to the context of the research, may be preferable.

LCA focuses on environmental impacts of products and production processes (Roer et al., 2013). It seeks to integrate effects from all stages of the product cycle:

- Production and transport of required materials, machinery and fuels
- Production of the investigated product
- Distribution and use of the product
- Disposal or recycling of the product

This is a very complex and work intensive task and requires a substantial amount of information. It is usually facilitated by the use of LCA specific software and often draws on information from external, commercially available LCA databases (see www.nexus.openlca.org/databases for an overview). However, for a comparison between products or productions processes for which a part of the life cycle is considered to be identical, a full LCA may not be necessary. For different wheat varieties, for example, use and disposal will most likely be the same and an LCA comparing them may therefore focus on the remaining stages of the life cycle. Consequently, a number of studies use only partial LCAs, replacing the underlying “cradle to grave” or “cradle to cradle” concept with “cradle to farm-gate” or “farm-gate to farm-gate” (Brock et al., 2016; Hijazi et al., 2016).

LCAs can be used complementary to impact assessments when specific items (e.g. energy use) need to be assessed in detail along the value chain. Complex, systemic processes such as those related to ecosystem services are very difficult to be assessed with LCA.

Spatio-temporal boundary setting adequate for the respective assessment is necessary to ensure the validity of assessment results. Godinot et al. (2016) show the relevance of including production, transport and losses of inputs into the assessment of nitrogen use efficiencies. Likewise, Roer et al. (2013) demonstrate the considerable effect boundary setting has on assessment results by calculating LCAs of the same products using different system boundaries.

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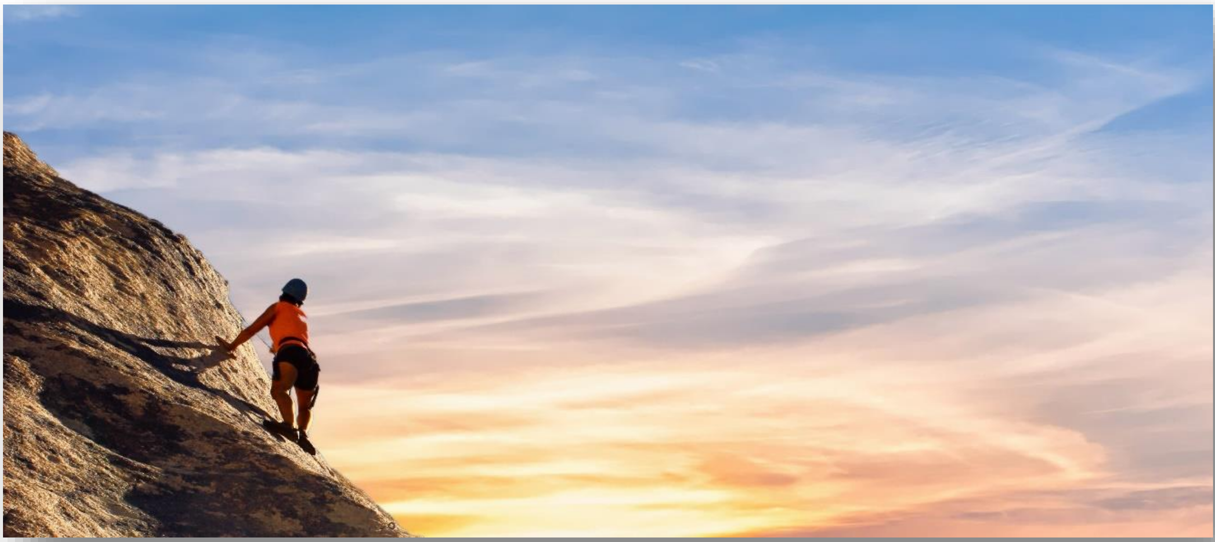


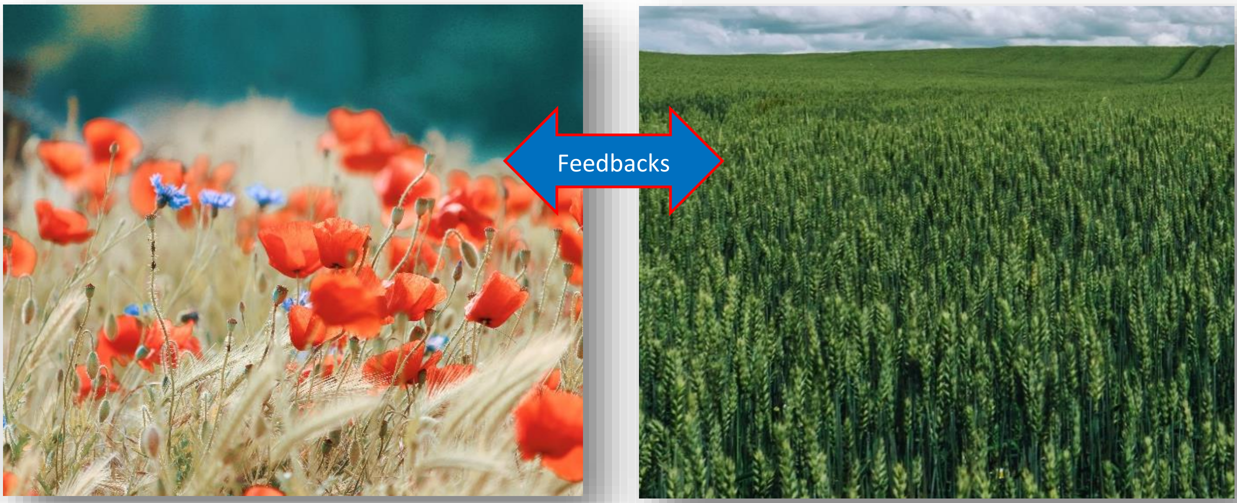
4. Assessment Challenges

Impact assessment builds on knowledge about causalities. This knowledge allows linking the assessed options to the impacts they are likely to cause. Where knowledge about causalities is still limited and impacts are therefore difficult to quantify, they are usually not considered in assessments. This is particularly relevant for reactions and feedback loops involving humans, who are typically influenced by a complex set of factors.

However, some of these reactions and feedback loops may have very serious impacts that need to be considered in sustainable decision making. Even where predictive power of theories and models is still limited and only rough estimates about impacts can be made, **we strongly argue for them to be considered in assessments** in order to safeguard against false results and perverse policy recommendations.

In this handbook we address **leakage effects, indirect land use changes and rebound effects** as assessment challenges. The importance of such effects can be illustrated by a simple (hypothetical) example: if an assessment were to analyse the impacts of afforesting all agricultural areas in Europe, ignoring leakage effects and indirect land use changes would lead to assessment results showing overwhelmingly positive effects for climate change mitigation. However, it is more realistic to assume that such a scenario would also lead to strongly increased food imports, a surge in food prices and a conversion of natural areas in other countries to farmland. This conversion would then partially or fully offset any positive effects on the global climate. Assessments that ignore these effects may therefore produce misleading assessment results.

4. Assessment Challenges: Leakage Effects



4.1 Leakage Effects

Info-Box

In the context of impact assessments, we use the term **leakage effect** to describe situations where measures for improving the situation (e.g., environmental protection) in one location result in a shift of problematic activities to other locations. Leakage effects may partly or fully offset intended positive effects of environmental policies by displacing, rather than alleviating environmental pressures.

Leakage effects gained public attention mainly with regard to a displacement of greenhouse gas emission. The problem of potential "carbon leakage" has been addressed both in in policy (European Commission, 2014) and science (Franzen & Mader, 2018; Naegele & Zaklan, 2019). Leakage effects can also cause Indirect Land Use Changes (ILUC). Lambin & Meyfroidt (2011) present an example of how land zoning for environmental protection may lead to displacement of populations or a stronger reliance on agricultural imports, resulting in land use changes and an encroachment of natural ecosystems elsewhere. Additional forms of leakage effects may exist, such as stricter regulations for pesticide and fertilizer application leading to higher imports of agricultural commodities from countries where regulations are laxer. However, studies addressing this are currently lacking.

The possibility of leakage effects should be considered in the design and evaluation of policies. For impact assessments to be able to detect them, spatial scales must be set wide enough to also capture effects that appear outside the area of policy application.

What are leakage effects?

In the context of impact assessments, we use the term **leakage effect** to describe situations where measures for increased environmental protection in one location result in a shift of problematic activities to other locations. Leakage effects may partly or fully offset intended positive effects of environmental policies by displacing, rather than alleviating environmental pressures. They can occur at all geographical scales (Henders & Ostwald, 2014). According to Watson et al. (2000), four leakage mechanisms can be distinguished:

4. Assessment Challenges: Leakage Effects

- Activity displacement
- Demand displacement
- Supply displacement
- Investment crowding

The term "investment crowding" describes a situation where (additional) investment in a project results in a reduction of investment from other sources. This mechanism is different from the other three in that it does not cause a shift of negative activities but rather a withdrawal of positive activities. It does stricto sensu not fall under our definition of leakage effects.

Leakage effects gained public attention mainly with regard to a potential displacement of greenhouse gas emission, the so called "carbon leakage". For example, if efforts are undertaken in a region to mitigate emissions, emission intensive production processes may simply be relocated, so that emission reductions at the global scale are smaller than those in the targeted region (Calvin et al., 2009). Because assessments only consider effects if they occur within the system boundaries, emissions due to leakage effects may easily be overlooked if the spatio-temporal scales are set too narrow, thereby resulting in a an overestimation of mitigation efficiencies (Watson et al., 2000). The potential problem of carbon leakage is addressed both in in policy (European Commission, 2014) and science (Franzen & Mader, 2018; Naegele & Zaklan, 2019).

With regards to land use, leakage effects can cause so called "indirect land use changes" (ILUC), often through a cascade of different displacement activities (Lambin & Meyfroidt, 2011; Watson et al., 2000). Accordingly, several studies dealing with land-use changes or agricultural intensification, greenhouse gas emissions or deforestation connect these to leakage effects due to policy changes (Calvin et al., 2009; Henders & Ostwald, 2014; Lambin & Meyfroidt, 2011; Lee et al., 2006; Lim et al., 2016).

Additional forms of leakage effects are possible, such as stricter regulations for pesticide and fertilizer application leading to higher imports of agricultural commodities from countries where regulations are laxer. However, studies addressing this are currently lacking.

Why are leakage effects relevant for assessing soil management?

Policies and innovations aimed at improving soil management and reducing pressures on the environment may, depending on their specific design, come with the risk of leakage effects. This could compromise their environmental benefit. For example, reducing the area of intensive agricultural production may alleviate pressures on the environment locally, but due to a global demand for agricultural commodities may also lead to more agricultural production in other regions. Impact assessment that fail to consider leakage effects overestimate the overall positive impacts of measures or arrive at wrong conclusions.

The risk of leakage effects should already be considered in the design of innovative soil management strategies. Impact assessments can be used to identify such risks and facilitate the design of additional measures to mitigate or prevent them.

4. Assessment Challenges: Leakage Effects

How can leakage effects be considered in assessments of soil management?

Identifying the consequences that changes in one production system have on other systems at different geographical scales is challenging (Henders & Ostwald, 2014). Only where impact assessment also consider effects occurring outside of the locations targeted by policies or innovations will they be able to identify potential leakage effects. A wide setting of the spatio-temporal system boundaries, considering effects up to the global scale, may be necessary for this.

Focussing on land use change, Lim et al. (2016) describe in detail economic mechanisms relevant for the occurrence of leakage effects from conservation measures or agricultural intensification. They present a framework of market responses that can be used to identify where unintended policy outcomes are likely.

Henders & Ostwald (2014) reviewed applicability, strengths and weaknesses of assessment methods for quantifying leakage effects as unintended consequences of policies. This included methods such as equilibrium modelling, statistical correlations and causal-descriptive methods. For global overviews and for determining the drivers and the occurrence of leakage, these authors recommend top-down approaches such as Computable General Equilibriums (CGE) and Multi-Regional Input-Output Analysis (MRIO) models. To find the connection between consumption and production across regions, they state that extended Material-flow Analysis (MFA) methods, representing bottom-up assessments, have a better resolution and can be used to cover the national level or to compare several countries.

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4. Assessment Challenges: Indirect Land Use Changes

4.2 Indirect Land Use Changes

Info-Box

Indirect Land Use Changes (ILUC) are conversions from one land use type into another (such as from forest to cropland) that are caused by changes in the production level of an agricultural commodity in another, often far away location. They are closely related to leakage effects. ILUC gained attention both in scientific research and in public discussions due to concerns that crop based biofuels might accelerate deforestation in the tropics. This could diminish or offset the benefits of a reduced consumption of fossil fuels.

While there is scientific consensus about the existence of ILUC effects, their quantification is still challenging. Differences between model estimates and the uncertainties involved in the ILUC factors derived from them complicates the consideration of ILUC in politics. Due to these uncertainties, inclusion of ILUC factors in assessments of bioenergy crops has received strong criticism from interest groups but also from within the scientific community. However, ignoring ILUC effects will underestimate the potential for environmental damages and might even result in perverse outcomes of environmental policies.

What are Indirect Land Use Changes?

The term **Indirect Land Use Changes (ILUC)** refers to changes in the production level of an agricultural commodity that cause land use changes in another, often far away location (IPCC, 2014). Land use changes under this definition are conversions from one of the six IPCC land use categories into another (forest land, cropland, grassland, wetlands, settlements, other lands), usually from forest land to grassland or from forest land or grassland to cropland (IPCC 2006). ILUC gained attention both in scientific research and in public discussions with regard to potential effects of crop based biofuels. Searchinger et al. (2008) argued that American biofuel production has a net negative effect on the global climate because it triggers deforestation in the tropics and thereby cause greenhouse gas emissions that outweigh any savings from the reduced reliance on fossil fuels.

4. Assessment Challenges: Indirect Land Use Changes

The introduction of biofuels and the policy support they received in industrialized countries resulted in changes that re-distributed a share of the global harvest towards this novel purpose (Searchinger et al., 2008). If a large area of cropland is no longer used to produce a specific commodity, then this affects the relationship between supply and demand for this commodity and increases the commodity's price. Higher prices in turn motivate conversion of other land use types (often forest or savannah) into cropland to produce this commodity (Ahlgren & Di Lucia, 2014; Villoria & Hertel, 2011, Wicke et al., 2012). Because many agricultural goods are traded internationally, land use changes can occur in other world regions, with global markets creating a tele-connection (Yu et al., 2013).

However, ILUC can also occur non market-mediated in the form of a competition for land resources. For example, the conversion of rangeland into cropland to produce biofuels (direct land use change) can cause a migration of cattle herders to other locations where they clear forests to create new rangeland (indirect land use change) (Lapola et al., 2010).

Indirect land use changes can have a significant impact on the economic, social and environmental dimensions of sustainability (Ahlgren & Di Lucia, 2014). Within the policy arena, awareness of the problem of ILUC due to biofuel production and of the greenhouse gas emissions resulting from ILUC is well-established. In the US, ILUC accounting was incorporated into the federal Renewable Fuel Standard in 2007 and into California's Low-Carbon Fuel Standard in 2009 (Breetz 2017). In 2015, the ILUC calculations for the Low-Carbon Fuel Standard were revised (Leland et al., 2018). In Europe, Directive 2015/1513 was passed in 2015 to account for ILUC effects from bioenergy and biofuels. The directive includes provisional values for greenhouse gas emission from ILUC for three feedstock groups and amends the Renewable Energy Directive (2009/28/EC) and the Fuels Directive (98/70/EC).

While there is scientific consensus on the existence of ILUC effects, their quantification is very challenging with various modelling approaches resulting in very different ILUC factors (Finkbeiner, 2014). This is not only due to technical difficulties, but also a result of conceptual differences (Flysjö et al., 2012). The resulting uncertainties make it difficult to consider ILUC in politics, where precise estimates of ILUC factors are sought for (Ahlgren & Di Lucia, 2014). Uncertainties are the basis for strong criticism from interest groups but also from within the scientific community (Finkbeiner, 2014).

Why are ILUC relevant in the context of soil management for a sustainable bioeconomy?

The transformation from a fossil fuel based economy to a bioeconomy that is based on renewable resources, a policy target in countries such as Germany, creates additional demand for agricultural commodities. While crop choices of individual farmers may not seem relevant for global land use, the sum of management decisions taken at farm level determines national production, affects the balance between supply and demand and contributes to the formation of global prices. Indirect land use changes are likely to occur if domestic food production is reduced while domestic demand remains unchanged. Where the gap between production and demand increases, prices increase as well and motivate conversion of new areas into farmland. Conversely, where the gap between production and demand is reduced, prices drop and motivate extensification and land abandonment. In this regard, indirect land use changes also depend on the degree by which productivity is increased through agricultural intensification.

4. Assessment Challenges: Indirect Land Use Changes

Land conversions, in particular conversion of forests and grasslands into cropland, incur massive ecological costs. This also includes multiple soil related ecosystem services, such as ability of soils to provide habitats or to provide climate regulation services (Haines-Young & Potschin, 2018). This is particularly relevant because soils represent the biggest terrestrial pool of organic carbon on earth and therefore play a substantial role in the global carbon cycle (Ciais et al., 2013). Global land use change has a considerable effect on that cycle by changing accumulation rate and turnover of carbon in soils, vegetation biomass and soil erosion (Deng et al., 2014).

Considering ILUC effects is highly relevant for soil related impact assessments and ignoring them could result in a false assessment of the overall sustainability of policies or practices. An example are first-generation biofuels which were originally regarded as a practical, technical solution for climate change mitigation. The consideration of ILUC effects, however, caused less favourable assessment results in a number of studies due to concerns about additional greenhouse gas emissions from deforestation and implications for global biodiversity (Searchinger et al., 2008; Di Lucia et al., 2012; Bentivoglio & Racetti, 2015*).*

How can ILUC be considered in assessment of soil management?

One major obstacle to considering ILUC is that they can not be directly observed. Multiple factors influence global rates of land use change and in particular conversion rates of tropical forests into farmland. Commodity prices play an important role, but many other factors such as land rights, population pressure, legislation or law enforcement are equally important. It is therefore very difficult to define what share of observed land uses changes is caused by price changes in global markets. This also complicates communication with stakeholders. For example, farmers in Europe who change from food crop production to energy crop production can not observe indirect effects of their actions and if land use change occurs somewhere else in the world, their contribution to this can not be proven (Ahlgren & Di Lucia, 2014).

ILUC effects can be assessed qualitatively or quantitatively. Qualitative assessments can be expert-based and could result in an assessment of whether or not a specific policy or management option is likely to cause ILUC or an assessment of which options is likely to cause the highest amount of ILUC.
Quantitative assessments can only be generated by models, with various modelling approaches resulting in very different ILUC factors (Finkbeiner, 2014). This is not only due to technical difficulties, but also a result of different concepts. The challenges of including ILUC effects into assessments are demonstrated by Flysjö et al. (2012). For a life cycle assessment of organic and conventional dairy farms, these authors included estimates of greenhouse gas emissions from land use changes caused by the import of soy meal. Estimates were based on different concepts for land use change:

- A: All use of agricultural land contributes equally to deforestation and expansion of farmland.
- B: Use of agricultural land contributes to deforestation and expansion of farmland.
- C: Only specific crops (such as soy) are responsible for deforestation and expansion of farmland.

They found that the land use change emissions dominated their results and that depending on the concept behind the estimates, either conventional or organic systems performed better. An overview of different ILUC model types and examples for their application is provided by Schmidt et al. (2015). Most of these models currently focus on greenhouse gas emissions.

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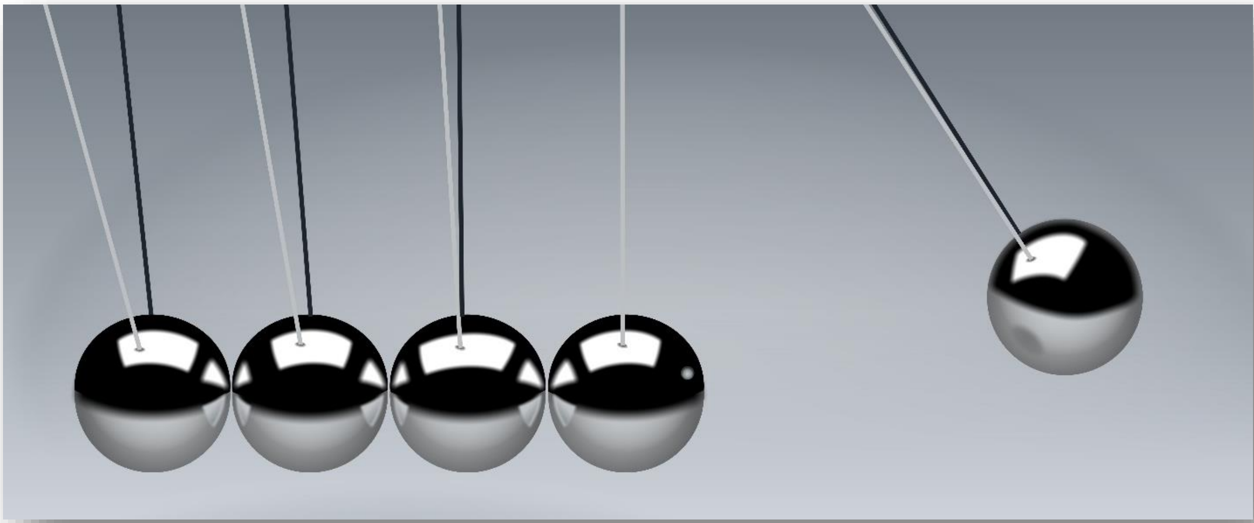
4. Assessment Challenges: Rebound Effects



4.3 Rebound Effects

Info-Box

Improvements in the technical efficiency of production processes are a means to reduce resource consumption. However, reductions achieved under real life conditions are usually smaller than what would theoretically be expected under *ceteris paribus* assumptions. Relevant actors (producers and consumers) will adapt their behaviour to account for the increased efficiencies, often causing additional resource use. This is called **rebound-** or **take back effect.** Rebound effects offset part or all of the resource savings that would otherwise be achieved by efficiency improvements. In extreme cases, they can even lead to an increase of total resource consumption (Jevons's paradox). Rebound effects are mainly explained by economic feedbacks. However, social-psychological factors are also relevant, especially for the consumer side.

Rebound effects can be divided into **direct effects**, **indirect effects** and **economy-wide effects.** They should be taken into account in all assessments dealing with efficiency improvements in order to obtain realistic estimates of resource savings and to adequately inform policy making.

What are rebound effects?

In many cases of efficiency increases, ex-post analysis reveals that reductions in total resource consumption are smaller than could be expected based on efficiency improvements. This effect is called **rebound-** or **take back effect** and results from changes in the behaviour of relevant actors after efficiency increases. The size of rebound effects (Equation 1) is defined as the share of resource savings which would be expected from efficiency increases if all other factors remained unchanged (*ceteris paribus situation*), but which do not materialise due to adaptions in the behaviour of relevant actors.

4. Assessment Challenges: Rebound Effects

$$\textit{Rebound Effect [\%]} = \left(1 - \frac{Resource\ Saving_{actual}}{Resource\ Saving_{ceteris\ paribus}}\right) * 100 \qquad \textit{Eq. 1: Definition of rebound effect size}$$

Rebound effect sizes vary and can be greater than 100% in extreme cases. This phenomenon is called backfire or Jevons´s paradox, after British economist W.S. Jevons who observed in his book "The coal question" (1865) that more efficient use of coal through technical innovation led to higher instead of lower total consumption. Based on this, he postulated that higher fuel efficiencies always led to higher resource exploitation (Jevons, 1865; Alcott, 2005). The same theory was reiterated in the 1980s by economists Khazzoom and Brookes and termed by Saunders (1992) the Khazzoom–Brookes postulate. While Jevons's paradox was formulated for the energy sector, cases where both resource use efficiency and total resource use increased are also documented in the agricultural sector for increases in irrigation efficiency (Pfeiffer & Lin, 2014; Sun et al., 2016; Perry et al., 2017).

Rebound effects are best documented for the field of energy efficiencies but are also applicable to the efficient use of other resources relevant to agricultural management. The majority of studies on rebound effects focus on rebound effects occurring due to economic feedbacks, as more efficient resource use lowers production costs and the price of goods and services based on the respective resources. Under the assumption of rational economic agents, this price effect leads to increased production and/or consumption, thereby offsetting part or all of the initial resource savings (Kolstad et al., 2014). However, de Haan et al. (2015) point out that especially for consumers, the assumption of purely rational behaviour is not valid, and that social-psychological factors can also create rebound effects. Policies promoting efficiency increases can affect both economic variables and social-psychological factors thereby causing both forms of rebound effects.

Rebound effects can be divided into a) **direct effects**, b) **indirect effects** and c) **economy-wide effects** (Kolstad et al., 2014). For rebound effects to occur, the resource use efficiency of a **process** must improve. In this section, we use the term process to refer to production processes which consume resources in order to provide services. These services can be immaterial (e.g. light, heat) or material (e.g. food, drinking water) and explicitly include the production of goods.

a) Direct rebound effects

Direct rebound effects occur if efficiency increases in a process result in an increasing demand for it, thereby also creating additional demand for the resource it consumes.

From an economic point of view, higher efficiencies mean lower production costs. This can motivate producers to increase production. Furthermore, producers may opt to use the more efficient process to substitute other production factors. Where lower costs result in lower prices, consumers usually react to this with increased consumption which in turn creates demand for increased production.

From a social-psychological point of view, services produced by processes that consume fewer resources are perceived as more positive than those produced conventionally. This is especially the case if those services are labelled as socially or environmentally friendly (e.g. energy from renewable sources). Where consumers restrict their consumption due to awareness of resource use implications, they may be less hesitant to consume services from more efficient processes, thereby creating additional demand.

b) Indirect rebound effects

Indirect rebound effects occur if efficiency increases in a process result in an increasing demand for other processes that consume the same resource.

Where higher efficiency translates into financial gains for producers and/or consumers, all or part of this money is usually spent on additional consumption of goods and services. For example, if households switch to more energy efficient appliances, they will save on energy consumption and money. However, if this money is used to pay for a holiday flight, part or all of the savings are offset in an indirect rebound effect.

From a social-psychological point of view, de Haan et al. (2015) argue that many consumers implicitly evaluate their own behaviour and apply a budget to their resource consumption. Being more environmentally friendly in one respect therefore may lead to less self-restriction in other areas.

c) Economy-wide rebound effects

Economy-wide effects are caused if not only individual actors are affected by the efficiency gains, but if there are resource use implications for the whole economy through increases in wealth, production and consumption, or through the introduction of technological innovations made possible by the more efficient process. For example, the introduction of mineral fertilizer dramatically increased the productivity (crop yield per hectare) of agricultural production, but had implications that also affected economies as a whole.

Why are rebound effects relevant for assessing soil management?

Increasing **resource use efficiency (RUE)** is considered to create a win-win situation by improving farmers' economic performance while at the same time alleviating pressures on the environment (O´Brien et al., 2014). However, different actor groups may react to such efficiency increases and cause rebound effects that offset part or all of the potential resource savings. In extreme cases, resource consumption can even be higher after efficiency increases than before. Not accounting for rebound effects can therefore result in a severe overestimation of resource savings.

For the agricultural sector, there are a number of studies on rebound effects related to more efficient irrigation technologies (Berbel & Mateos, 2014; Dumont et al., 2013; Loch & Adamson, 2015; Mehmeti et al., 2016), some of them reporting rebound effects greater than 100% i.e. an increase in resource use rather than a reduction. Pfeiffer & Lin (2014) report an example from western Kansas/USA, where a shift to more efficient irrigation technologies correlated with increased total water use. They explain this effect by farmers changing from non-irrigated to irrigated crops, reducing fallow periods and increasing per hectare irrigation. Similarly, Sun et al. (2016) found that increased water use efficiencies in Bayannur/Inner Mongolia in the period between 2000 and 2010 coincided with increased total water consumption. These authors state that savings were utilised to increase production rather than to alleviate pressures on the environment. In a global FAO review on improved irrigation technology, Perry et al. (2017) concluded that the introduction of high-tech irrigation without accompanying controls on water allocation usually results in an increase rather than a reduction of water use.

Several studies also address rebound effects related to higher agricultural yields in the context of the land sharing vs. land sparing debate (Mertz & Mertens, 2017; Phalan et al., 2011). Lambin & Meyfroidt (2011) discuss potential implications of rebound effects from raising agricultural productivity on the global demand for agricultural land. Ewers et al. (2009) show in a global study on yields and land use between 1979 and 1999 that land sparing occurred in some cases, but that for developed countries there was no evidence that increases in agricultural productivity resulted in lower per capita cropland demand. Addressing the link between agricultural productivity and greenhouse gas emissions, Valin et al. (2013) used the partial equilibrium model GLOBIOM to assess options for meeting the projected global food demand in 2050. In a scenario where all additional production was achieved through higher yields, they found strong demand-side rebound effects that reduced potential greenhouse gas savings by 50%. Studies on rebound effects related to efficiency increases in the agricultural use of other resources are mostly lacking. For a review on rebound effects in agriculture and an assessment of their relevance for emerging soil management practices, see Paul et al. (2019).

Impact assessments of efficiency measures need to consider potential rebound effects to ensure correct estimates of resource savings and to allow for an unbiased comparison between alternatives for reducing resource consumption. Assessments results that identify a risk of rebound effects can also be used to support the design of measures aimed mitigating or preventing such effects.

How can rebound effects in soil management be assessed?

To assess rebound effects, it is useful to first determine whether economic or socio-psychological preconditions are fulfilled. Only if the efficiency increases result in lower production costs or affect consumers' perception of the final products are rebound effects likely to occur. While these conditions are usually met to some degree and rebound effects are therefore common, the actual cost reduction or change in consumers' perception may be very minor and the size of the resulting rebound effects too small to be of relevance. What effect sizes are considered relevant must be determined individually within the context of a particular assessment.

Paul et al. (2019) present a framework for assessing rebound effects in agricultural land and soil management (Figure 4). Similar to a flow chart, users can answer "yes" or "no" questions to find out whether the occurrence of different rebound types is likely. Factors determining the size of the rebound effects are given in blue boxes.

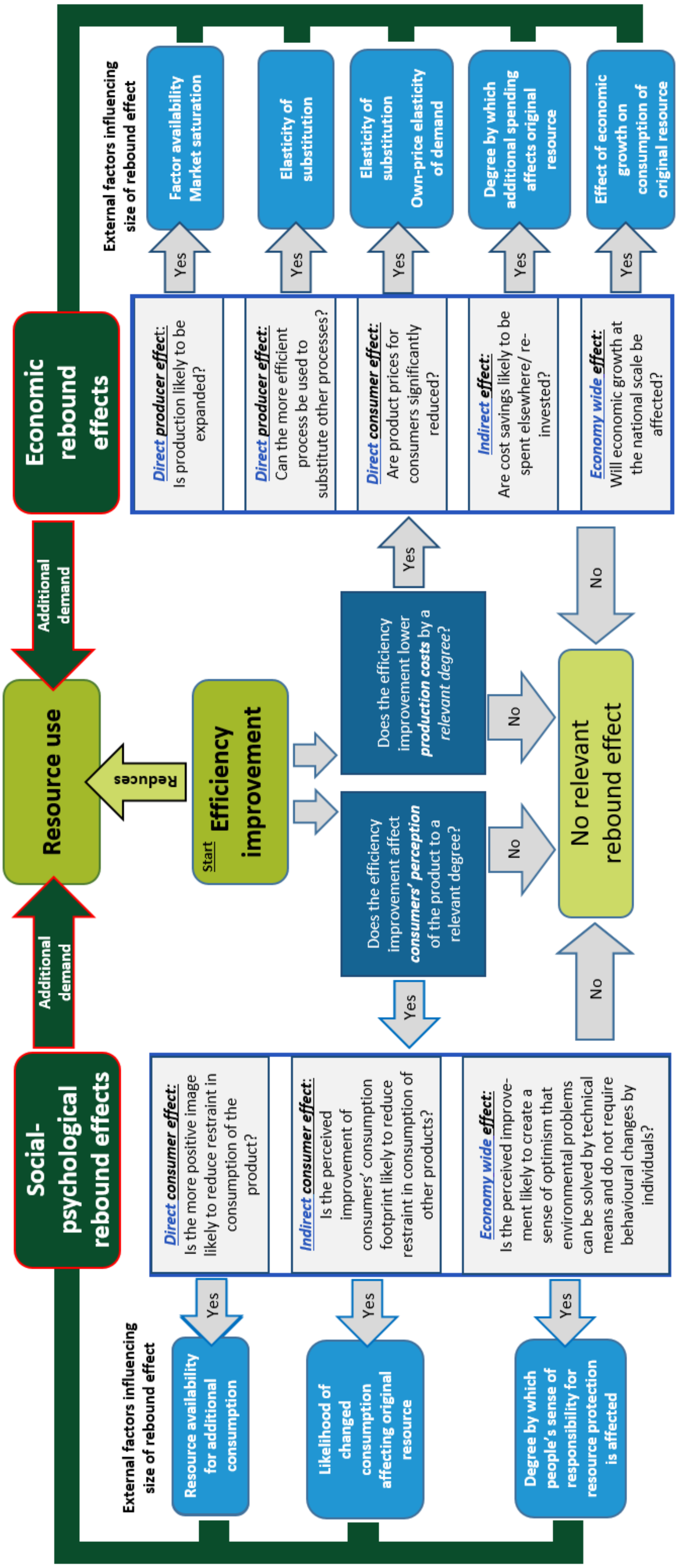


Figure 4: Framework for assessing economic or socio-psychological rebound effects from efficiency improvements. Factors determining the size of the rebound effects are presented in the blue boxes (Paul et al.*, 2019).*

4. Assessment Challenges: Rebound Effects

There is scientific consensus regarding the existence of rebound effects, but little agreement on their size. Values provided in scientific literature for increases in energy efficiency vary considerably (for an overview of studies, see Huesemann & Huesemann, 2008 and Kolstad et al., 2014). However, while it is difficult to estimate effect sizes, the implicit assumption of zero rebound effects made by studies that choose to ignore them is not supported by scientific evidence (Maxwell et al., 2011).

De Haan et al. (2015) discuss complications involved in assessing the size of rebound effects and argue that studies based on top-down evaluations of time series tend to overestimate effect sizes due to problems of separating increased resource consumption caused by economic growth and increasing consumers' wealth from increased resource consumption caused by increased efficiencies. On the other hand, they consider process-based, bottom-up studies to generally underestimate rebound sizes because they cannot account for economy-wide effects.

Because rebound effects result from changes in the behaviour of different actors and are based on multiple variables, it is challenging to assess their effect size, particularly in ex-ante assessment. In many cases only a very rough estimate will be possible. For rebound effects resulting from economic feedbacks, general equilibrium models can help to estimate reactions of market participants. For rebounds due to social-psychological factors, a major research gap exists (Maxwell, 2011). The use of the rebound effect assessment framework (Paul et al., submitted) may facilitate an assessment of rebound effect sizes. In general, the following factors are considered to be positively related with the size of rebound effects (de Haan et al., 2015):

- Relative size of the efficiency improvement (%)
- Relative share of savings of total production costs (%)
- Degree to which production is limited by availability of the more efficiently used resource (e.g. limited allocation of irrigation water, limited amount of allowable fertiliser application)
- Degree to which the more efficient process can be used to substitute other production factors
- Relative price reductions for the final product (%)
- Degree by which the consumption of the product is perceived as more positive (e.g. social or environmentally friendly) than before
- Demand elasticity, degree to which is demand is currently unsaturated
- Low degree of other costs associated with product consumption (e.g. in the field of mobility, travel-time is often more relevant than travel-cost)
- Degree to which the more efficiently used resource is also used in the production of alternative goods and services (the demand for which might increase with increasing wealth)
- Degree to which the more efficient resource use triggers technical innovation in other sectors

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5. Impact Areas

5.1 Impact Area Selection

Info-Box

At the start of an assessment, researchers must decide what (potential) impacts they want to analyse. This is the stage in the assessment with the greatest risk of introducing bias: If a relevant area of impacts is not included, the study will inevitably underestimate the positive or negative effects of investigated options. This may also disrupt a ranking of options, resulting in good options to appear worse than they really are (because their particular advantages are not seen) and bad options to appear better than they are (because their negative implications do not become evident). Which impact areas are likely to be relevant for a given assessment should be determined by a scoping study before the start of the actual assessment. Such scopings can be based on analysing scientific literature and policy documents, or on consulting with experts. Ideally, stakeholders should be involved in the process.

Assessment studies **may not be able to address all of the impact areas that have been identified as relevan**t, due to limitations in time, funding, personnel or data availability. This mostly applies to studies based on modelling or measurements. While those studies are still highly valuable for generating knowledge for a sub-set of impact areas**, they can not provide a full assessment or justify a ranking of options**. For this, the results of those studies need to be complemented with data from other studies. A practical way to achieve a full assessment in spite of research limitations is to complement modelling or measurement approaches with expert assessments. In this case, the BonaRes Assessment Platform recommends to first determine which of the relevant impact areas can be addressed by modelling or measurement and to generate values based on expert assessment for the remaining ones.

What is impact area selection?

Impact area selection is the process of deciding which impacts are going to be investigated in a given assessment. Because this selection is highly relevant for the assessment results, the selection process should be systematic, transparent and well documented. The rationale on which the selection is based should be purely thematic, i.e. based on the question which impacts are likely to be affected.

Selecting impact areas based on data availability will not produce reliable assessment results that are suited to evaluate sustainability or compare options.

Why is impact area selection relevant for the assessment of soil management?

Investigating impacts is complicated and usually requires thorough planning, as well as one or more of the following: calibrating and running models, conducting measurements, setting up and performing experiments or preparing literature reviews, stakeholder consultations or expert interviews. For this reason, impacts that are not specifically investigated, will most likely not be detected.

5. Impact Areas: Impact Area Selection

Where a management or policy option investigated in an assessment has significant positive or negative impacts, not addressing them will inevitably result in a biased assessment result. As an example, please consider two hypothetical management options A and B, both with their respective advantages and disadvantages. Figure 5 shows their performance for 11 impact areas from the assessment perspective ecosystem services. In the figure, impacts are given as percentages relative to a business as usual option, with values higher than 100 indicating a higher provision of the respective ecosystem service and values lower than 100 indicating a lower provision of the services.

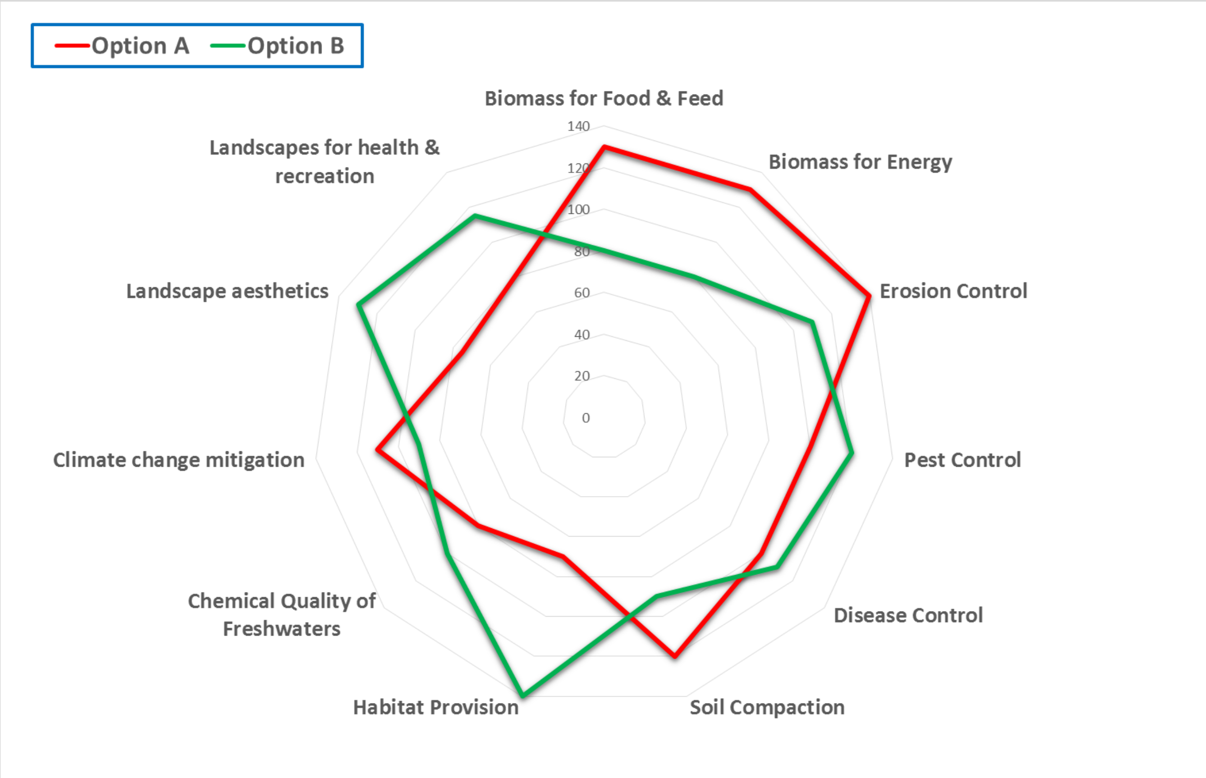

Figure 5: Two hypothetical management options (A and B) and their performance for 11 ecosystem services. Values represent percentages of service provision relative to business as usual.

In the example, both A and B have advantages and disadvantages, with neither being clearly better than the other. However, if only a sub-set of impact areas are addressed, this may change and depending on the selection, either option A (Figure 6) or option B (Figure 7) may appear to be more favourable.

5. Impact Areas: Impact Area Selection

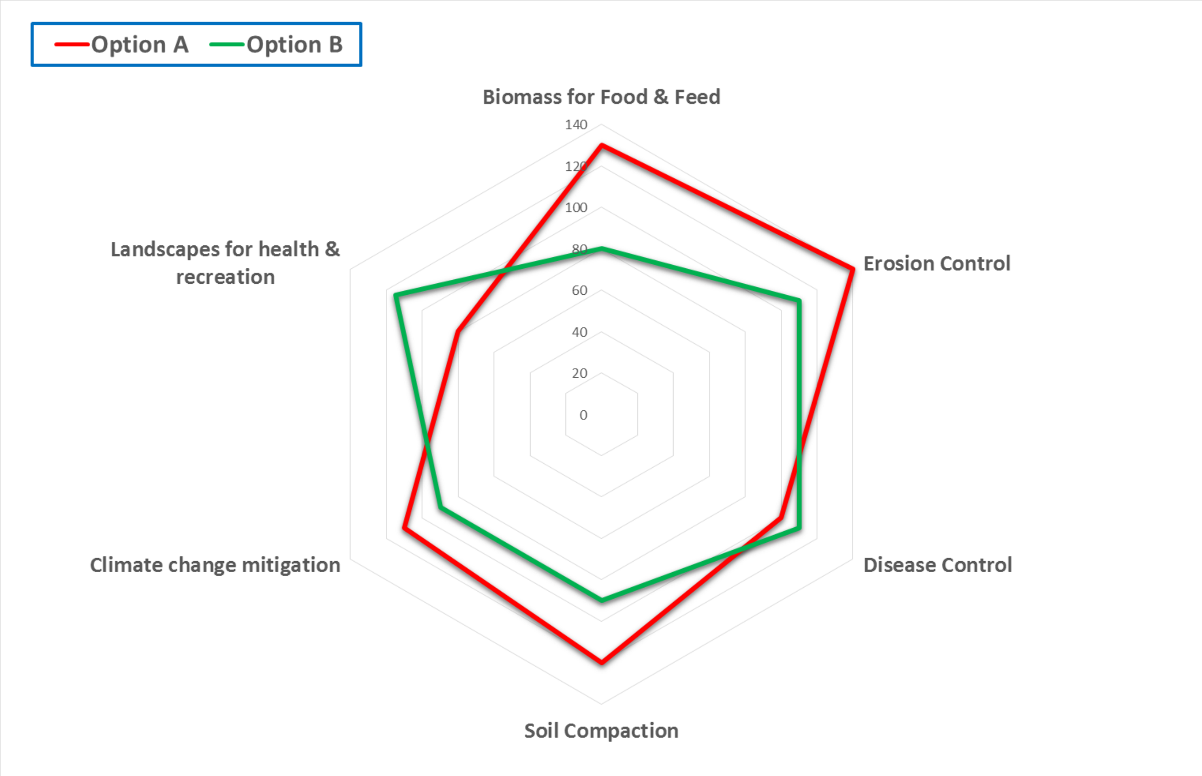

Figure 6: Two hypothetical management options (A and B) and their performance for 6 out of the 11 ecosystem services presented in Figure 5. Values represent percentages of service provision relative to business as usual. Option A appears to be more favourable than option B.

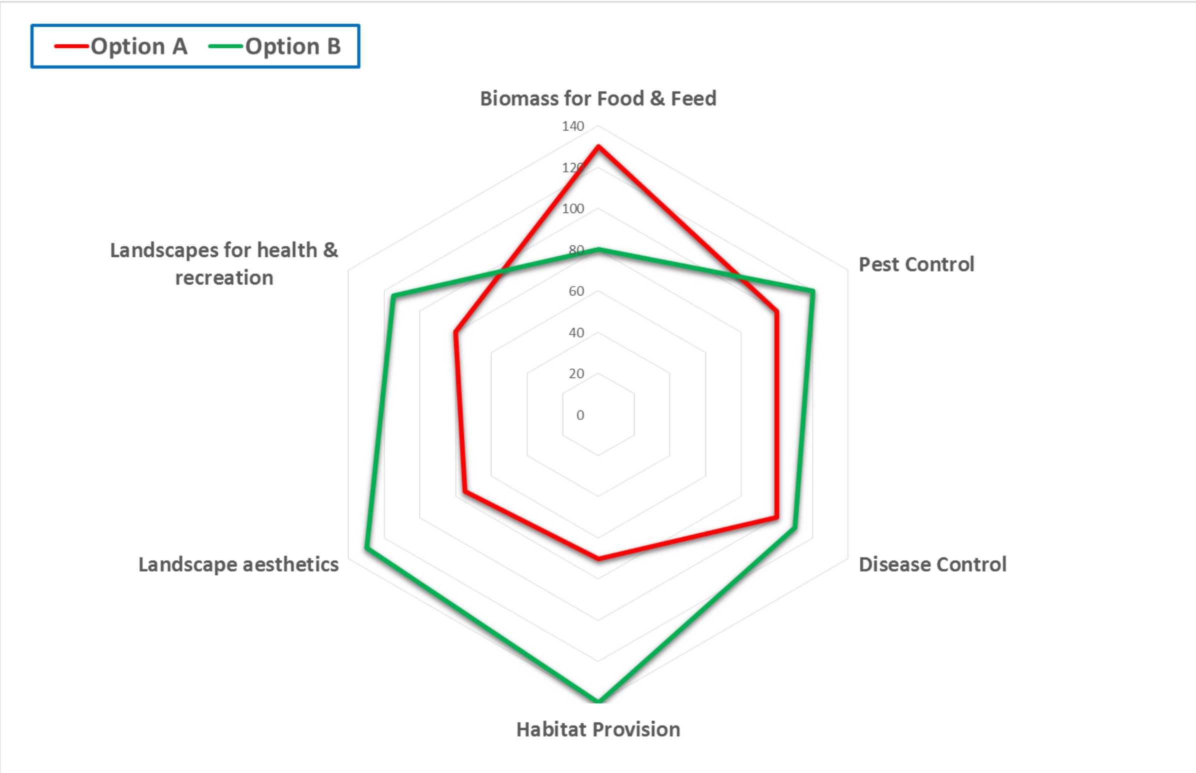

Figure 7: Two hypothetical management options (A and B) and their performance for 6 out of the 11 ecosystem services presented in Figure 5. Values represent percentages of service provision relative to business as usual. Option B appears to be more favourable than option A.

5. Impact Areas: Impact Area Selection

Not addressing all relevant impact areas will introduce a bias that makes the assessment results unreliable, even though all data generated may be state of the art. In a worst-case scenario, this may result in providing incomplete information to decision makers and lead to sub-optimal management or policy choices which miss opportunities or have unforeseen, negative consequences.

How can impact areas be selected systematically?

Impact areas likely to be affected by an option may be identified through a (short) literature review, expert consultation, or a combination of both approaches. Ideally, stakeholders are also involved in the process to add their perspective as to what impacts should be considered.

For including stakeholders' views, assessments may adapt an approach applied in sustainability reporting, which is called "*materiality analysis*" which is. It is used to identify topics for reporting that are material (i.e. relevant) and rank them according to their importance. Only the material topics are be reported and the ranking determines the amount of detail with which each topic is covered (GRI Standards, 2016). For a materiality analysis, the Global Reporting Initiative recommends to create a matrix, where one axis represents the expected seriousness of economic, ecological or social impacts and the other axis represents the importance of the respective impact area for stakeholders. Impact areas that score high on either of the axes are considered material (GRI Standards, 2016).

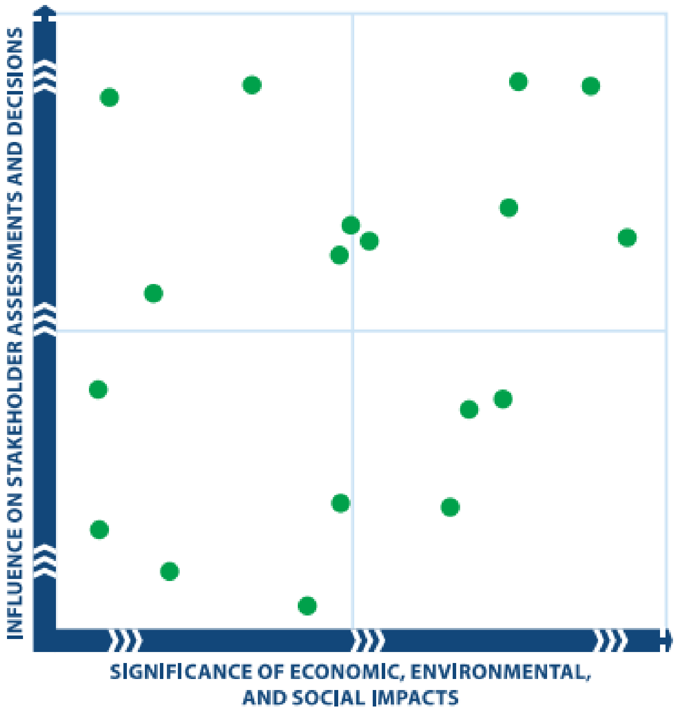

Figure 8: GRI materiality matrix (GRI Standards, 2016)

As part of the GRI standards, there is no fixed requirement on how the importance for stakeholders should be assessed. Consultations and workshops are possible, as are in-house assessments by the reporting organizations. In the context of impact assessments, this handbook recommends direct interaction with a well documented and balanced selection of stakeholders. Where this is not possible, an analysis of policy strategy documents to identify relevant impact areas may be used as an alternative (Hermanns et al., 2017), though both approaches should ideally complement each other.

A materiality analysis always needs to be performed for a specific assessment, because the impact areas identified and the prioritization derived are based on the socio-economic and geophysical conditions at a certain point in time and space. Therefore, results of materiality analyses cannot be readily transferred to similar assessments at other locations. Even at the same location, the relevance of impact areas and their respective priorities may change over times due to the dynamics of norms and values, as well as due to advances in knowledge.

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5. Impact Areas: Assessment Perspectives

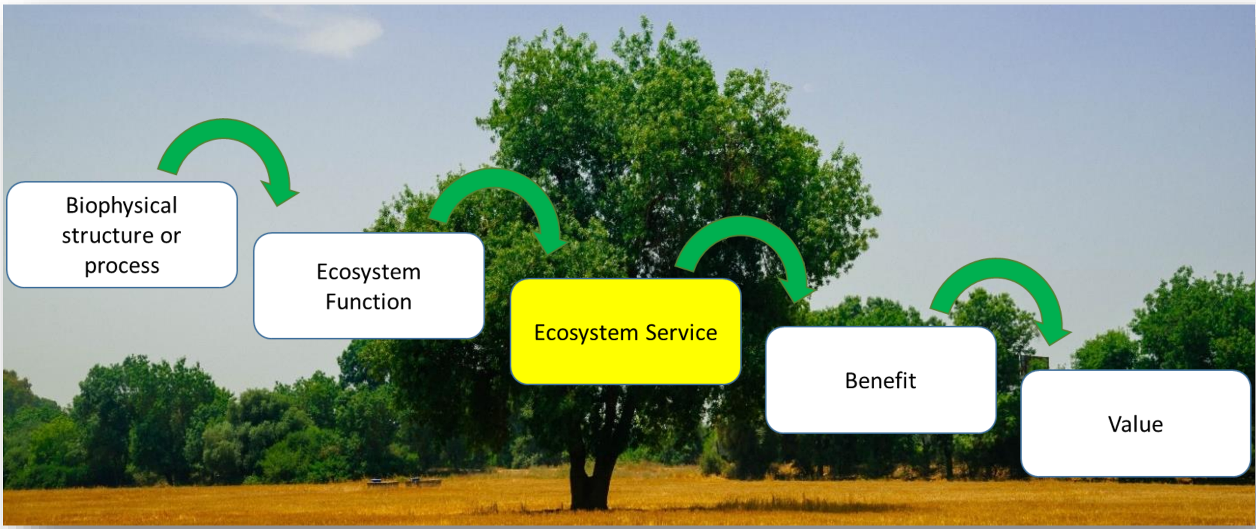
5.2 Assessment Perspectives

Info-Box

In impact assessments, effects of management or policy options on societal targets are analysed. Because the number of societal targets and of potential impacts is too high to be investigated in a single assessment, some restriction or **focus** is required. **Assessment perspectives are a thematic focus** that can be used to make assessments operable. For example, assessments could choose to only analyse effects related to animal welfare or to economic performance. More than one perspective may be chosen for an impact assessment, each with their own impact areas.

The BonaRes Assessment Platform currently supports the two perspectives: **Ecosystem Services** and **Resource Use Efficiency**. Information on impact areas and indicators relevant for these perspectives are provided. Both perspectives are largely complementary and highly relevant for agriculture in industrialised countries where yield gaps are low, and in particular for evaluating transformation processes from a fossil fuel based economy towards a bioeconomy based on renewable resources. Additionally, the platform highlights which impact areas and indicators from these perspectives are most relevant for targets defined under the United Nations' **Sustainable Development Goals**.

Obviously other perspectives, such as impacts on human health, equity or food security are also highly important. They may be added with future updates of the platform.



5.2.1 Ecosystem Services

Info-Box

The concept of ecosystem services aims at demonstrating the value of ecosystems for human societies. Multiple ecosystem services are based on soils. They are particularly relevant for targets of a sustainable bioeconomy and for the internationally agreed Sustainable Development Goals (**SDG**). However, the operationalisation of linkages between soil management, soil functions and ecosystem services remains a challenge. The BonaRes Assessment Platform addresses this by supporting impact assessments from the perspective of soil related ecosystem services. It provides methods for designing assessments and offers guidance for selecting, applying and linking indicators related to ecosystems services.

What are soil related ecosystem services?

The concept of ecosystem services aims at demonstrating the value of nature for human societies (Costanza et al., 1997, 2017; MEA, 2005; Haines-Young & Potschin, 2013; Potschin-Young et al., 2017; TEEB 2010). Ecosystem services can be seen as arising from the interaction of biotic and abiotic processes, and refer specifically to the 'final' outputs of ecological systems, i.e., the goods and services directly consumed or used by people (Haines-Young & Potschin, 2013; Potschin-Young et al., 2017). According to the **cascade model**, first introduced by Haines-Young & Potschin (2010), ecosystem services are based on biophysical structures or processes and the functions they serve. From ecosystem services humans generate benefits and values. Figure 9 shows the full cascade, starting from a biophysical structure or process and leading to values for humans. The figure also provides a soil related example.

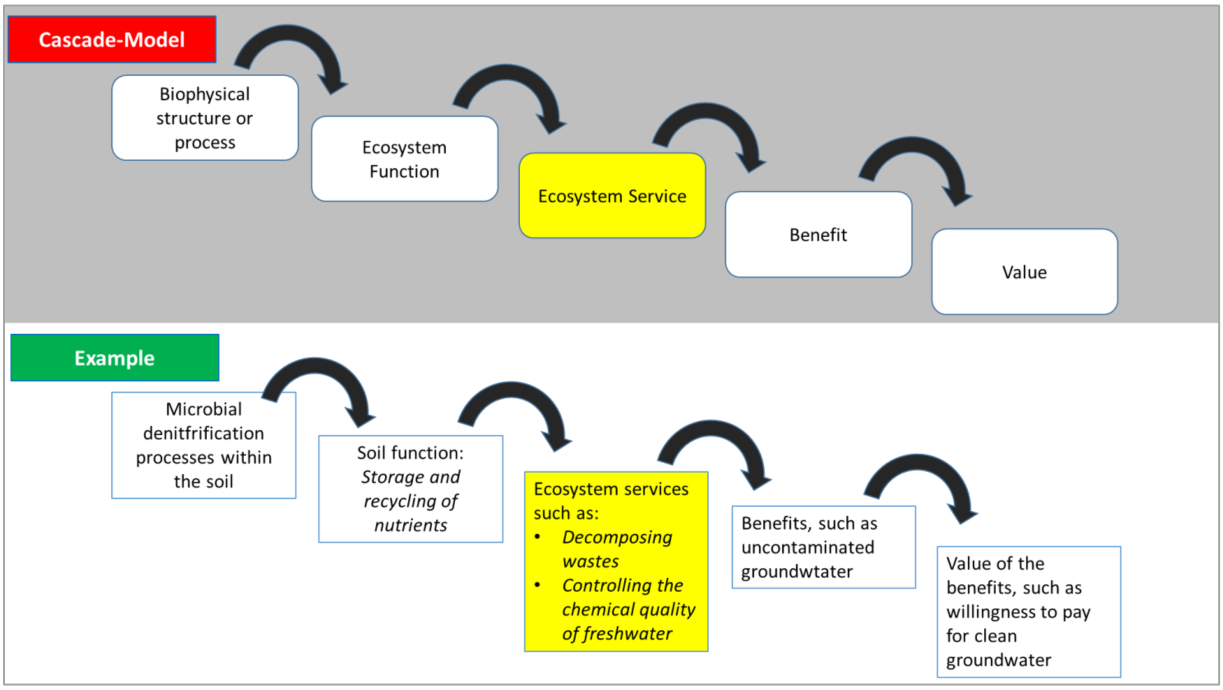


Figure 9: Ecosystem service cascade model and example for a soil related cascade. Own representation based on Potschin & Haines-Young (2011)

Defining which ecosystem services are provided by soils or to what degree soils contribute to specific ecosystem services is almost impossible, because most services are generated by the interaction of multiple ecosystem components. For example, soils allow cereals like wheat to grow and to produce food for humans. However, the wheat plant itself is not part of the soil, and neither are rainfall, sunlight or human inputs such as seedbed preparation and sowing. Only the combination of these factors creates the ecosystem service “provision of food”. For this reason, it is more practical to refer to **soil related ecosystem services**. In the context of impact assessments, we consider those ecosystem services to be soil related that are either **based exclusively** on soil functions, or where soil functions ***make a significant contribution*** to the final service. An example for services from the first category is the purification of water during its passage through the soil matrix, examples for services from the second category are the aboveground habitats created by plants growing in the soil. Both types of soil related ecosystem services have in common that they are affected by agricultural management.

The role of soils within the ecosystem services concept has been addressed in a number of recent publications (Stavi et al., 2016; Schwilch et al., 2016; Keesstra et al., 2016; Baveye et al., 2016; Adhikari & Hartemink, 2016; Helming et al., 2018) supplementing earlier studies on distinct aspects of soil functions and soil related ecosystem services (Blum, 1993; Daily et al., 1997; Robinson et al., 2009; Dominati et al., 2010; Bennett et al., 2010; Breure et al., 2012; Pulleman et al., 2012; Bouma, 2014; Schulte et al., 2014). The following figure illustrates how different categories of soil related ecosystem services are based on soil functions.

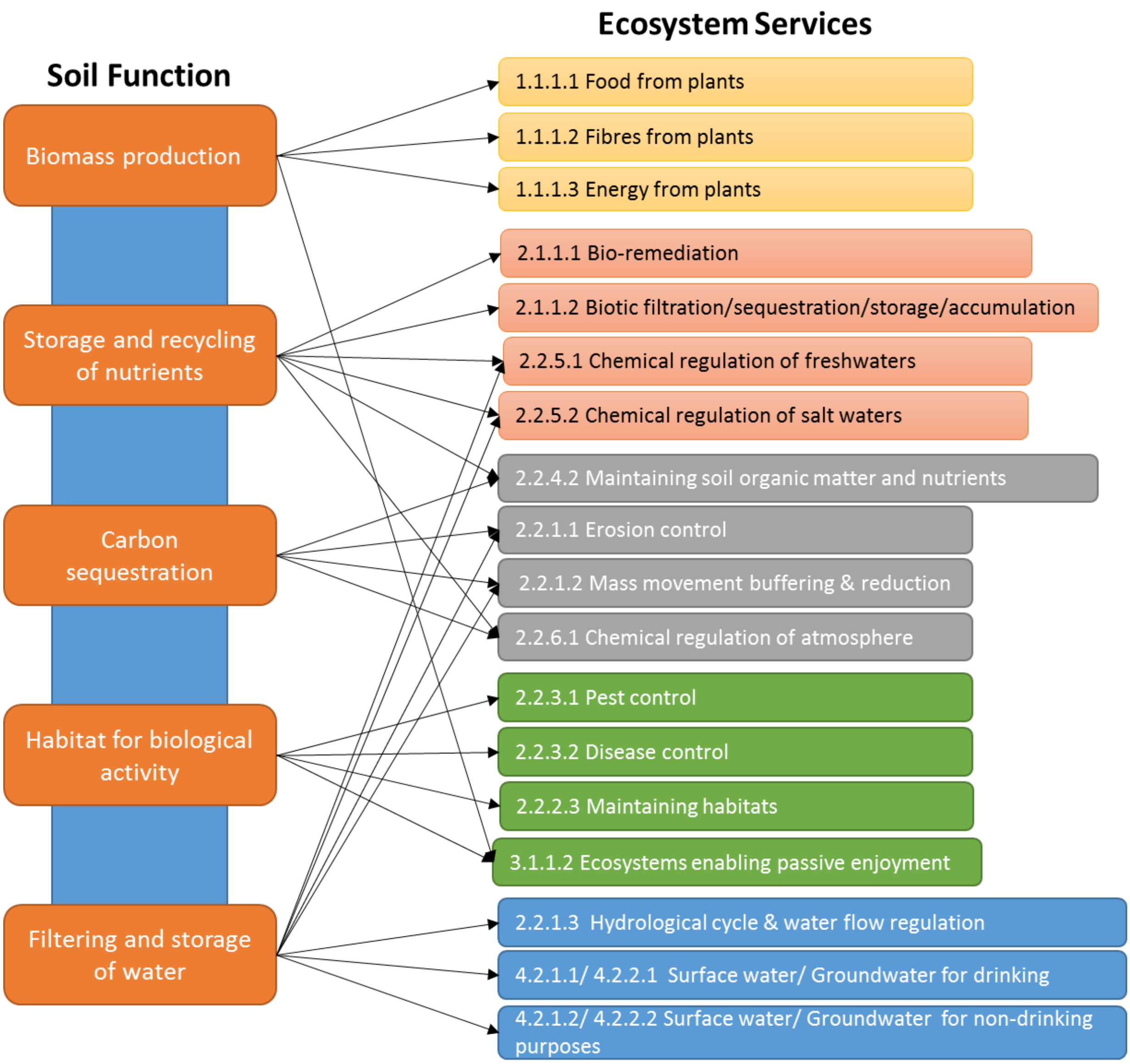


Figure 10: Connection between soil functions and selected soil related ecosystem services. Please note that all soil function are connected to and influencing each other.

Why are ecosystem services relevant for soil management?

The concept of ecosystem services has been proven very useful in policy support to demonstrate and assess human-environment interactions (Helming et al., 2013). Soil related ecosystem services are fundamental to achieving targets of a sustainable bioeconomy and of the internationally agreed Sustainable Development Goals (SDG (General Assembly, 2015). For agricultural soils, management determines the degree by which different services are provided. **No management exists that can maximize all ecosystem services at the same time**. Farmers must make decisions on trade-offs between different services, wherein trade-offs between provisioning services (such as *provision of food, feed and fibre*) and regulating and maintenance services (such as *provision of habitats* or *chemical quality of freshwaters*) are the most typical. For example, intensive management with high applications of fertilizers and pesticides will maximize yields and the ecosystem service: *provision of food*. On the other hand, the ecosystem service: *provision of habitats* for insects and birds will be reduced and the ecosystem service: *chemical quality of freshwaters* (groundwater or rivers) may be compromised by seepage.

How can ecosystem services be considered in assessments of soil management?

Impact assessments need to account for the fact that agricultural management always implies trade-offs between different ecosystem services. Each management option will enhance the provision of some ecosystem services while reducing the provision of others. **Where impact assessments only consider a selected number of services, this selection may therefore already determine the assessment results**. Depending on which ecosystem services are considered, assessments will find different policy options to have the most positive impacts. Sound research design is required to safeguard against such arbitrariness and potential for manipulation (see chapter on *Impact Area Selection*), and to instead highlight the trade-offs between options.

For the assessment of ecosystem services, the BonaRes Assessment Platform uses the **Common International Classification of Ecosystem Services (CICES)** (Haines-Young & Potschin, 2018) which provides a complete and standardized classification scheme designed to support accounting of ecosystem services. It was developed in collaboration with the **European Environment Agency (EEA)** and builds on earlier classifications as part of *The Economics of Ecosystems and Biodiversity* (TEEB, 2010) and the *Millennium Ecosystem Assessment* (MEA, 2005). CICES makes a substantial contribution to a standardisation of ecosystem service assessments and links up with efforts to integrate ecosystem services into national and European accounting systems (Edens and Hein, 2013).

CICES has a strictly hierarchical, five-level structure, organized from bottom to top into: section, division, group, class and class type. Three sections provided by the biotic components of ecosystems are distinguished: **Provisioning**, **Regulation & Maintenance** and **Cultural**. In an extension introduced in 2018 with CICES version 5.1, abiotic sections for Provisioning, Regulation & Maintenance and Cultural services were added. Figure 11 shows how the production of cereals as food crops fits into the CICES hierarchy.

5. Impact Areas: Assessment Perspectives - Ecosystem Services

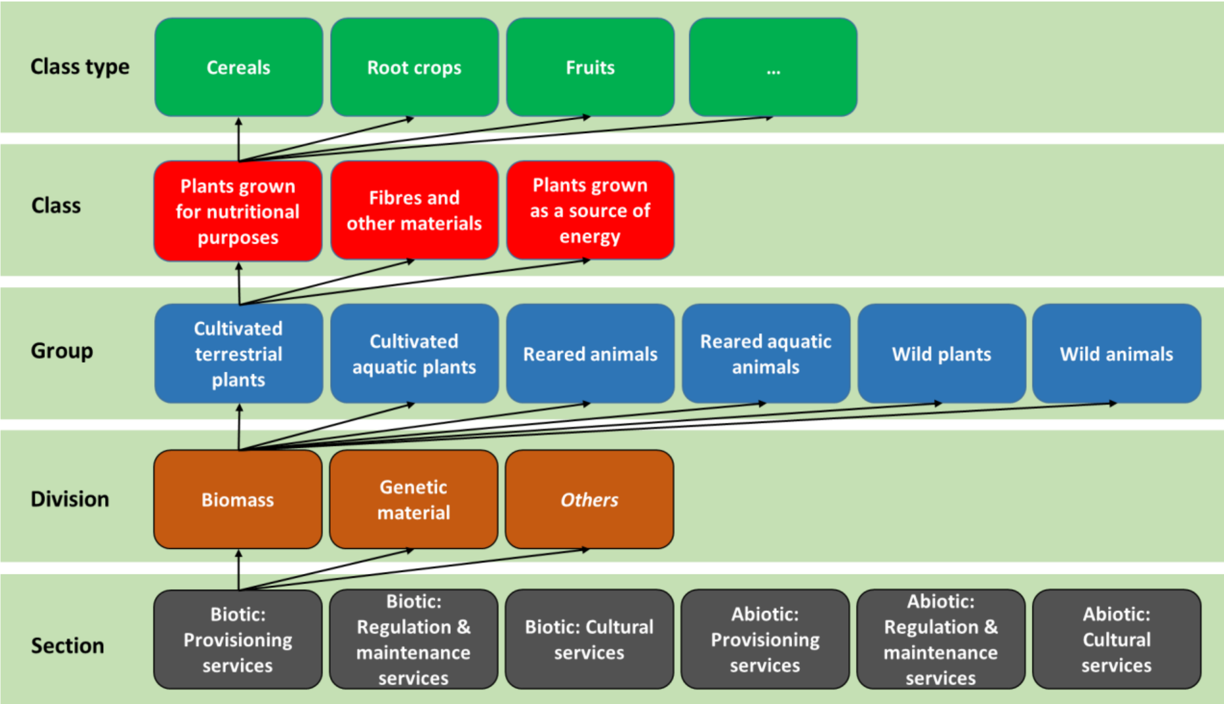


Figure 11: CICES 5.1 hierarchy from section to class type: Example of cereals produced as food crops.

The hierarchical structure is intended to facilitate ecosystem service accounting at different spatial scales. **The BonaRes Assessment Platform supports assessments at the class level**, because the higher hierarchical levels lack the detail usually required for meaningful investigations into the effects of agricultural management. However, where data availability is insufficient, users are encouraged to move to group level for individual categories. An example of such a strategy is provided by Maes et al. (2016).

In total, there are 84 ecosystem service classes in CICES. In a preliminary assessment conducted by the author, half of them (42) were considered to be related to agricultural soils and their management (Tables 2 and 3). These service classes are mainly from the biotic regulating and maintenance section. In the context of the BonaRes Assessment Platform, soil related ecosystem service classes are considered potential **impact areas.** For each of them**,** a description**,** lists of indicators, information on strength and weaknesses associated with their measurement, and references to examples in published research are provided. Additionally, the platform offers guidelines for selecting impact areas suited to requirements of specific assessment. Though multiple correlations exist between ecosystem services, these correlations are very difficult to account for in the design of measurements. Published studies generally assess ecosystem services individually and only address correlations between them as part of their results and discussion. This approach is also followed in the methods presented on the BonaRes Assessment Platform.

The CICES classes are not an ideal fit for the context of soil related impact assessments. Some classes are very broad (e.g., *maintaining soil organic matter and nutrients*), some are very specific (e.g., *biomass for food & feed*, *biomass for material use*, *biomass for energy*) some overlap (e.g., bio*-remediation of wastes, maintaining soil organic matter and nutrients, chemical regulation of freshwaters)* and some important soil physical categories such as aggregate stability or resistance against compression are not accounted for. However, CICES offers a comprehensive standard for ecosystem services assessment, thereby increasing policy relevance and comparability of study results.

In spite of these shortcomings in the current CICES version, the BonaRes Assessment Platform therefore builds on the CICES classification and aims at facilitating its use in soil related assessments.

Please note: As is the case for all assessment perspectives, the ecosystem services concept only addresses a limited set of societal targets while ignoring targets from other valuation systems, such as resource use efficiency, equity or health (Sandifer et al., 2015; Schröter et al., 2017, Helming et al., 2018). Using more than one perspective will provide a more comprehensive picture of (potential) impacts. Though still uncommon, a number of soil related studies have recently shown how linkages to other valuation systems are possible, e.g., as outlined by Bouma (2014) and Keesstra et al. (2016). For this reason, the BonaRes Assessment Platform also supports the two complementary perspectives of **Ecosystem Services** and **Resource Use Efficiency** and encourages researchers to use them both.

5. Impact Areas: Assessment Perspectives - Ecosystem Services

Table 2: CICES V5.1 (Common International Classification of Ecosystem Services) biotic ecosystem service classes. Class names were simplified by us to allow for a better overview. Classes considered relevant for the assessment of soil management and soil functions changes are marked in black.

No.	Provisioning	Regulation & Maintenance	Cultural
1	**Food from plants**	**Bio-remediation**	**Ecosystems enabling active enjoyment**
2	**Fibres from plants**	**Filtration/sequestration/storage/accumulation**	**Ecosystems enabling passive enjoyment**
3	**Energy from plants**	**Smell reduction**	**Ecosystems enabling research**
4	Food from plants (aquaculture)	**Noise reduction**	**Ecosystems enabling education**
5	Fibre from plants (aquaculture)	**Visual screening**	**Culturally/historically important ecosystems**
6	Energy from plants (aquaculture)	**Erosion control**	**Aesthetically important ecosystems**
7	Food from animals (terrestrial)	**Mass movement buffering & reduction**	**Symbolically important ecosystems**
8	Fibre from animals (terrestrial)	**Hydrological cycle & flow regulation**	Ecosystems with sacred/religious meaning
9	Energy from animals (terrestrial)	**Wind protection**	**Ecosystems used for entertainment/representation**
10	Food from animals (aquaculture)	Fire protection	**Ecosystems with existence value**
11	Fibre from animals (aquaculture)	**Pollination**	**Ecosystems with option/bequest value**
12	Energy from animals (aquaculture)	**Seed dispersal**	Other - Cultural (Biotic)
13	Food from wild plants	**Maintaining habitats**	
14	Fibre from wild plants	**Pest control**	
15	Energy from wild plants	**Disease control**	
16	Food from wild animals	**Weathering & soil development**	
17	Fibre from wild animals	**Maintaining soil organic matter and nutrients**	
18	Energy from wild animals	**Chemical regulation of freshwaters**	
19	**Seeds for breeding purposes**	**Chemical regulation of salt waters**	
20	**Plants for breeding purposes**	**Chemical regulation of atmosphere**	
21	Genetic material from wild plants	**Regulation of temperature/humidity/ ventilation/transpiration**	
22	Collection of animal material	Other - Regulation & Maintenance (Biotic)	
23	Breeding with wild animals		
24	Genetic material from wild animals		
25	Other - Provisioning (Biotic)		

Table 3: CICES V5.1 (Common International Classification of Ecosystem Services) abiotic ecosystem service classes. Class names were simplified by us to allow for a better overview. Classes considered relevant for the assessment of soil management and soil functions changes are marked in black.

No.	Abiotic		
	Provisioning	**Regulation & Maintenance**	**Cultural**
1	**Surface water for drinking**	Dilution by water bodies	Physical nat. features enabling active/passive interaction
2	**Surface water for non-drinking purposes**	Dilution by atmosphere	Physical nat. features enabling intellectual interactions
3	Freshwater for energy	**Abiotic mediation by filtration/ sequestration/ storage or accumulation**	Physical nat. features enabling spiritual/symbolic interactions
4	Seawater for energy	Mediation of nuisances by abiotic structures/processes	Physical nat. features with existence/option/bequest value
5	**Groundwater for drinking**	Physical barriers to mass flows	Other - Cultural (Abiotic)
6	**Groundwater for non-drinking purposes**	**Physical barriers to liquid flows**	
7	Groundwater for energy	Physical barriers to gaseous flows	
8	Other aqueous- Provisioning (Abiotic)	**Maintenance and regulation by inorganic processes**	
9	Food from minerals	Other - Regulation & Maintenance (Abiotic)	
10	Materials from minerals		
11	Energy from minerals		
12	Food (contribution) from non-minerals		
13	Materials from non-minerals		
14	Wind energy		
15	Solar energy		
16	Geothermal		
17	Other non-aqueous - Provisioning (Abiotic)		

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5. Impact Areas: Assessment Perspectives - Resource Use Efficiency

5.2.2 Resource Use Efficiency

Info-Box
In the context of agricultural soil management we define resource use efficiency as the ratio between benefits generated by agricultural production processes and the amount of stressed resources used. Stressed resources are resources for which competing demands or resource use conflicts exist. This can be due to the fact that a resource is scarce (such as land, water or money) or because its use conflicts with other societal targets (such as pesticide application with biodiversity conservation). Increasing agricultural resource use efficiency is a target at the global, European and German national policy level. It is central to achieving a highly productive agricultural sector that minimises harmful externalities. Agriculture is characterised by the production of multiple benefits while utilising multiple resources. A comprehensive assessment therefore requires the appraisal of several efficiency indicators.

What is resource use efficiency?

In the context of soil management we define resource use efficiency as the ratio between benefits generated by agricultural production processes and the amount of resources used. We distinguish between stressed resources for which there is competing demand and abundant resources for which there is no competition. Competition is context-specific and can be due to demand exceeding resource limitations or due to conflicts between resource use and other societal targets (e.g. pesticide application conflicting with biodiversity preservation or emissions of greenhouse gasses conflicting with climate change mitigation). Only the amount of stressed resources is considered in efficiency assessments (di Maio et al., 2017).

$$\textbf{Resource Use Efficiency (RUE)} = \frac{\text{Benefits Generated}}{\text{Stressed Resource Used}}$$

Eq.2: Definition of resource use efficiency

5. Impact Areas: Assessment Perspectives - Resource Use Efficiency

As benefit, any quantifiable, positive characteristic of a product can be chosen. This choice exerts a strong influence on calculated efficiencies and often determines efficiency rankings between production alternatives. For example, if weight of harvested biomass is considered, growing strawberries is less efficient than growing ryegrass whereas the situation is reversed if the evaluation is based on gross profits.

As resource, any of the resources used in the production process can be chosen. Here, the system boundaries must be clearly defined because substantial resource consumption can occur in pre- or post-processes related to the process under investigation. For example, if energy efficiency is assessed, it is relevant whether or not energy used in the production of inputs or machinery is considered. Finally, leakage effects, indirect land use changes and especially rebound effects should be accounted for (Lambin & Meyfroidt, 2011).

The following figure illustrates how different categories (impact areas) of soil related resource use efficiency are based on soil functions.

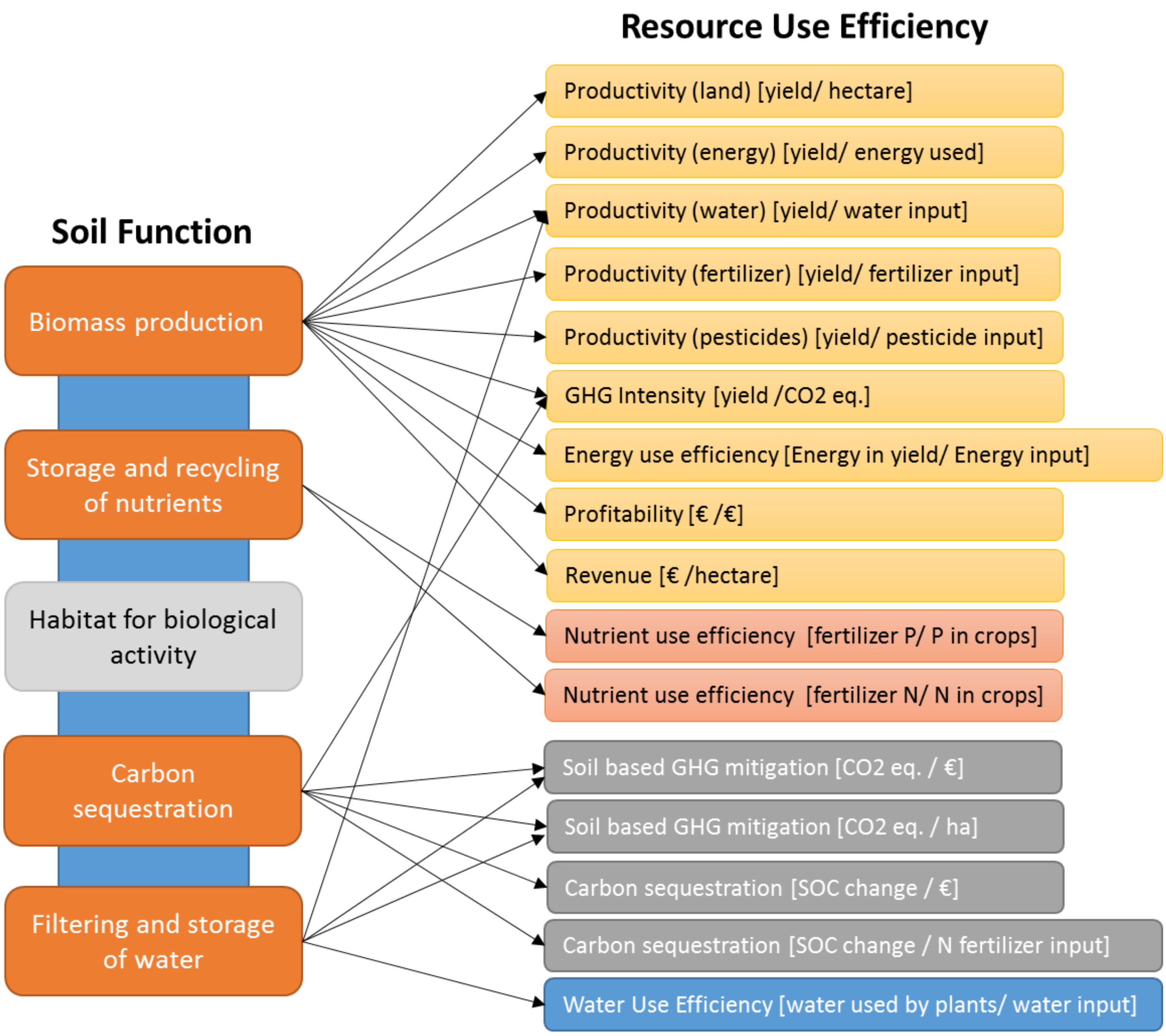

Figure 12: Connection between soil functions and selected resource use efficiency categories relevant for agricultural soil management. Please note that all soil function are connected to and influencing each other.

5. Impact Areas: Assessment Perspectives - Resource Use Efficiency

Why is resource use efficiency relevant for soil management?

Increasing agricultural resource use efficiency is a target at many policy levels.

- **Globally**, the Sustainable Development Goal (SDG) 12 "sustainable consumption and production" target 12.2 seeks to achieve sustainable management and efficient use of natural resources by 2030 (General Assembly, 2015).
- **At the European level**, a 20% reduction of resource inputs within the food chain has been set as a milestone for 2020 (European Commission, 2011). This is to be achieved through measures targeting production, distribution and consumption. Increasing efficiency of agricultural production is also a priority within the EU Rural Development Act (European Union, 2013; Spicka, 2015) in which energy efficiency, greenhouse gas emissions per unit produced and water use efficiency are explicitly named.
- **At German national level**, the 2014 Policy Strategy on Bioeconomy seeks to improve efficiency of agricultural production by increasing productivity while protecting natural resources and minimising greenhouse gas emissions (BMEL 2014).

Farmers have always sought to optimize their efficiency in the use of limited resources. The most obvious example are efforts to optimize crop yields or revenues per hectare of land. The use of efficiency indicators in this regard is very common. However, increased awareness of the environmental consequences of resource use have extended the list of stressed resources. Today, there is a growing interest to achieve good yields with as little use of pesticides, fertilizer or greenhouse gas emissions as possible. Policy related assessment that do not focus on the individual farm or field but operate at higher spatial scales may also need to consider additional benefits, such as calories produced in a region or energy content of harvest. This results in the need for a whole suit of efficiency indicators.

So far, only few assessments of agricultural resource use efficiency documented in scientific literature *explicitly* address the role of soils. However, due to the paramount role of soils for crop growth, assessment results often *implicitly* reflect changes in soil functions. In the context of research to support the transition from a fossil fuel based economy towards a sustainable bioeconomy, it is necessary to determine to what degree agricultural management can increase resource use efficiencies by targeting soil functions and to what degree efficiencies can be increased irrespective of soils.

Resource use efficiency is central to achieving a highly productive agricultural sector while minimising harmful externalities. However, increasing efficiencies may also lead to trade-offs. For example, reducing the amount of labour required to produce a fixed amount of biomass may have negative implications for job provision, nutrient use efficiencies greater than 100% are indicative of nutrient mining by harvest (Scholz & Wellmer, 2015), and maximising the share of biomass harvested relative to net primary production (HANPP) conflicts with biodiversity targets (BIO Intelligence Service, 2012). The potential for such trade-offs should be considered in assessments of resource use efficiency.

How can resource use efficiency of soil management be measured?

The resource use efficiency of agricultural soil management can be assessed by means of efficiency indicators, calculated as the ratio between the amount of benefit generated and the amount of resources used (see Equation 2 above). What type of efficiency indicator is assessed, i.e. the selection of which benefits and which resource use are to be compared, should be based on the intended result of the agricultural management, its main inputs, and the characteristics of management alternatives. More than one efficiency indicator may be required for a comprehensive assessment or to show trade-offs between different types of efficiency.

Temporal scale: Inter-annual yield variability is typical for agriculture. The use of multi-year averages for benefits and resources therefore results in indicators that are more representative of the management systems than single-year variants. Integrating production over several years is also necessary to account for perennial management effects, crop rotations and pre-crops effects (Preissel et al., 2015; Zhang et al., 2017). Where crop rotations of different length are to be compared, normalization can be achieved by dividing total benefits and resource use by the number of years in each rotation (Reckling et al., 2016).

Spatial scale: The spatial scale most suited to the analysis of resource use efficiency depends on the purpose and on the users of the assessment results. Farm or field level efficiency is of particular interest to practitioners, especially where they affect economic performance. Typical efficiency indicators at this scale are *productivity* (Alam et al., 2017; Moreau et al., 2012; Zhang et al., 2016), *water use efficiency* (WUE) (Pascual et al., 2016; Wei et al., 2016; Zhao et al., 2010), *benefit cost ratio* (BCR) (Alam et al., 2017; Rehman et al., 2011), *nutrient use efficiency* (NUE) (nitrogen: Buckley et al., 2016; Gu et al., 2017; phosphorous: Gerber et al., 2014; Zhang et al., 2016) *and energy use efficiency* (Hill et al., 2006; Arodudu et al., 2017).

Assessments with landscape or global implications are particularly relevant for policy makers. Indicators include *greenhouse gas emissions per yield* (Khakbazan et al., 2017; Steyn et al., 2016) or *GDP contribution per domestic material consumption* (European Commission, 2011; European Environment Agency, 2016). Efficiency indicators can also be tailored to assist the assessment of specific research or policy questions. Examples are efficiencies for job provision (jobs generated per hectare of farmland by different biomass usages) (BMEL, 2014) or economic risk efficiency (how does irrigation affect the amount of revenue that can be expected under a defined probability level) (Meyer-Aurich et al., 2016).

Tables 4 and 5 give an overview about recent studies on resource use efficiency in agricultural and on the efficiency indicators used.

Combining multiple benefits or resources

Agriculture is characterised by the production of multiple benefits such as food, feed, fibre or fuel while using multiple resources like land, water, energy and nutrients. It also generates profits, provides employment and creates habitats for plant and animal species while requiring capital and labour. For a comprehensive efficiency assessment, the appraisal of several benefits and resource uses is therefore essential. For this, assessments can either use multiple efficiency indicators, or apply a single indicator in which multiple benefits or resources are combined.

5. Impact Areas: Assessment Perspectives - Resource Use Efficiency

The most common method to create combined indicators is to express all benefits or resources in monetary terms, based on their explicit or implicit market value. While practical, this approach is often criticised due to the difficulties of reflecting environmental and social consequences in monetary terms, and because values attributed to non-marketed goods vary strongly between assessments (Baveye, 2016).

Alternative methods are often based on benchmarking. For example, Lin & Hülsbergen (2017) determine land use efficiency for a crop rotation by first calculating multiple efficiency indicators for the rotation (such as dry matter per hectare, lipid production per hectare). In a second step, each of these indicators is divided by the respective efficiency of average production in the region. This creates a set of dimensionless values. Finally, the average of these values is calculated and used as the combined indicator. Another benchmarking approach derived from economics is data envelopment analysis (DEA). Linear combinations of all enterprises or processes within in an assessment are used to calculate combinations with the highest (positive) output and smallest use of resources. These combinations constitute the so called efficiency frontier, for which an efficiency of 100% is assumed. Efficiency values of individual enterprises or processes are calculated relative to this frontier. All types of efficiencies can be combined in DEA. For example, Hoang & Alauddin (2010) used data envelopment analysis to create efficiency scores for 30 OECD countries based on agricultural production, costs, use of fertiliser and use of energy. Masuda (2016) calculated eco-efficiency of Japanese wheat production based on global warming potential, aquatic eutrophication potential and yield.

While combined efficiency indicators allow for a quick overview and may facilitate monitoring and communication with stakeholders and policy makers, the selection and weighing of the included efficiency categories and the interpretation of the resulting indicators is challenging. In detailed impact assessment, the use of multiple efficiency indicators may be preferable because they make trade-offs between different efficiency categories directly visible whereas such relationships may be obscured in the values of combined indicators.

Please note: Efficiency assessments are well suited to assess performance of agricultural production within a limited number of categories. However, they are unable to provide a holistic view of ecosystem interactions. In this regard, assessment of ecosystem services provides a well suited complementary perspective.

5. Impact Areas: Assessment Perspectives - Resource Use Efficiency

Table 4: Non-integrated indicators for measuring efficiency in agriculture. Overview of studies published between 2008 and 2017 based on Web of Science Core Collection, applying search terms: "efficiency" (title) and "agriculture", "indicator" (topic). Indicators used in three or more studies are highlighted in red.

	Seperate Indicators					
Benefit → / Resource ↓	**Grain/fruit/ tuber weight**	**Aboveground biomass**	**Embodied nitrogen**	**Embodied energy**	**Revenue**	**Sequestered carbon**
Agricultural land	Moreau et al., 2012 Phengphaengsy & Okudaira, 2008 Rehman et al., 2011 Steyn et al., 2016 Zhao et al., 2010			Lin et al., 2017	Baumane et al., 2014 Li et al., 2013 Miklovicova & Miklovicova, 2012 Moreau et al., 2012 Rehman et al., 2011 Spicka, 2015	
Energy	Alluvione et al., 2011 Khakbazan et al., 2017			Alluvione et al., 2011 Deike et al., 2008a Deike et al., 2008b Giambalvo et al., 2009 Khakbazan et al., 2017 Lin et al., 2017 Romanelli et al., 2012 Wasiak & Orynycz, 2015		
GHG emissions	Khakbazan et al., 2017 Steyn et al., 2016			Khakbazan et al., 2017		
Water	Pascual et al., 2016 Steyn et al., 2016 Wei et al., 2016 Zhao et al., 2010	Pascual et al., 2016			Phengphaengsy & Okudaira, 2008	
Human labour					Li et al., 2013	
Fertiliser - N	Steyn et al., 2016 Rehman et al., 2011		Buckley et al., 2016 Godinot et al., 2014 Godinot et al., 2016 Gu et al., 2017 Moreau et al., 2012 Rehman et al., 2011	Clark & Tilman, 2017		
Fertiliser - P	Korkmaz et al., 2009 Steyn et al., 2016					
Pestcides					van Lierde et al., 2009	
Money (Costs)					Li et al., 2013 Miklovicova & Miklovicova, 2012 Rehman et al., 2011	Khakbazan et al., 2017

5. Impact Areas: Assessment Perspectives - Resource Use Efficiency

Table 5: Integrated indicators for measuring efficiency in agriculture. Overview of studies published between 2008 and 2017 based on Web of Science Core Collection, applying search terms: "efficiency" (title) and "agriculture", "indicator" (topic).

Integrated Indicators		
	Stochastic Frontier Analysis (SFA)	**Data Envelopment Analysis (DEA)**
Efficiency scores based on multiple inputs and outputs	Boshrabadi et al., 2008 Li et al., 2013	Aldanondo-Ocha et al., 2014 Azad & Ancev 2014 Bojnec & Latruffe, 2011 Gomes et al., 2009 Hoang, 2011 Hoang & Alauddin, 2012 Lee & Park 2017 Masuda, 2016 Pagotto & Halog, 2015 Sabiha et al., 2017 Spicka, 2014 Spicka & Smutka, 2014 Wang et al., 2015

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5.2.3 Sustainable Development Goals (SDG)

Info-Box

The Sustainable Development Goals (SDG) are a set of goals and targets to be reached by 2030, agreed upon by, and requiring action from all United Nations (UN) member states. Soil functions play an important role for the realisation of multiple targets and underpin the achievement of the SDG agenda as a whole. Although the UN indicators for measuring progress on these targets are designed at the global level, successful achievement of the agenda requires national, regional and local efforts. It is therefore important to also connect the sustainable development goals to effects of policy and management options at these levels. The BonaRes Assessment Platform supports the operationalisation of the SDG targets by defining linkages to soil related ecosystem services and agricultural resource use efficiency.

What are the Sustainable Development Goals?

The Sustainable Development Goals (SDG) were adopted in 2015 by the General Assembly of the United Nations (UN) as part of the 2030 Agenda for Sustainable Development. There are **17 SDG**s in total which include goals to end poverty and hunger, combat inequality and injustice, and tackle climate change by 2030. The SDGs are specified into 169 individual time-bound targets and there currently exist 230 indicators against which they can be measured. The goals are not binding under international law but UN member states have a legal obligation to issue progress reports up to the 2030 target date.

The SDGs are universal in nature meaning that they apply to and should be achieved in all countries. Furthermore, the 17 goals are “integrated and indivisible”, designed to balance the three dimensions of sustainable development: the economic, social and environmental (UN, 2016). Efforts to achieve one goal should not hinder progress in other areas.

5. Impact Areas: Assessment Perspectives - Sustainable Development Goals (SDG)

The SDGs build on the Millennium Development Goals (MDG) which were adopted by world leaders in 2000 to be reached by 2015. These were eight goals aimed primarily at reducing extreme poverty and the related consequences.

Why are the Sustainable Development Goals relevant in the context of soil management?

Despite only 4 explicit references to soils in the 169 SDG targets (targets 2.4, 3.9, 12.4, 15.3), soil functions play an important role for the realisation of a vast number of targets and underpin the achievement of the SDG agenda as a whole (Montanarella & Alva, 2015). Not only do they have a clear role in achieving SDGs related to the biophysical system (e.g. SDGs 2, 3, 6, 7, 12–15), they are also important to the achievement of socio-economic goals due to the multifunctional benefits they provide (Keesstra et al., 2016).

Unlike the MDGs which focussed on the donor recipient relationship between more developed and poorer countries, the SDGs are universal in nature, meaning that they apply to all countries and should be achieved in all countries. In Germany, Agenda 2030 forms the basis of the latest Sustainable Development Strategy (German Federal Government, 2017), which presents measures to implement the 17 SDGs across the economic, social and environmental spheres.

How can the SDGs be considered in assessments of soil management?

The UN indicators for measuring progress towards SDGs are designed for impact assessment at the global level and therefore do not directly relate to local, regional or even national decisions. However, successful achievement of the agenda requires efforts at every level in all UN member states. It is therefore important to draw connections between management options and the relevant SDG targets. The BonaRes Assessment Platform supports this by highlighting **25 SDG** targets that are directly connected to **agricultural management** and impact areas **under the assessment perspectives** of **soil related ecosystem services** and **agricultural resource use efficiency**. Table 6 lists the SDG targets for which connections are drawn and provides the codes for the respective impact areas, Table 7 lists the impact areas supported by the BonaRes Assessment Platform to which these codes refer to. The highest numbers of connections were found for target 2.04 (sustainable food production and resilient agriculture) and target 6.06 (protect and restore water related ecosystems).

On the platform, users may first select one of the SDG targets and then choose from a list of connected impact areas. A factsheet for the chosen impact area is then created which provides information on indicators, strengths & weaknesses pertaining to measurements, correlations with agricultural management and examples of published research.

Additional SDG targets and assessment perspectives, such as equity or human health impacts, may be added in future updates to the assessment platform.

5. Impact Areas: Assessment Perspectives - Sustainable Development Goals (SDG)

Table 6: SDG targets related to agricultural management and impact areas under the perspectives of resource use efficiency and ecosystem services.

SDG	Target	Description
1 NO POVERTY	01.02	By 2030, reduce at least by half the proportion of men, women and children of all ages living in poverty in all its dimensions according to national definitions. **Related Impact Areas**: 6, 11, 16, 22, 27, 32, 36, 41, 47, 52, 63, 64, 65, 66, 67, 83
1 NO POVERTY	01.05	By 2030, build the resilience of the poor and those in vulnerable situations and reduce their exposure and vulnerability to climate-related extreme events and other economic, social and environmental shocks and disasters. **Related Impact Areas:** 73, 74, 75, 76, 80, 81, 87, 103
2 ZERO HUNGER	02.04	By 2030, ensure sustainable food production systems and implement resilient agricultural practices that increase productivity and production, that help maintain ecosystems, that strengthen capacity for adaptation to climate change, extreme weather, drought, flooding and other disasters and that progressively improve land and soil quality. **Related Impact Areas:** 1, 2, 8, 9, 12, 13, 14, 15, 16, 17, 18, 19, 20, 21, 22, 23, 24, 25, 28, 29, 30, 31, 32, 33, 34, 35, 36, 37, 38, 39, 40, 41, 42, 43, 44, 49, 50, 54, 55, 56, 57, 58, 59, 60, 61, 62, 63, 64, 65, 66, 67, 68, 69, 73, 75, 77, 78, 79, 80, 81, 83, 84, 85, 86, 102, 103, 104
2 ZERO HUNGER	02.05	By 2020, maintain the genetic diversity of seeds, cultivated plants and farmed and domesticated animals and their related wild species, including through soundly managed and diversified seed and plant banks at the national, regional and international levels, and promote access to and fair and equitable sharing of benefits arising from the utilization of genetic resources and associated traditional knowledge, as internationally agreed. **Related Impact Areas:** 66, 67
3 GOOD HEALTH AND WELL-BEING	03.09	By 2030, substantially reduce the number of deaths and illnesses from hazardous chemicals and air, water and soil pollution and contamination. **Related Impact Areas:** 37, 38, 39, 40, 41, 42, 61, 68, 69, 84, 85, 102, 86
4 QUALITY EDUCATION	04.07	By 2030, ensure that all learners acquire the knowledge and skills needed to promote sustainable development, including, among others, through education for sustainable development and sustainable lifestyles, human rights, gender equality, promotion of a culture of peace and non-violence, global citizenship and appreciation of cultural diversity and of culture's contribution to sustainable development. **Related Impact Areas:** 90, 91
6 CLEAN WATER AND SANITATION	06.03	By 2030, improve water quality by reducing pollution, eliminating dumping and minimizing release of hazardous chemicals and materials, halving the proportion of untreated wastewater and substantially increasing recycling and safe reuse globally. **Related Impact Areas:** 28, 29, 30, 31, 32, 33, 34, 35, 36, 37, 38, 39, 40, 41, 42, 59, 60, 61, 68, 69, 75, 84, 85, 102, 103, 104
6 CLEAN WATER AND SANITATION	06.04	By 2030, substantially increase water-use efficiency across all sectors and ensure sustainable withdrawals and supply of freshwater to address water scarcity and substantially reduce the number of people suffering from water scarcity. **Related Impact Areas:** 18, 19, 20, 21, 22, 23, 57, 63, 64, 65, 66, 67, 87, 99, 101

Table 7: SDG targets related to agricultural management and impact areas under the perspectives of resource use efficiency and ecosystem services. (continued)

07.02	By 2030, increase substantially the share of renewable energy in the global energy mix. **Related Impact Areas:** 2, 9, 13, 19, 25, 29, 35, 38, 44, 50, 5, 10, 15, 21, 26, 31, 40, 46, 65
07.03	By 2030, double the global rate of improvement in energy efficiency. **Related Impact Areas:** 8, 9, 10, 11, 55
08.02	Achieve higher levels of economic productivity through diversification, technological upgrading and innovation, including through a focus on high-value added and labour-intensive sectors. **Related Impact Areas:** 6, 11, 16, 22, 27, 32, 36, 41, 47, 52, 63, 64, 65
08.04	Improve progressively, through 2030, global resource efficiency in consumption and production and endeavour to decouple economic growth from environmental degradation, in accordance with the 10-year framework of programmes on sustainable consumption and production, with developed countries taking the lead. **Related Impact Areas:** 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 55, 18, 19, 20, 21, 22, 23, 57, 34, 35, 36, 60, 63, 64, 65, 73, 75, 77, 78, 79, 83, 84, 85, 86
09.04	By 2030, upgrade infrastructure and retrofit industries to make them sustainable, with increased resource-use efficiency and greater adoption of clean and environmentally sound technologies and industrial processes, with all countries taking action in accordance with their respective capabilities. **Related Impact Areas:** 63, 64, 65, 66, 67, 68, 69, 73, 75, 77, 78, 79, 80, 81, 83, 84, 85, 86, 102, 103, 104
10.01	By 2030, progressively achieve and sustain income growth of the bottom 40 per cent of the population at a rate higher than the national average. **Related Impact Areas:** 63, 64, 65
11.04	Strengthen efforts to protect and safeguard the world's cultural and natural heritage. **Related Impact Areas:** 88, 92, 93, 95, 96, 97
12.01	Implement the 10-year framework of programmes on sustainable consumption and production, all countries taking action, with developed countries taking the lead, taking into account the development and capabilities of developing countries. **Related Impact Areas:** 63, 64, 65, 66, 67, 68, 69, 73, 75, 77, 78, 79, 80, 81, 83, 84, 85, 86, 102, 103, 104
12.02	By 2030, achieve the sustainable management and efficient use of natural resources. **Related Impact Areas:** 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 55, 18, 19, 20, 21, 22, 23, 57, 34, 35, 36, 60, 73, 79, 83, 84, 85, 86
12.04	By 2020, achieve the environmentally sound management of chemicals and all wastes throughout their life cycle, in accordance with agreed international frameworks, and significantly reduce their release to air, water and soil in order to minimize their adverse impacts on human health and the environment. **Related Impact Areas:** 68, 69, 70, 84, 85, 86, 102, 104
13.01	Strengthen resilience and adaptive capacity to climate-related hazards and natural disasters in all countries. **Related Impact Areas:** 55, 56, 57, 58, 59, 60, 61, 62, 54, 66, 67, 74, 75, 76, 80, 81, 87, 103

Table 8: SDG targets related to agricultural management and impact areas under the perspectives of resource use efficiency and ecosystem services. (continued)

SDG	Target	Description
13 CLIMATE ACTION	13.02	Integrate climate change measures into national policies, strategies and planning. **Related Impact Areas:** 7, 12, 13, 14, 15, 16, 17, 23, 33, 42, 48, 53, 56, 83, 86, 87, 90, 91
13 CLIMATE ACTION	13.03	Improve education, awareness-raising and human and institutional capacity on climate change mitigation, adaptation, impact reduction and early warning. **Related Impact Areas:** 12, 13, 14, 15, 16, 17, 56, 90, 91
14 LIFE BELOW WATER	14.01	By 2025, prevent and significantly reduce marine pollution of all kinds, in particular from land-based activities, including marine debris and nutrient pollution. **Related Impact Areas:** 28, 29, 30, 31, 32, 33, 59, 34, 35, 36, 60, 37, 38, 39, 40, 41, 42, 61, 68, 69, 73, 85, 102, 103, 104
15 LIFE ON LAND	15.01	By 2020, ensure the conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems and their services, in particular forests, wetlands, mountains and drylands, in line with obligations under international agreements. **Related Impact Areas:** 68, 69, 73, 79, 84, 87, 88, 89, 92, 93, 96, 97, 98, 99, 100, 101, 102, 103, 104
15 LIFE ON LAND	15.03	By 2030, combat desertification, restore degraded land and soil, including land affected by desertification, drought and floods, and strive to achieve a land degradation-neutral world. **Related Impact Areas:** 73, 75, 79, 83, 103

5. Impact Areas: Assessment Perspectives - Sustainable Development Goals (SDG)

*Table 9: Impact areas supported by the BonaRes Assessment Platform. Areas from the perspective **Resource Use Efficiency** on the left (black numbers), areas from the perspective **Ecosystem Services** (blue numbers) on the right.*

№	Impact Areas Resource Use Efficiency
1	Grain/Fruit/ Tuber yield *per* Area of land
2	Aboveground Biomass *per* Area of land
3	Jobs *per* Area of land
4	Embodied Nitrogen *per* Area of land
5	Embodied Energy *per* Area of land
6	Financial benefits *per* Area of land
7	Sequestered Carbon *per* Area of land
8	Grain/Fruit/ Tuber yield *per* Energy
9	Aboveground Biomass *per* Energy
10	Embodied Energy *per* Energy
11	Financial benefits *per* Energy
12	Grain/Fruit/ Tuber yield *per* GHG emissions
13	Aboveground Biomass *per* GHG emissions
14	Embodied Nitrogen *per* GHG emissions
15	Embodied Energy *per* GHG emissions
16	Financial benefits *per* GHG emissions
17	Sequestered Carbon *per* GHG emissions
18	Grain/Fruit/ Tuber yield *per* Water
19	Aboveground Biomass *per* Water
20	Embodied Nitrogen *per* Water
21	Embodied Energy *per* Water
22	Financial benefits *per* Water
23	Sequestered Carbon *per* Water
24	Grain/Fruit/ Tuber yield *per* Human labour
25	Aboveground Biomass *per* Human labour
26	Embodied Energy *per* Human labour
27	Financial benefits *per* Human labour
28	Grain/Fruit/ Tuber yield *per* Nitrogen fertilizer
29	Aboveground Biomass *per* Nitrogen fertilizer
30	Embodied Nitrogen *per* Nitrogen fertilizer
31	Embodied Energy *per* Nitrogen fertilizer
32	Financial benefits *per* Nitrogen fertilizer
33	Sequestered Carbon *per* Nitrogen fertilizer
34	Grain/Fruit/ Tuber yield *per* Phosphorus fertilizer
35	Aboveground Biomass *per* Phosphorus fertilizer
36	Financial benefits *per* Phosphorus fertilizer
37	Grain/Fruit/ Tuber yield *per* Pesticides
38	Aboveground Biomass *per* Pesticides
39	Embodied Nitrogen *per* Pesticides

№	Impact Areas Ecosystem Services
63	Food from plants
64	Fibres from plants
65	Energy from plants
66	Seeds for breeding purposes
67	Plants for breeding purposes
68	Bio-remediation
69	Biotic filtration/sequestration/storage/accumulation
70	Smell reduction
71	Noise reduction
72	Visual screening
73	Erosion control
74	Mass movement buffering & reduction
75	Hydrological cycle & water flow regulation
76	Wind protection
77	Pollination
78	Seed dispersal
79	Maintaining habitats
80	Pest control
81	Disease control
82	Weathering & soil development
83	Maintaining soil organic matter & nutrients
84	Chemical regulation of freshwaters
85	Chemical regulation of salt waters
86	Chemical regulation of atmosphere
87	Regulation of temperature/humidity/ventilation/transpiration
88	Ecosystems enabling active enjoyment
89	Ecosystems enabling passive enjoyment
90	Ecosystems enabling research
91	Ecosystems enabling education
92	Culturally/historically important ecosystems
93	Aesthetically important ecosystems
94	Symbolically important ecosystems
95	Ecosystems used for entertainment/representation
96	Ecosystems with existence value
97	Ecosystems with option/bequest value
98	Surface water for drinking
99	Surface water for non-drinking purposes
100	Groundwater for drinking
101	Groundwater for non-drinking purposes

5. Impact Areas: Assessment Perspectives - Sustainable Development Goals (SDG)

*Table 10: Impact areas supported by the BonaRes Assessment Platform. Areas from the perspective **Resource Use Efficiency** on the left (black numbers), areas from the perspective **Ecosystem Services** (blue numbers) on the right. (continued)*

№	Impact Areas Resource Use Efficiency
40	**Embodied Energy** *per* **Pesticides**
41	**Financial benefits** *per* **Pesticides**
42	**Sequestered Carbon** *per* **Pesticides**
43	**Grain/Fruit/ Tuber yield** *per* **Money**
44	**Aboveground Biomass** *per* **Money**
45	**Embodied Nitrogen** *per* **Money**
46	**Embodied Energy** *per* **Money**
47	**Financial benefits** *per* **Money**
48	**Sequestered Carbon** *per* **Money**
49	**Grain/Fruit/ Tuber yield** *per* **Long-time investment**
50	**Aboveground Biomass** *per* **Long-time investment**
51	**Jobs** *per* **Long-time investment**
52	**Financial benefits** *per* **Long-time investment**
53	**Sequestered Carbon** *per* **Long-time investment**
54	**Yield stability** *per* **Long-time investment**
55	**Yield stability** *per* **Energy**
56	**Yield stability** *per* **GHG emissions**
57	**Yield stability** *per* **Water**
58	**Yield stability** *per* **Human labour**
59	**Yield stability** *per* **Nitrogen fertilizer**
60	**Yield stability** *per* **Phosphorus fertilizer**
61	**Yield stability** *per* **Pesticides**
62	**Yield stability** *per* **Money**

№	Impact Areas Ecosystem Services
102	**Abiotic mediation by filtration/sequestration/storage/accumulation**
103	**Physical barriers to liquid flows**
104	**Maintenance and regulation by inorganic processes**

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6. Obtaining Data

6.1 Measurement, Modelling or Expert Assessment

Once the impact areas have been selected, researchers are faced with the challenge to generate values that represent the impacts within them. Typically, researchers generate these values from either measurement, modelling or expert assessment. The three approaches are outlined below with a short description of their respective strengths and weaknesses.

Measurements

Experiments or empirical studies can be used to generate the required information, usually in the form of quantitative data. Researchers first need to decide which indicators to use for each impact area. After that, they may either conduct the measurements themselves or analyse results of published studies. This approach is supported by the BonaRes Assessment Platform through factsheets that list sample indicators for each impact area and provide references to examples of their application in current research.

In the context of impact assessments, core strengths of approaches based on measurements are the high level of detail that can be achieved, the low need for a-priori assumptions and the comparatively low risk of bias introduced by the researchers. Weaknesses are the high amount of time and effort required, which may limit the number of impact areas that can be covered, and the difficulty to measure effects of future scenarios that have no parallel in the present. Furthermore, the generalisation of measurement results is often challenging, which limits their transferability to other locations or situations.

6. Obtaining Data: Measurement, Modelling or Expert Assessment

Modelling

Where a high degree of prior knowledge on causal relationships already exists, researchers may use models for assessing impacts. Usually, models are designed to produce quantitative data, although qualitative models are also used. As above, researchers first need to select indicators to represent the respective impact areas. The information provided by the factsheets on the BonaRes Assessment Platform can be used to select indicators that are part of the output of existing models or to conceptualize requirements for new models.

In the context of impact assessments, core strengths of modelling approaches are the reproducibility of results, opportunities to investigate in detail the link between effects and underlying causes, including analyses of sensitivity, and options to analyse future scenarios that have no parallel in the present. Weaknesses are the high requirement of time, effort and information for generating new models, which also limits the options of tailoring a model to the needs of a specific assessment, and the high number of assumptions that need to be made by researchers in order to represent complex socio-ecological systems in a (simplified) model. Nevertheless, substantial progress has been made in the last decade in integrated modelling and model coupling to address complex socio-ecological systems and assess land use impact scenarios (Helming et al., 2011; Ewert et al. 2015; Lotze-Campen et al., 2018).

Expert Assessment

Experts may be consulted to asses impacts, based on their expertise and experience with regard to the topic and the selected impact areas. Values for the impacts are usually provided as qualitative data, often expressed on an ordinal or categorical scale (e.g., a Likert-type scale with ratings ranging from "very negative" to "very positive"). In the context of expert assessments, the indicators provided by the factsheets on the BonaRes Assessment Platform can be used to specify what the expert assessment is based on and to facilitate communication between different experts. For example, if experts evaluate how the ecosystem service: *provision of habitats* would be affected by a specific agricultural management, they could draw on the indicators to explain whether their assessment is mainly based on the expected species diversity of farmland-birds, or on their abundance, or on the abundance of endangered plant species, or on a mix of criteria.

In the context of impact assessments, core strengths of expert based approaches are fourfold: the adaptability to the specific context of the impact assessment, the low amount of time, work and data required, the opportunity for stakeholder involvement and shared learning. In addition, biases of impact area selection due to data availability can be avoided. Weaknesses are the danger of introducing bias through the selection of the consulted experts, the risk of low reproducibility with other experts or with the same group of experts at another point in time, and the lack of transparency of how the experts arrive at their assessment result. Thorough and comprehensible protocols for expert selection should be used to minimize the risk of introducing bias.

More information on expert based assessments is also provided in the chapter: *Framework of Participatory Impact Assessment (FoPIA).*

Please note: All three approaches have their limitations and advantages. For very complex assessments, it is therefore advisable to combine one or more of them. For example, an expert based assessment could be done first as a scoping activity to address a high amount of impact areas and identify critical ones. These impact areas could then be investigated in more detail through measurements or modelling activities.

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6. Obtaining Data: Framework of Participatory Impact Assessment (FoPIA)

6.2 Framework of Participatory Impact Assessment (FoPIA)

Participatory Impact Assessment is a method that uses stakeholder input at all stages. This includes definition of scenarios, assessment of impacts and integration of results in a multi-criteria analysis (MCA).

The FoPIA framework was developed by Morris et al. (2011) for the assessment of land use scenarios in European regions. Three stages can be distinguished:

1.) **Defining Scenarios**: After the general topic of the assessment has been set, stakeholder interviews are conducted to determine the form that policy interventions or management changes are likely to have within the different national circumstances. For example, farmers could be asked what types of climate change mitigation measures are likely to be introduced in their country. Land use scenarios are then designed based on the identified interventions or management changes.

2.) **Assessing Scenario Impacts**: In a second step, a stakeholder workshop is conducted where participants (as experts) assess the likely impacts of the scenarios on nine impact categories. Of these categories, three represent the ecological dimension, three the economic dimension and three the social dimension of sustainability. The assessed impacts are then compared to sustainability limits set by the workshop participants.

3.) **Integrated Evaluation**: Finally, a multi-criteria analysis is conducted. Participants assign weights to each impact category that reflect the importance they attribute to it. This allows the calculation of an overall score for each scenario and a ranking of the different scenarios according to this score. *A more detailed description of multi-criteria analysis is provided in the next chapter.*

While FoPIA was originally developed for policy impact assessment in a European context, several case studies have since proven its adaptability and suitability for non-policy situations (Hermanns et al., 2017) and for use in non-European countries, and in particular for developing countries (König et al., 2013; König et al., 2014; Schindler et al., 2016; Graef et al., 2018).

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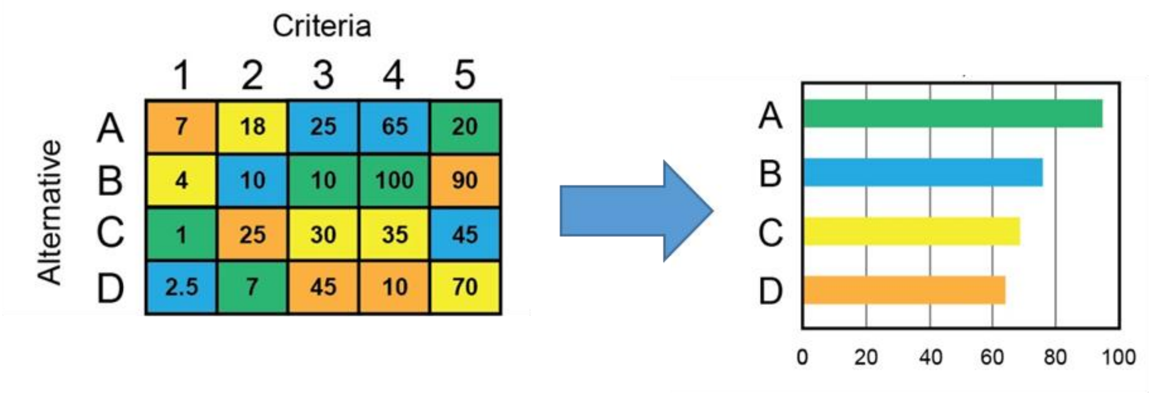
7. Integrating Results: Multi-Criteria Analysis (MCA)



7. Integrating Results: Multi-Criteria Analysis (MCA)

7.1 Multi-Criteria Analysis (MCA)

Info-Box

Multi-criteria analysis (**MCA**) is a class of procedures designed to aid decision-making where options are evaluated based on more than one criterion, in particular where these criteria are measured in different units which can not be readily converted into each other. Many different MCA procedures exist, but all are based on the principles of data normalization and of assigning weights to the different criteria in order to arrive at an integrated score for each option.

MCA procedures are very useful in multifunctional disciplines such as agriculture, where all management choices have multiple effects and involve **trade-offs** within and between the **social**, **economic** and **environmental** spheres.

What is multi-criteria analysis?

Multi-criteria analysis (MCA) is a structured procedure to compare and rank options while considering more than one criterion. This is achieved by normalizing all quantitative and/or qualitative information about an option, assigning weights to all criteria, and then combining the information into a single, integrated value. Multi-criteria analysis is a tool for dealing with the inevitable trade-offs within complex decision-making situations, which may also feature high uncertainty, different forms of information, and multiple stakeholder interests and perspectives. For this reason, MCA has gained momentum as a methodology for the evaluation of sustainability (Adams & Ghaly, 2007). In the context of impact assessments, multi-criteria analysis can be used at the end of the research to combine the indicator values of multiple impact areas into integrated results and to rank the investigated management options.

Multi-criteria problems are common occurrences. For example, a farmer's decision over which crops to plant may depend on factors such as expected revenues, risk of crop failure, time requirements, or personal preferences. Weighing them against each other in order to find the optimal solution is difficult because factors are assessed in different units that can not objectively be converted into each other (e.g., how much additional revenue would exactly balance a higher risk of crop failure or a reduction in spare time?). These decisions always involve an element of normative valuation. Usually, they are made intuitively based on previous experience. In impact assessments, however, a structured approach is required to avoid subjectivity, guarantee transparency, and to address problems that are too complex to solve intuitively. Multi-criteria analysis facilitates this by providing a method to combine scientific assessments with normative value choices. Indicator values from different impact areas are first normalized to a common, dimensionless scale and then weights are assigned, based on the perceived importance of the impact area for the respective assessment. This second step is purely normative and should ideally be based on stakeholder involvement or on societal norms codified in laws or regulations. Because normative decisions are always subjective and the weights assigned may not be reproducible with a different set of stakeholders or with the same set of stakeholders at another time, it is very important to document how decision makers or stakeholder groups were selected, or by what method weights were derived from laws or regulations. Finally, for each option analysed in the assessment, the normalized indicator values representing an impact area are multiplied by the weights assigned to that area and all products are summed up to generate a total, integrated score. If the impact assessment investigated different options, they may then be ranked based on the integrated score.

Why is multi-criteria analysis relevant for the assessment of soil management?

Sustainability impact assessment is characterized by investigating multiple impact areas related to the **economic, ecological and social dimension**. The findings for each impact area will be represented by an indicator that comes in a specific unit, such as "*number of ground-nesting bird species*" or "*tons of wheat*", which can not be easily converted into a common unit. However, to rank options and to decide which management is the most sustainable, exactly that is needed. Multi- criteria analysis provides a structured and transparent way to achieve integration of assessment results. MCA is particularly useful with regard to assessments of agricultural management, where management choices have multiple effects and usually involve trade-offs.

How can multi-criteria analysis be considered in assessments of soil management?

To explain the application of a multi-criteria analysis, we use a simplified example where a farmer needs to choose between **three alternative fertilisation options** and where only two impact areas from the perspective of resource use efficiency are assessed: *productivity*, measured as harvested ton of wheat per hectare, and *nitrogen use efficiency (NUE)*, measured as kilogram nitrogen taken up by the crop per kilogram of nitrogen applied as fertilizer.

- **Option A:** Low fertiliser application rate
- **Option B:** High fertiliser application rate
- **Option C:** Intermediate fertiliser application rate

Table 8 shows hypothetical values for productivity and nitrogen use efficiency for each of the options.

7. Integrating Results: Multi-Criteria Analysis (MCA)

Table 11: Hypothetical values for productivity and nitrogen use efficiency resulting from three fertilization intensities.

Option	Productivity [t/ha]	Nitrogen Use Efficiency [%]
A	3.9	65%
B	8.2	40%
C	6.3	55%

Step 1 - Normalizing indicator values

The first step of the multi-criteria analysis is to convert all indicator values onto a numerical (dimensionless) scale to enable direct comparison. One approach often used is to **allot scores** between 0 and 100, with 100 representing the best possible result within the respective category and 0 representing the worst (Dodgson et al., 2009). Where a large number of options are compared, transformation into z-scores is also possible.

It is crucial to the validity of the MCA result that scores are determined as objectively as possible. For the current example, it is assumed that a productivity between a **maximum of 10 t/ha** and a **minimum of 2 t/ha** could normally be expected. A maximum score of 100 is therefore assigned to yields of 10 t/ha (or more) and a minimum score of 0 is assigned to yields of 2 t/ha (or less). Intermediate values are given proportionate scores along the scale. For nitrogen use efficiency, a maximum of 80% and a minimum 10% is assumed. The table below shows the resulting scores for each option.

Table 12: Hypothetical scores for productivity and nitrogen use efficiency resulting from three fertilization intensities.

Option	Productivity Score [0-100]	Nitrogen Use Efficiency Score [0-100]
A	24	79
B	78	43
C	54	64

Step 2 - Assigning weights to each impact area and multiplying with scores

Once values have been converted onto a comparable numerical scale, the next step is **to assign weights to the impact areas** that r**epresent their relative importance**. For example, the farmer could decide to value both criteria equally (50/50) or put stronger emphasis on either productivity or nitrogen use efficiency (e.g., 60/40).

Step 3 - Calculating integrated scores and final assessment

Weighted scores are calculated by multiplying the score of an impact area with the weight it has been assigned. All weighted scores of an option are then summed up to create a total, integrated score. By comparing these integrated scores, a ranking can be achieved and the best option can be identified.

For example, if productivity and nitrogen use efficiency are considered by the farmer to have **equal** importance (50% each), Option B achieves the best outcome with a total score of 60, closely followed by option C with a total score of 59 (see Table 10).

7. Integrating Results: Multi-Criteria Analysis (MCA)

Table 13: Calculation of integrated scores based on productivity and nitrogen use efficiency for an example of three fertilization intensities. A weighting of productivity: 50% and nitrogen use efficiency: 50% is assumed.

Option	Productivity Score [0-100]	Weighted Score (Score * 50%)	Nitrogen Use Efficiency Score [1-100]	Weighted Score (Score * 50%)	Total Weighted Scores
A	24	12	79	39	51
B	78	39	43	21	60
C	54	27	64	32	59

If the farmer considers nutrient use efficiency to be more important and assigns a weight of **40% to productivity** and a weight of **60% to nitrogen use efficiency**, the best outcome is achieved by option C (see Table 11).

Table 14: Calculation of integrated scores based on productivity and nitrogen use efficiency for an example of three fertilization intensities. A weighting of productivity: 40% and nitrogen use efficiency: 60% is assumed.

Option	Productivity Score [0-100]	Weighted Score (Score * 40%)	Nitrogen Use Efficiency Score [1-100]	Weighted Score (Score * 60%)	Total Weighted Scores
A	24	10	79	47	57
B	78	31	43	26	57
C	54	22	64	39	60

Obviously, which option is considered to achieve the best outcome is strongly dependent on the weights assigned to each impact area. The weighing process is inherently subjective and only has a limited precision. Where options achieve very similar integrated scores, it is therefore practical to consider their outcomes to be equal. In our example where productivity and nitrogen use efficiency were assigned equal weights, options B & C could therefore be considered to have the same merit (scores of 69 and 60).

In the interest of transparency, it is important to explain the reason for the assigned weights. This includes documenting if decision makers or larger groups of stakeholders were involved, what criteria were used for their selection and what methods were applied for arriving at the final weighting. **Please consider that the assigning of equal weights for all impact areas is also a normative decision that has a strong influence on the final assessment results.**

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7.2 Benchmarking

Info-Box

Benchmarking uses information of similar organizations or processes to assess the gap between current and achievable performance. In the context of agriculture, benchmarking usually refers to a systematic comparison of the performance of farms producing under similar bio-geophysical conditions.

Benchmarking assesses performance in relative, rather than in absolute terms. This is particularly suited for assessing effects of agricultural management, because the strong influence of external factors such as climate, weather or soils on farm performance is difficult to isolate. Simple **benchmarking identifies best practices** from a set of samples and then **evaluates all samples relative to these best practice examples**. In more complex benchmarking where multiple evaluation criteria are applied, samples are not evaluated relative to a single best practice but relative to a so called **efficiency frontier,** which represents the best performances considered achievable and which is calculated based on data from all samples. The most common parametric and non-parametric approaches to this are Stochastic Frontier Analysis (SFA) and Data Envelopment Analysis (DEA), respectively.

What is Benchmarking?

Benchmarking is a relative performance evaluation that compares the efficiency of similar organizations or processes. It is used in many sectors. In the context of agriculture, benchmarking usually refers to the systematic comparison of the performance of farms that produce under similar bio-geophysical conditions, using the same resources and producing the same type of products.

In the context of benchmarking, efficiency is defined more loosely than in the definition of resource use efficiency described in the previous chapters. Here, efficiency implies a maximization of positive outputs and a minimization of inputs and of negative outputs. Evaluations of efficiency can be based on very different criteria, such as on provision of ecosystem services, effects on biodiversity or resource use. All impact areas presented on the BonaRes Assessment Platform can therefore be used in benchmarking.

Simple benchmarking first identifies best practices within a set of samples. It then evaluates all samples by assessing the difference between their performance and the performance of the best practices (Oh & Hildreth, 2016). More complex benchmarking methods that apply multiple evaluation criteria use data from all samples to calculate a so called efficiency frontier. This frontier represents the best performances considered achievable. The performance of all samples is then assessed relative to this frontier. The most common methods for this are Stochastic Frontier Analysis (SFA) and Data Envelopment Analysis (DEA).

Benchmarking is often conducted to improve organizational processes and to achieve higher standards of performance. This can be accomplished by learning from the experience of best practices or by copying their methods or processes (Malano et al., 2004). Benchmarking studies can also be used to assess the impact of policies (Bogetoft & Otto, 2010).

Why is benchmarking relevant for assessing soil management?

The performance of farms strongly depends on bio-geophysical site conditions. Where assessment seeks to evaluate different types of soil management, it is challenging to isolate and quantify the influence of external factors. For example, if a farmer in Finland achieves a harvest of 4.5 tons of wheat per hectare and a farmer in France achieves a yield of 7.0 tons, how much of this difference is then due to their respective managements and how much is an effect of the different agro-climatic zones?

To avoid such problems, performance is often measured in relative terms through benchmarking. In agriculture, benchmarking plays an important role in the context of assessing environmental impacts of farm performance, such as impacts related to water conservation or soil protection (Kuo et al., 2014). Results of benchmarking studies are useful at multiple decision making levels, enabling policymakers to assess the effect of agricultural policies (Quiroga et al., 2017) and allowing farmers to learn from best practice examples.

How can benchmarking be applied in assessments of soil management?

Benchmarking studies can be applied with different degrees of complexity. In the simplest form, only one criterion is used to evaluate performance and a best practice is identified which scores highest for this criterion. The best practice is then used to evaluate all other samples and to serve as an example. For instance, several wheat producing farms in a region could be compared in a benchmarking to identify the one that achieves highest yields. All other farms could then be evaluated relative to this and farmers could try to use the example to optimize their own management.

In a more complex form, several criteria are evaluated and the information of fall samples are mathematically combined to form a so called "**efficiency frontier**". The most common parametric method for this is **S**tochastic **F**rontier **A**nalysis (**SFA**), the most common non-parametric method is **D**ata **E**nvelopment **A**nalysis (**DEA**). Both methods have been developed as economic tools to evaluate the performance of companies, enterprises or non-profit organizations, which are referred to as "decision-making units (DMU)” in the literature (Malana & Malano, 2006). The abstract term DMU is used to highlight that the methods are applicable for all types of organizations where management (decision making) occurs. In the context of soil related impact assessments, the DMUs are usually farms.

Both SFA and DEA address the question of how the highest amount of positive outputs can be achieved with the lowest amount of inputs and the lowest amount of negative outputs. For this, they calculate an efficiency frontier line, which represent the (hypothetical) optimal combination of inputs (Oh & Hildreth, 2016). The efficiency of each DMU is calculated as the distance to this efficiency frontier and often reported as a percentage (with DMU positioned on the frontier line having an efficiency of 100%). SFA and DEA can be used to estimate overall efficiency of farms or management processes and rank them based on their performance. Additionally, the methods can be used to identify areas of inefficiency in order to support improvement.

Stochastic Frontier Analysis (SFA)

Stochastic Frontier Analysis is a method developed from stochastic frontier production function models (Aigner et al., 1977). SFA is based on **parametric models**. Such models are defined a-priori except for a finite set of unknown parameters that need to be estimated from data. In SFA, these parameters often include effects of noise (random shocks unrelated to management, such as weather effects) and efficiencies (Bogetoft & Otto, 2010). An efficiency frontier is calculated based on actual input/output relationships, and the efficiencies of all DMU are measured relative to it. It is not necessary for any DMU in the sample to be positioned on the efficiency frontier, i.e. to achieve full efficiency.

With SFA approaches, hypotheses about production structure and degree of inefficiency can be assessed statistically. However, the method requires explicit parametric functions representing relationships between inputs and outputs, and distributional assumptions. For considering measurement errors and effects which cannot be controlled by the individual DMU, such as weather, an error term is incorporated in the calculations. The validity of SFA results depends on the accuracy of the assumptions made (Odeck, 2007).

Data Envelopment Analysis - DEA

DEA calculates the efficiency frontier line (highest positive outputs from lowest amount of inputs and negative outputs) from linear combinations of the samples. DMU that score highest in at least one criterion will always be positioned on the efficiency frontier. Two types of DEA models can be distinguished: The input-oriented model aims towards minimizing inputs while holding the amount of outputs constant. Output-oriented models on the other hand aim to increase outputs with the same amount of inputs (Malana & Malano, 2006).

As a difference to parametric models, **non-parametric** approaches such as DEA do not require any a-priori assumptions regarding the quality or distribution of the data, avoiding potential errors in this regard (Oh & Hildreth, 2016). However, this makes the methods sensitive to measurement errors or other noise affecting the dataset. In DEA, all parameters are calculated from the data in the sample.

Environmental factors are not explicitly considered. It is therefore important to carefully select the sample to avoid comparing farms that work under different environmental conditions (Odeck, 2007).

Please note: Both SFA and DEA allow to evaluate performance with regard to multiple inputs and outputs and facilitate an integrated evaluation (Bogetoft & Otto, 2010). Both SFA and DEA also have specific advantages and disadvantages to be considered when choosing methods. Alternatively, the methods can be combined to overcome the disadvantages, and such combinations have been applied in several studies dealing with benchmarking processes (Oh & Hildreth, 2016).

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8. Conclusion & Outlook

8. Conclusion & Outlook

This handbook is the first to offer a short guide to soil related impact assessments. It provides in a citable book format the information presented online on the BonaRes Assessment Platform. Both handbook and the platform have been produced in response to a growing awareness of the fundamental role of soils and of the importance of sustainable soil management for coping with societal challenges such as food security, adaptation and mitigation of climate change, or biodiversity loss.

While the handbook details the process and challenges of conducting soil related impact assessments, it was not possible to provide detailed information on the individual impact areas and suitable indicators. This gap will be closed by a complementing publication that exclusively focuses on this topic and will be published in 2020.

The Handbook of Soil Related Impact Assessment is designed as a living document. We envision the information presented therein to be continually updated and a substantial amount of new content to be added during and beyond the duration of the BonaRes program. On the online platform, this will be considered in the form of regular updates. For the handbook, new versions will be published at irregular intervals to account for these changes.

We strongly encourage feedback from readers to improve both handbook and assessment platform, in order to make them practical tools to support soil related impact assessments.

9. Index of Figures & Tables

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9.1 Figures

9. Index of Figures & Tables

9.1 Tables

Previous publications

2019/3 Gerdes et al. Workshop-Protokoll: Den Unterboden nutzen, um auf Trockenperioden besser vorbereitet zu sein - Eine Akzeptanzanalyse von Maßnahmen zur Aufwertung des Unterbodens
DOI: 10.20387/BonaRes-W02N-H27N

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