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Soil threats in Europe

Status, methods, drivers and effects on ecosystem services

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RECARE
Preventing and Remediating degradation of soils in Europe through Land Care

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Soil threats in Europe: status, methods, drivers and effects on ecosystem services

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Land and Urban Management - RECARE

The challenge

As soil formation is an extremely slow process, soil can be considered a non-renewable resource. Soils should thus be adequately protected and conserved to ensure that soil functions are not lost or diminished. Soil functions are, however, threatened globally by a wide range of processes, and in Europe, a number of threats have been identified in the European Soil Thematic Strategy. The challenge is to prevent degradation and its adverse effects on soil functions and ecosystem services, while simultaneously improving lively-hoods.

Project Objectives

Main objectives of RECARE are to:

1. Fill knowledge gaps in our understanding of the functioning of soil systems under the influence of climate and human activities,
2. Develop a harmonized methodology to assess state of degradation and conservation,
3. Develop a universally applicable methodology to assess the impacts of soil degradation upon soil functions and ecosystem services,
4. Select in collaboration with stakeholders, innovative measures, and evaluate the efficacy of these regarding soil functions and ecosystem services as well as costs and benefits,
5. Upscale results from case studies to European scale to evaluate the effectiveness of measures across Europe,
6. Evaluate ways to facilitate adoption of these measures by stakeholders,
7. Carry out an integrated assessment of existing soil related policies and strategies to identify their goals, impacts, synergies and potential inconsistencies, and to derive recommendations for improvement based on RECARE results,
8. Disseminate project results to all relevant stakeholders.

Methodology

As degradation problems are caused by the interplay of bio-physical, socio-economic and political factors, all of which vary across Europe, these problems are by definition site specific and occur at different scales. Therefore, 17 Case Studies of soil threats are included in RECARE to study the various conditions that occur across Europe and to find appropriate responses using an innovative approach combining scientific and local knowledge.
The recently completed FP6 DESIRE project developed a successful methodological approach to evaluate mitigation and restoration measures against desertification in collaboration with stakeholders. This approach will be adapted to include other soils threats, and to evaluate ecosystem services. By integrating results from the Case Studies, knowledge gaps in our understanding of soil systems and their interaction with humans can be addressed, and more general conclusions can be drawn for each soil threat at the broader European level.

Expected Results

RECARE will improve the scientific understanding of complexity and functioning of soil systems and interaction with human activities. The main RECARE scientific innovations are related to the integrated trans-disciplinary approach for assessing preventing, remediating and restoring soil degradation in Europe. RECARE will contribute scale-appropriate solutions to soil degradation problems, which will in addition restore soil functionality and ecosystem services throughout Europe.
The engagement of relevant stakeholders will help to i) identify existing obstacles to the integration of soil protection objectives into and between relevant policies and ii) to reveal solutions to overcome these impediments. RECARE will support improved implementation and coherence across a number of relevant EU policies and strategies.

13 SOIL FUNCTIONS & ECOSYSTEM SERVICES

Gudrun Schwilch, Lea Bernet, Heleen Claringbould, Luuk Fleskens, Elias Giannakis, Julia Leventon, Teodoro Marañón, Jane Mills, Chris Short, Jannes Stolte, Hedwig van Delden, Simone Verzandvoort

13.1 Introduction

In order to fulfil RECARE's aim to quantify in a harmonized, spatially explicit way impacts of degradation and conservation on soil functions and ecosystem services, it is important to understand the concept and review the current scientific debate. This will lay the foundation for the development and selection of appropriate methods to measure, evaluate, communicate and negotiate the services we obtain from soils with stakeholders in order to improve land management.

Despite various research activities in the last decades across the world, many challenges remain to integrate the concept of ecosystem services (ES) in decision-making, and a coherent approach to assess and value ES is still lacking (de Groot *et al.*, 2010). There are many different, often context-specific, ES frameworks with their own definitions and understanding of terms. This chapter therefore aims to identify the state of the art and knowledge gaps in order to develop an operational framework of the ES concept for the RECARE project. It will provide an overview on existing soil functions and ES frameworks and on approaches to monitor and value ES, with a special focus on soil aspects. Furthermore, it will address the question how the ES concept is operationalized in research projects and land management in Europe so far. Based on this review, the chapter concludes with a suggestion of an adapted ES framework for RECARE and on how to operationalize it for practical application in preventing and remediating degradation of soils in Europe.

13.2 Soil functions and ecosystem services concept

The soil functions concept emerged in the European soil science community during the early 1970's (Glenk *et al.*, 2012) and was adopted for the development of the EU Soil Framework Directive with seven key soil functions (European Commission, 2006):

- Biomass production, including in agriculture and forestry
- Storing, filtering and transforming nutrients, substances and water
- Biodiversity pool such as habitats, species and genes
- Physical and cultural environment for humans and human activities
- Source of raw materials
- Acting as carbon pool (store and sink)
- Archive of geological and archaeological heritage.

This concept exists in many different forms. Blum (2005) categorized the soil functions in 'Ecological functions' and 'Non-ecological functions'. The *Ecological functions* consist of 'biomass production', 'protection of humans and the environment' and 'gene reservoir'. The *Non-ecological functions* cover 'physical basis of human activities', 'source of raw materials' and 'geogenic and cultural heritage'. However, soil functions, soil roles and soil ES are often used interchangeably and thus many lists of soil functions exist. This is due to the term 'function', which, according to Jax (2005), is primarily used in four ways (see Glenk *et al.*, 2012):

- Functions used as a synonym for processes
- Function used to mean the operation (function(ing)) of a system
- Functions used as a synonym for roles
- Functions as services.

In RECARE, we understand soil functions as synonym for roles (and partly services), in order to avoid confusion with the well-understood term soil processes. Dominati *et al.* (2010) stated that the existing literature on ES tends to focus exclusively on the ES rather than holistically linking these services to the natural capital base from which they arise. Although soils are major suppliers of critical ES, soil services are often not recognised, generally not well understood and thus not incorporated into the framework, nor is the link between soil natural capital and these services (Breure *et al.*, 2012). Haygarth and Ritz (2009) suggested combining ES with soil functions that are relevant to soils and land use in the UK. They presented for each of their identified 18 services an associated soil function. Dominati *et al.* (2010) suggested the following roles of soils in the provision of services:

- Fertility role
- Filter and reservoir role
- Structural role (i.e. physical support)

- Climate regulation role
- Biodiversity conservation role
- Resource role.

These correspond roughly to the soil functions as presented by the European Commission (2006) above, and are, in our view, overlapping with what is generally considered an ES. One aspect that might be added is the increasing awareness of cultural services. Under this ES category knowledge systems associated with soils might be considered. Figure 13.1 shows the number of soil function and ecosystem service publications in ISI journals between 1976 and 2013. "Soil functions" appeared in the literature substantially earlier than "soil ecosystem services", i.e. first occurrence in 1976 and 1996, respectively. "Soil function" publications started steadily increasing from the early 1990s, while "soil ecosystem service" publications did so from the late 1990s. From the middle of the 2000s, the rate of increase in ISI publications with "soil" and "ecosystem service" in the title, abstract, or key words, outstripped that of "soil functions", resulting in five times more publications by 2013 (Figure 13.1). This trend may be explained by an increase in research and publications on the general topic, and/or by a partial switch from authors using the term "ecosystem service" instead of "soil function".

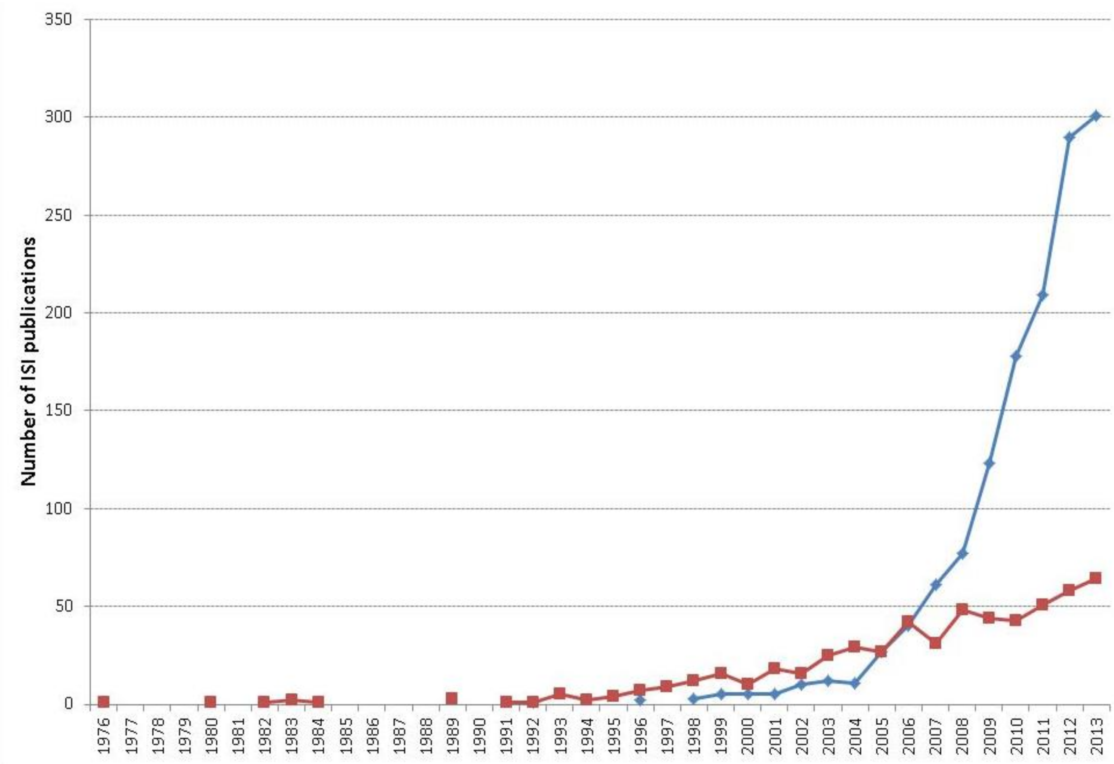

Figure 13.1: Temporal trends in ISI papers on "soil functions" and "ecosystem services". In red are ISI papers with "soil function" in the title, abstract, or key words. In blue are ISI papers with "soil" and "ecosystem service" in the title, abstract, or key words. All searches were done in SCOPUS (25-09-2014).

Focusing on soils, as in the RECARE project, requires differentiating ES delivered specifically by soils from those services generally provided by land (of which soil is as part). Often, the value of soil has only implicitly been valued within that of land (Robinson *et al.*, 2014). Increased pressure on policymakers to consider soil multi-functionality in their decision-making regarding the use of land, justifies that soil functions and ES are prominent in decision-making frameworks (Robinson *et al.*, 2014).

Glenk *et al.* (2012) considered the following frameworks as the most comprehensive and consistently classifying and describing the linkages between soil and its management and resulting impacts on ES: Robinson and Lebron (2010), Dominati *et al.* (2010) and Bennet *et al.* (2010). Glenk *et al.*'s (2012) key message is that "soil functions should be viewed as (bundles of) soil processes that are providing input into the delivery of (valued) final ecosystem services" (p. 35). Robinson *et al.* (2013) suggest an earth-system approach to provide more visibility to soils and other compartments of the earth-system in the supply chain for ES. Although it includes many valuable considerations and a useful focus on soils, its stock-flow model becomes rather complex for practical application.

For the RECARE project, we will link the state of soil degradation to soil processes that in turn affect soil functions and ES. As many soil processes and ES are interconnected, damages from soil threats are potentially affecting all ES. This is also reflected in RECARE's definition of soil threats. While the ENVASSO project (Jones *et al.*, 2008) defined a 'soil threat' as "a phenomenon that causes a deterioration or loss of one or more soil functions", RECARE's definition refers to the "loss of one or more soil-based ecosystem services".

13.3. ES frameworks

13.3.1 History

The ecosystem services (ES) concept is considered to be a useful communication tool to highlight the dependence of human well-being on ecosystems. It has the potential to bridge the gaps between ecology, economics and society in order to achieve sustainable resource management (Braat and de Groot, 2012). Its most recent definition as proposed by Braat and de Groot (2012) is "Ecosystem services are the direct and indirect (flux of) contributions of ecosystems to human well-being". The term "ecosystem services" was first proposed in early 1980s to increase public awareness about the negative consequences of biodiversity loss on the human welfare (Ehrlich and Ehrlich, 1981; Mooney and Ehrlich, 1997). Ecologists and natural scientists were stressing that beyond the ethical value of biodiversity, *per se*, there was the utilitarian reason to preserve biodiversity because it supports the ES needed for human wellbeing. The ES concept also considered the 'intergeneration equity argument', i.e. that future generation have the same rights to natural resources as the current generation.

Since then, the number of papers addressing ES has increased exponentially (Vihervaara *et al.*, 2010) with a broader focus on natural capital beyond biodiversity aspects (Fisher *et al.*, 2009). Economists recognized that the contributions of ecosystems to human welfare were more wide-ranging than previously thought and heavily undervalued in decision-making (Braat and de Groot, 2012). Thus, from the 1990s, a growing interest on methods to estimate the economic value of ES can be found in order to evaluate the impact of alternative ecosystem management strategies on the provision of ES and to visualize their value in decision-making. A significant milestone was the first economic valuation of the Earth's natural capital and ES (Costanza *et al.*, 1997). A new discipline, 'Ecological Economics', was launched to analyse the economic system as a subsystem of the ecosphere.

The release of the Millennium Ecosystem Assessment (2003, 2005) finally led to the widespread integration of ES in policy decision-making (Gómez-Baggethun *et al.*, 2010). The potential of ecosystems to provide ES depends on ecosystem functioning, which in turn depends on the biophysical structure of the system and processes therein (de Groot *et al.*, 2010). Soils are part of the biophysical structure, and provide, through its processes, ES for human wellbeing. Recently, soil science has recognised the importance of the ES concept for prevention and mitigation of soil degradation. There are many efforts to incorporate the ES concept in soil policy making (Breure *et al.*, 2012; Robinson *et al.*, 2012), as it legitimates soil conservation practices by illustrating the broad value of healthy soils and it helps to evaluate them regarding trade-offs.

13.3.2 Comparing ES frameworks

The Millennium Ecosystem Assessment (MEA, see www.maweb.org), supported by the United Nations, represented a formidable cooperative work of more than 1,300 scientists and experts of 95 countries producing the first comprehensive audit of the Earth's natural capital. The aim of MEA was to provide scientific information about the effects of global change drivers on world ecosystems and to evaluate the consequences of ecosystem degradation for human well-being. While there is no single, agreed method of categorizing all ES, the MEA (2005) is widely accepted and is seen as a useful starting point. MEA defines four types of ecosystem services as summarized below.

(i) Provisioning services: products obtained from ecosystems including food, fibre, fuel, land, water, natural medicine, biochemical and genetics, ornamental resources.
(ii) Regulating services: benefits obtained from the regulation of ecosystem processes including carbon sequestration, erosion control, flood protection, pollination, water purification and waste management.
(iii) Cultural services: non-material (use and non-use) benefits that individuals obtain from ecosystems including spiritual, religious and cultural heritage, recreation and tourism, landscape and amenity.
(iv) Supporting services: services that are necessary for the production of all other ecosystem services including soil formation and retention, cycling processes and habitat provision.

The identification and assessment of the direct (land-use change, climatic change, exotic species, contamination, etc.) and indirect (demographic, socio-economic, etc.) drivers on the degradation of the ES were recommended as tools for the decision makers (MEA, 2005). A critique to the MEA was that processes (means) for achieving services, and the services themselves (ends), have been mixed within the same classification category, e.g. water regulation is a process to achieve potable water (Wallace, 2007). One needs to distinguish between intermediate service (e.g. water regulation), final service (e.g. clean water provision) and benefit (e.g. drinking water) (Boyd and Banzhaf, 2007; Fisher *et al.*, 2009).

In response to these critiques, 'The Economy of Ecosystems and Biodiversity' (TEEB, 2010) developed a new cascading framework, which distinguishes between biophysical structure, function, service, benefit and value. It was supported by the United Nations (UNEP) and the European Commission and it is currently considered as the best available framework for ecologically-based, social and economic decision making (Braat and de Groot, 2012), see Figure 2. TEEB approach recommends three steps:

1. Identify and assess the full range of ES. This includes definition and mapping of indicators of biodiversity and ES; quantification and modelling of trade-offs between ES.
2. Estimate and demonstrate the value of ES, both in physical units and in monetary terms, including recognition of changes over time.
3. Capture and manage the values and seek solutions to overcome their undervaluation. This entails providing information about ecosystem benefits and values to help policy-makers, business and society reaching decisions that consider the full (market and non-market) costs and benefits of a proposed use of an ecosystem.

In a recent report about different approaches to value ES in Europe (Brouwer *et al.*, 2013) authors concluded that "one of the main findings is that there does not exist one single, standard "TEEB" method or approach." To reach the common target of valuation of ES in Europe (mandated by the EU 2020 Biodiversity Strategy) the existing frameworks need further integration and implementation (Brouwer *et al.*, 2013).

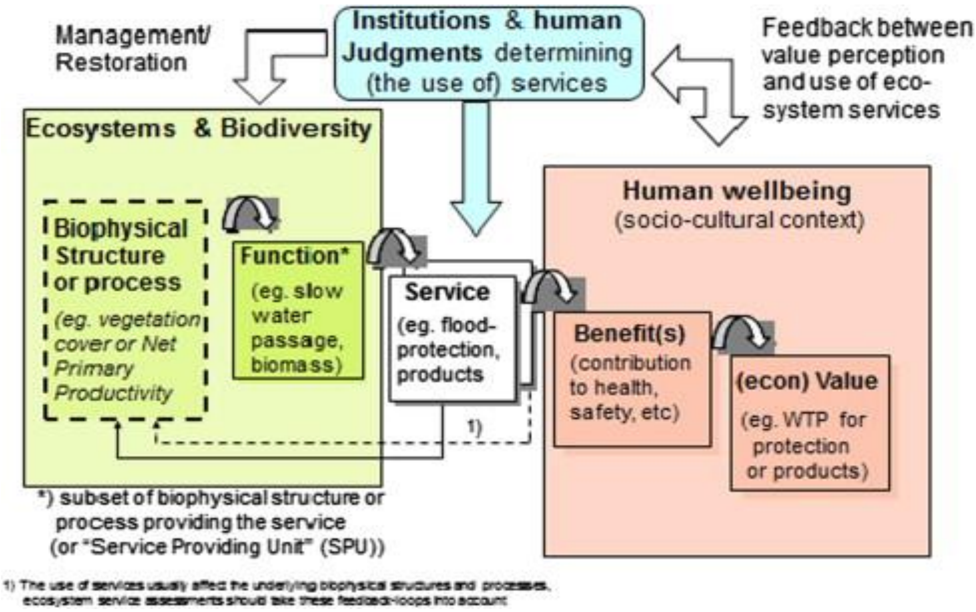

Figure13.2: The Economics of Ecosystems and Biodiversity (TEEB) overview diagram. Braat and de Groot et al. (2012), adapted from Haines-Young and Potschin (2010). As this framework was designed for economic valuation purposes it focuses mainly on economic values without considering other value systems.

Related to the frameworks for ES is the Common International Classification of Ecosystem Services (CICES) initiative developed from the work on environmental accounting undertaken by the European Environment Agency (EEA) (Haines-Young and Potschin, 2013). It supports their contribution to the revision of the System of Environmental-Economic Accounting (SEEA) which is currently being led by the United Nations Statistical Division (UNSD). Since the original proposal interest in CICES has grown. It has now become clear that in addition to the need for standardization in the context of environmental accounting, work on mapping and valuing ES and ecosystems assessments more generally would benefit from more systematic approaches to naming and describing ES.

For the purposes of CICES, ES are seen as arising from the interaction of biotic and abiotic processes, and refer specifically to the 'final' outputs or products from ecological systems; that is, the things (goods or services) directly consumed or used by people. Following common usage, the classification recognises these outputs to be provisioning, regulating and cultural services, but it does not cover the so-called 'supporting services' originally defined in the MEA. The supporting services are treated as part of the underlying structures and processes that characterise ecosystems. This is particularly important for RECARE given the positioning of soils in ES.

The latest version of CICES (V4) has a five level hierarchical structure (section – division – group – class – class type). At the highest level are the three familiar sections from the MEA (see CICES V4, www.cices.eu). CICES has contributed considerably to a standardized naming of ES, but it is mainly natural science based with a weak inclusion of social aspects and has at the same time become rather complex using many scientific terms.

MEA, TEEB, CICES and consecutive researcher groups have tried to clarify the jumble of terms in ES frameworks. However, a clear and generally accepted framework and an agreement on terms is lacking. For example, the biophysical structure of the ecosystem (TEEB) is often called biophysical process or property (Braat and de Groot, 2012; Maes *et al.*, 2012; Müller and Burkhard, 2012; and others). Together with the ecosystem functions, it supports/provides, this ecosystem side of the framework is also named 'natural capital stocks' (Dominati *et al.*, 2010) or 'ecosystem potential' (Bastian *et al.*, 2013; Haines-Young *et al.*, 2012; Rutgers *et al.*, 2012). On the human wellbeing part of the framework, TEEB suggests to distinguish service, benefit and (economic) value, while others talk about 'intermediate service' and 'final service' (Crossman *et al.*, 2013), also highlighting the distinction of services supply and demand. Some authors describe the 'service' in TEEB as 'provision' and the 'benefit' as 'use/service', while the value considered the 'importance or appreciation of a service'. This lack of consistent typology leads to the increasing use of interchangeable terms such as: properties, processes, functions and services (Robinson *et al.*, 2013). Other preferential terms used are 'stocks of natural capital' and 'flows of ecosystem services' (Crossman *et al.*, 2013 and others). One of the aims of this review is to develop an agreed framework for RECARE with clearly defined and consistently used terms (see par. 13.7).

13.4 Measuring, monitoring and mapping ES

ES research has undertaken major efforts to quantify and measure ES. Considerable focus has been put in identifying the relevant indicators and how to measure them in order to map and quantify ES at different spatial and temporal scales. This has presented some challenges, particularly for cultural services, which are more difficult to quantify and measure than other ES. As far as a possible, all changes in ES need to be identified and quantified and excluding some classes of services because they are difficult to quantify and measure should be avoided (Braat and de Groot, 2012). Quantifying bundles of ES and recognizing the interrelations between components of indicator sets, however, remain major challenges to monitoring ES flows.

Müller and Burkhard (2012) understand ES as ecological indicators and made various suggestions on how to improve the quality of the indicators, such as improving knowledge about relevant cause-effect relations, recognizing the interrelations between indicators, improving the transparency of the indicator derivation strategies, finding case-specific optimal degree of indicator aggregation, assessing indicator uncertainties or estimating the normative loadings in the indicator set.

De Groot *et al.* (2010) suggested that "indicators are needed to comprehensively describe the interaction between the ecological processes and components of an ecosystem and their services" (p. 262). There are state as well as performance indicators needed to differentiate between the component of the service provision and the sustainable use of it. In fact, for each element in the ES framework, specific indicators are needed. On the ecosystem side, property and function indicators provide information about the potential service of an ecosystem, which are also called state indicators, while performance indicators provide information on how much of the service is actually provided and/or used (van Oudenhoven *et al.*, 2012).

A quantitative review of 153 regional ES case studies by Seppelt *et al.* (2011) concluded by highlighting four aspects that will help to ensure the scientific quality and holistic approach of further ES studies: (a) biophysical realism of ecosystem data and models; (b) consideration of local trade-offs; (c) recognition of off-site effects (i.e. ES provision at different scales); and (d) comprehensive but critical involvement of stakeholders in assessment studies. Seppelt *et al.* (2012) have thereafter developed a blueprint for ES assessment clarifying purpose, scope, analysis, recommendations and monitoring and as such allowing comparison and synthesis of the results of ecosystem assessments.

There is a huge amount of research on mapping ES and the variety of approaches has triggered several review papers of these methodologies (e.g. Burkhard *et al.*, 2009; Eigenbrod *et al.*, 2010; Maes *et al.*, 2012; Crossman *et al.*, 2013). A review by Maes *et al.* (2012) reveals that while provisioning ES can easily be quantified and mapped directly, most regulating, supporting and cultural services are more difficult to map and require proxies for their quantification. Additionally, they claim that the connection between ecosystem status and the services they deliver is still poorly explored. A recent special issue of the journal 'Ecosystem Services' has presented the latest methods in modelling and mapping ES and their application to science, policy and practical decision making (Burkhard *et al.*, 2013). Crossman *et al.* (2013) present a blueprint for mapping and modelling ES in order to provide a template and checklist of information needed. They promote the mapping as a "useful tool for illustrating and quantifying the spatial mismatch between ES delivery and demand that can then be used for communication and to support decision making" (p. 4). Crossman *et al.* (2013) compare two recent reviews by Martínez-Harms and Balvanera (2012) and Egoh *et al.* (2012) with their own review and reveal key aspects of approaches used for mapping ES. Bastian *et al.* (2013) include 'ecosystem potentials' (regarded as stocks of ES, while the services themselves represent the actual flows) in their mapping approach, which is considered a more normative ascertaining of the potential use of particular services.

For RECARE, it is uncertain to what extent ES mapping is the right approach for monitoring ES, as the case studies are working at the local scale. The above discussed mapping approaches are mostly used at national or even continental scale. Additionally, they are often in support of decision making for changes in land use rather than land management, as required in RECARE. However, mapping ES might be used as a complementary tool in RECARE.

There are only few studies quantifying and measuring ES specifically related to soil. Schulte *et al.* (2014) suggest working with five soil functions, which in RECARE we would consider ES: (i) Production of food, fibre and (bio) fuel; (ii) Water purification; (iii) Carbon sequestration; (iv) Habitat for biodiversity and (iv) Recycling of (external) nutrients/agro-chemicals. Schulte *et al.* (2014) admit that this categorization of soil functions should be refined or expanded on. A preliminary method for the quantification of soil quality indicators on arable farms was developed by Rutgers *et al.* (2012). Through scoring of various ES indicators by land users and experts for their importance and informative value respectively, they obtained a final indicative value for each indicator. This differs from valuing ES (see section 13.5 below), as it is considered a preliminary step before assessing the actual provision of the service (which itself might be compared to a maximum ecological potential and thus results in an ES performance index, as in Rutgers *et al.*, 2012). Another effort to develop a method for the quantification of soil services was undertaken by Dominati *et al.* (2014), who worked with a comprehensive list of proxies for each service and its measuring unit. Unfortunately, cultural services were not considered due to their non-biophysical nature and the challenge to quantify. The use of proxies is often inevitable, but requires careful consideration. A study by Eigenbrod *et al.* (2010) has compared primary data for biodiversity, recreation and carbon storage in the UK with land cover based proxies and found a poor data fit and potentially large errors associated with proxy data. They recommend investment in survey efforts rather than to use poor quality proxy data and that surveys can be more cost-effective in the end.

When it comes to land management, it is important to note that it can directly influence ecosystem properties, and functions and services. Van Oudenhoven *et al.* (2012) applied the stepwise cascade-model of Haines-Young and Potschin (2010) to an example from the Netherlands, assessing land management effects without confusing between ecosystem properties, functions and services and thus avoiding double-counting. They confirmed that function indicators are a "subset or combination of ecosystem property indicators, as earlier suggested by Kienast *et al.* (2009)" (van Oudenhoven *et al.*, 2012, p. 118).

Due to methodological challenges, cultural ES are only roughly included in ES assessments, although many authors underline the importance of these immaterial benefits, especially those of cultural landscapes (Plieninger *et al.*, 2013; Chan *et al.*, 2012). Plieninger *et al.* (2013) stressed that spatially explicit information on cultural ES, as perceived by the local populations, provides the basis for the development of sustainable land management strategies, including biodiversity conservation and cultural heritage preservation, and thereby fostering multifunctionality. A review of 107 publications revealed emerging themes in cultural ES research: these relate to improving methods for cultural ES valuation, studying cultural ES in the context of 'ES bundles', and more clearly articulating policy implications (Milcu *et al.*, 2013).

Work done in the UK by Kenter *et al.* (2014) suggests that analysis of cultural ES can be developed using quantitative indicators drawing on publically available datasets, such as surveys of recreation usage. They also emphasise the importance of participatory and interpretative research techniques developed in the social sciences to assess and understand cultural ES in location- and community-based contexts. Such approaches may involve surveying people about their general values and attitudes towards cultural ES, through the use of interviews and focus group discussions. They may also involve the use of deliberative and dialogue-based methods of research, such as extended in-depth discussion groups and mapping methods.

13.5 Valuing ES

The ES concept is intrinsically connected to values, i.e. providing a link between the supply of nature's goods and services and how it is valued by society. Much emphasis has been put on valuing ES to demonstrate that markets fail to adequately capture the full value put of ES by society and hence are often co-driving the degradation of ecosystems. The large body on ES valuation has consistently shown that non-market values nearly always outweigh market values (e.g. Ananda and Herath, 2003; Shiferaw and Holden, 1999), although ways in which the latter are derived are often contested. If we accept the importance of non-market values (whether they can be appropriately assessed or not), it is clear that environmental management decisions should not be based solely on the market value of ES. To support more informed decisions, three research traditions exist on valuing ESSs:

(i) One school emphasises the need to convert all values in monetary figures. Although mindful of various shortcomings, the rationale is that the likelihood of decision-makers and policy makers appreciating the full value of nature is larger when confronted with a single figure for total economic value of ES. For soils this is more difficult than for others, hence its significance is underplayed. Important examples include the Costanza *et al.* (1997) value of Earth's natural capital, and the TEEB initiative and the establishment of an Ecosystem Service Value Database (ESVD) (de Groot *et al.*, 2012).

(ii) A second school regards markets as inherently unsuitable to value nature and objects for expressing ecosystem value in monetary terms (e.g. Sagoff, 2008). Essentially, decisions will need to take into account different value systems and multiple criteria to assess value. Any attempt to capture value in monetary terms reduces the dimensions that need to be taken into account for sustainability (also referred to as "weak sustainability" – see e.g. Ayres *et al.*, 2001).

(iii) A third school focuses more on the operational difficulties to maximise the value of ES as managing land for one (bundle of) ES will often imply the need to sacrifice value derived from some other ES, i.e. there are trade-offs between different ES. The ES concept is well-suited to the study of such trade-offs. An important initiative taking this paradigm is the Natural Capital project, and the InVEST methodology it has developed (Kareiva *et al.*, 2011).

(iv) A fourth school is emerging that has an even stronger focus on values rather than valuation and thus provides an extension of schools 2 and 3 above. In this school, ES are seen as part of the social-ecological system (SES) (Folke, 2006; Olsson *et al.*, 2004). The values associated with ecological knowledge and understanding play an important part in the stock of ES as do the social networks associated with them. This is seen as being important for developing resilience within SES and ES (CGIAR Research Program on Water, Land and Ecosystems (WLE), 2014).

In ecological economics, a large volume of literature exists on valuation of ecosystems. The alternative 'types' of value can be classified into 'intrinsic', 'anthropocentric', and 'utilitarian and deontological'. Economic valuation is based on an anthropocentric approach and it defines value based on individual preferences. This approach typically sits within the first school indicated above. The Total Economic Value (TEV) framework captures the benefits derived from the ecosystem services. The TEV for any resource is the sum of use and non-use values (Figure 13.3).

'Use value' involves interaction with the resource and is subdivided into direct use and indirect use value. Direct use value relates to the use of natural resources in a consumptive (e.g. industrial water abstraction) or in a non-consumptive manner (e.g. tourism). With an ES perspective, 'direct use' values are often associated with provisioning and cultural ES. 'Indirect use' value relates to the role of natural resources in providing or supporting key ecosystem services (e.g. nutrient cycling, climate regulation, habitat provision). In the ES terminology, indirect use values are frequently applicable to regulation ES.

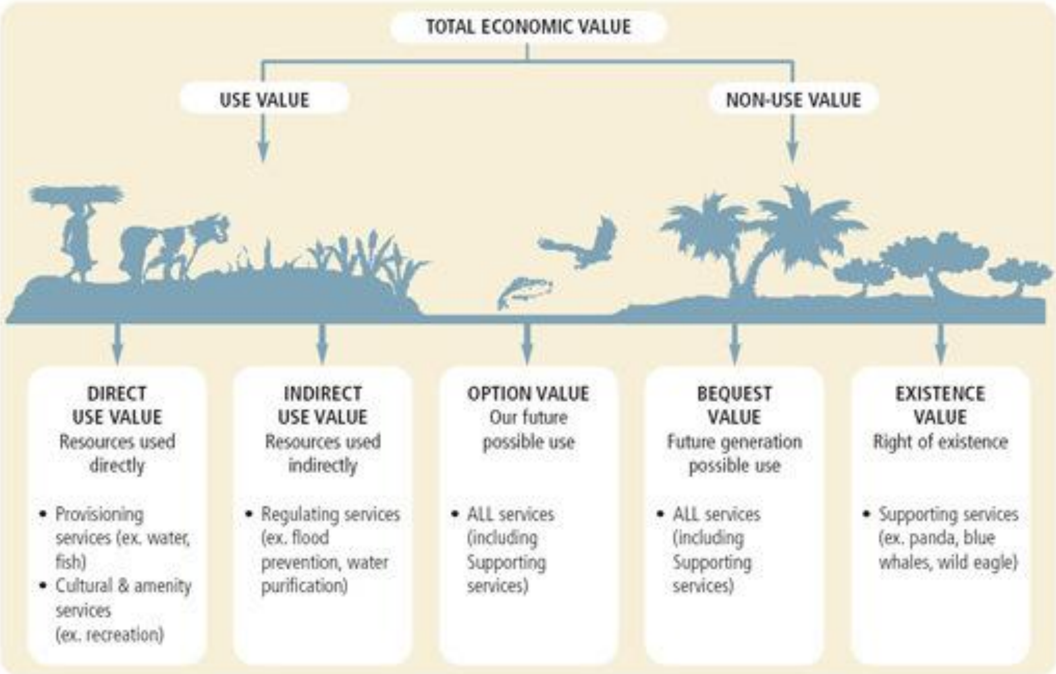

Figure 13.3: Decomposition of the Total Economic Value (TEV) of ecosystems (Smith et al., 2006).

'Non-use value' is associated with benefits derived from the knowledge that the natural resources and aspects of the natural environment are maintained. Non-use value can be split into two parts: (a) bequest value (associated with the knowledge that the area as a resource will be passed on to future generations), and (b) existence value (derived from the satisfaction of the knowledge that resources continues to exist, regardless of use made of it now or in the future) (Figure 3), while others distinguish also a third type of non-use value: the altruistic value (derived from the knowledge that contemporaries can enjoy the goods and services related to the area) (Hein, 2010; Kolstad, 2000). Option value can be both use or non-use value and it is not associated with the current use of resources but the benefit of keeping open the option to make use of them in the future. With regard to valuing nature, there has been particularly much debate on valid components and assessment methodologies to assess non-use values. Mainstreaming of the ES concept has partially solved some of the debates by offering a clear framework to link ecosystem functioning and human wellbeing (see Section 13.3.2). However, significant challenges remain, e.g. with regard to the risks of double-counting, appropriate assessment methods for the valuation of particular (bundles of) ES, and challenges to capture the short- and long -term spatial and temporal dynamics of ES.

In valuing ES, it is important to base this on common denominations of area, time and if applicable currency units (e.g. international dollars per ha per year) (de Groot *et al.*, 2012). Within the TEV framework, values are derived from information of individual preferences provided by market transactions that are related directly to ecosystem services. For example, some ecosystem services that are provided by natural resources have market values that reveal information about their economic value. Many uses and services provided by ecosystems are not traded in markets and are consequently 'non-market' goods. For these non-market goods, price information must be derived from parallel markets that are associated indirectly with the good to be valued. In the absence of both direct and indirect price information on ecosystem services, hypothetical markets might be created to elicit values. The valuation approaches that have been developed to estimate the economic value of ecosystem services are: (a) direct market valuation methods, (b) revealed preference methods, and (c) stated preference methods (Chee, 2004).

Direct market valuation methods are distinguished into three main approaches (a) market price-based approaches, (b) cost-based approaches, and (c) approaches based on production functions. These approaches are based on individuals' preferences and costs using data from actual markets. Market price-based approaches are used to obtain the value of provisioning services. Cost-based methods are based on the cost of avoiding damages due to lost services, the cost of replacing ecosystem services and the cost of providing substitute services (King and Mazzotta, 2000). Production function-based approaches aim to measure how the indirect use values provided through changes in ecosystem services enhance the productivity of economic activities (Pattanayak and Kramer, 2001).

'Revealed preference' techniques rely on the observation of individual preferences for a marketable good that is related to ecosystem services. Revealed preference methods are distinguished into market-based and surrogate markets related. Surrogate markets include travel cost (TC) method and hedonic pricing (HP). The travel cost method estimates the economic value of visiting recreational sites with specific environmental attributes including specific levels of ecosystem services. The hedonic pricing approach uses information on the implicit demand of the environmental attributes of market goods, e.g. price that people pay for properties within specific environmental attributes.

Stated preference methods use questionnaires to elicit individuals' preferences for changes in the provision of ecosystem services. Stated preference methods can be used to estimate both use and non-use values of ecosystems. These approaches include contingent valuation method (CVM) and choice experiment (CE). The contingent valuation method is a survey-based approach to value ecosystem services. The approach is based on the development of a hypothetical market in which respondents directly state their willingness to enhance the provision of an ecosystem service, or alternatively, their willingness to accept for its loss. Choice experiments are based on the notion that services can be described in terms of attributes and the levels that these attributes take. Respondents are presented with different combinations of these attributes and are asked to rank their preferences in order (Birol and Koundouri, 2008). However, gathering primary, site-specific data is costly and as a result, a popular alternative method is to conduct a "benefit transfer" (Plummer, 2009). The benefit transfer method is used to estimate economic values for ecosystem services by transferring information from existing studies in another location and/or context.

Given the complexity of the issues being discussed all of the methods outlined thus far have been criticised for being too hypothetical (Getzner *et al.*, 2005). There is now a move to develop more deliberative valuation techniques that allow for more open and potentially more grounded outputs by combining the stated preference approach with increased deliberation between experts and/or users. The outcomes are more culturally constructed and richer from a contextual perspective and able to consider a wider range of ES within any valuation.

The economic literature on valuing ES is largely based on individual preferences with limited incorporation of shared and cultural values. Kenter *et al.*, (2014) reviewing non-economic literature identified values considered to be transcendental, based on ethics and normative beliefs which are part of individual and community identity (cultural values), and act as guiding principles that transcend specific situations and are relatively stable (Schwartz and Bilsky, 1987). Also there are contextual values, which are based on opinions about the worth of something and hence are more allied to attitudes and preferences (Dietz *et al.*, 2005). Both these values, for example, can be important in understanding resistance to changing land management practices.

Whilst monetary valuation is important in understanding individual values, Kenter *et al.*, (2014) also suggest that to provide a comprehensive valuation other approaches are required to elicit the multiple dimensions of cultural values and to "translate deeper-held transcendental values into contextual values and preferences". They suggest that psychometric, non-analytic and interpretive methods using interviews and group discussions can help reveal those shared values. They can then be combined with deliberative-analytical methods, such as deliberative monetary valuation and multi-criteria analysis, which can express the outcome in monetary terms or as a quantitative ranking or rating (Fish *et al.*, 2011).

For the RECARE project to work on valuing ES, three aspects may help design an appropriate strategy:

(i) When undertaking a valuation, it is first of all fundamental to establish what the valuation is for (cf. Robinson *et al.*, 2014). This is likely to relate to the design, application and evaluation of improved (sustainable) land management technologies, which may affect several but not all soil-based ES.

Understanding which ES will be affected will reduce the complexity of the valuation exercise. Valuation will hence need to focus on comparing situations without and with Sustainable Land Management (SLM) options. Difficulties that may remain are that what is good soil quality, or sustainable management may depend on the specific context under consideration. Establishing indicators and threshold values below which the provisioning of certain ES is compromised may be helpful here (Robinson *et al.*, 2014). Special attention may need to be given to spatial and temporal variations (e.g. inter-annual variation) in the provisioning of ES by certain SLM measures (cf. Schipanski *et al.*, 2014; or Fleskens, 2012).

(ii) Given the complex and multiple contributions that soils make to ES especially regulating, provisioning and cultural services, it seems sensible to adopt some of the more innovative deliberative approaches to valuation. Such deliberative valuations techniques might include combining a stated preference technique with further ordering and participative mapping in focus groups (Malovics and Kelemen, 2009; Martín-López *et al.*, 2014), reports and recommendations from citizen juries (Getzner *et al.*, 2005) and expert/user deliberation to provide Deliberative Monetary Valuation (DMV). Given the inter-disciplinary nature of the research team on RECARE and the number of case studies involved the latter would seem to be the most logical way forward.

(iii) The focus on soil-based ES
In the above, we have laid out how valuation of ES can be approached within RECARE. Below we indicate how such valuations can be incorporated in a number of economic tools. As such tools also allow alternative, non-monetary, valuations (i.e. accommodating Schools 2 and 4) and allow comparisons based on multiple attributes (i.e. accommodating Schools 3 and 4), depending on the valuation context and stakeholder preferences, they are briefly introduced below.

The impacts of the changes in the provision of ecosystem services expressed in monetary terms can be encompassed in integrated economic tools such as cost-benefit analysis (CBA), cost-effectiveness analysis (CEA), multi-criteria analysis (MCA) and Deliberative Monetary Valuation (DMV) to evaluate policy options (e.g. prevention measures) and inform policy designers. CBA evaluates the social profitability of a measure by assessing its monetary social costs and benefits over a time period. A measure is deemed to be profitable if total benefits exceed total costs. CEA is a technique that enables comparison between different kinds of interventions with similar effects on the basis of the cost per unit achieved. CEA relates the costs of a measure to its key benefits, while CBA attempts to compare costs with the monetary value of the measures benefits. According to Turner *et al.* (2010) "the choice between CBA and CEA is determined by the nature of the policy problem under scrutiny". CEA is most useful if the objective is to find the least cost way to meet some environmental standards or achieve a target or in cases where major outcomes are either intangible or difficult to monetize. CBA is the most appropriate evaluation tool when comparing alternatives policy options to see which one achieves the greatest benefit to society or when analysing a single policy option to determine whether the total benefits to society exceed the costs. The major weakness with CBA is the difficulty to place values on all costs and benefits. MCA addresses interdisciplinary and complex environmental issues by combining economic, ecologic and social criteria (Khalili and Duecker, 2013). Multi-criteria decision analysis (MCDA) is a useful tool in the decision making process when a discrete number of alternatives is given (Busch *et al.*, 2012). MCDA takes into account policy intervention impacts that are not easily given monetary values or when there is a large amount of complex information and it can be used to identify the most preferred alternative and to rank alternatives against each other. One of the difficulties of evaluating options using the MCA approach (and CVM and CE) is that participants may not be knowledgeable enough about soil ES to make informed decisions. One possible way of overcoming this issue is to use a deliberated approach. DMV combines techniques such as stated preference with deliberation. So for example a contingent valuation method (CVM) might be used to generate a 'willingness to pay' valuation. The outputs from the CVM survey are then discussed and adjusted in a deliberative setting amongst experts and/or users of the identified ES. The result is a monetary valuation that is extended through deliberation to validate the outcomes and extend to include non-monetary aspects through the inclusion of shared knowledge and further exploration of shared values. Furthermore, there is some evidence that participants feel more confident about their deliberated values in MCA and DMV workshops compared to their individual values expressed in a survey (Kenter *et al.*, 2014).

13.6 Analysis of the operationalization of the soil ES concept in European research projects

There is a need to understand impacts of soil threats to ES. The aim of the following analysis is to examine the current extent to which such understandings are being sought in Europe. This analysis will highlight gaps in research that will need to be fulfilled, if soils are to be adequately reflected in ES management. A previous systematic review by Vihervaara *et al.* (2010) showed that in publications up to 2008, the ES concept had been under-explored in relation to soil quality and regulation compared with biodiversity; and in agricultural systems compared with watersheds and forestry, due to the roots of the ES concept (see section 3.1). This review, therefore, zooms into the topic of soil and examines current and recent research projects, particularly post-2008. It also focuses on Europe to ensure coherence with the Millennium Ecosystem Assessment, and demonstrate the extent to which such frameworks are being applied to soil systems.

In order to identify relevant research projects, a rapid systematic review approach was employed; the approach may miss some projects (e.g. those dealing with a specific ES without mentioning the term 'ecosystem services'), but was intended to be as efficient as possible while providing an extensive overview. The projects identified were therefore considered to be a good representation of the current state of research. The approach began with a search of Scopus. The key words 'ecosystem services' and 'soils' were used, and then the results were filtered for 'Europe'. This produced a list of 1,137 results. Using titles and abstracts, the list was then narrowed down to 200 papers by excluding those that did not match the combination of the three search criteria. The large reduction is due largely to those papers that examined non-soil ES and/or were not in Europe. Of those papers that remained, the text and acknowledgments were scanned for mention of the projects that supported or funded the research. Fifty identified projects were listed.

An internet search was then conducted for each project, locating website and any relevant project documentation. Using the information available, the projects were then compared and contrasted in order to identify characteristics that could be used to categorise and compare them. A table was constructed of each project and its characteristic under each identified category; these categories and characteristics are explained in the results (Annex I).

The broadest way to categorise the projects is by the way in which they frame soil-based ES. A small number of projects focused specifically on soil ES. These are highlighted in red in Annex I. These projects examine certain soil processes or characteristics as the final ES or endpoint. Examples include the SOIL SERVICE project that explicitly focuses on soil biodiversity as an ES, or SoilTrEc, which focuses on soil processes in river catchments. Other projects include soil ES more implicitly in their research (highlighted in yellow). In this way, they are considered as intermediary ES, contributing to the focus ES of the project. Many of these (e.g. RUBICODE, MULTAGRI, LIBERATION) have biodiversity as their focus, with soil included through its potential impact to biodiversity. Some projects form a hybrid, as highlighted by orange in the table.

The soil-focussed projects are usually large consortia funded by grants from the European Commission or similar international funding agency. These projects are split into multiple work-packages or sub-projects, and are interdisciplinary, studying multiple aspects of one particular overarching problem. Of the twenty-one identified projects that are such large consortia, two are soil focussed projects and the others were biodiversity or other ES focussed research. There were also a number of projects funded by national funding agencies to establish nationally-focussed research (e.g. MOUNTLAND) or small research centres (e.g. FuturES). These tended to have quite a broad ES focus, and so were in the hybrid category. There were a number of individual fellowships, though there was often insufficient information to really explore their content and focus.

A number of the projects could be described as 'baseline' projects that seek to characterise ES and understand their relationships. These are projects that monitor ES, observing changes or impacts of changes on benefits or other ES. In particular, this category of projects examines the impacts to ES from a range of environmental changes, including for example climate change, deforestation or flooding. In sum, these projects are building an understanding of which services exist, how they are linked or bundled through benefits, and therefore what trade-offs and gains are to be made in prioritising certain services. Much of the soil-focussed research falls into this category.

Projects that build upon this baseline by studying the impact of management interventions on ES can be called 'management' projects. Such management interventions are usually physical changes, such as planting to reduce erosion. Often such projects contribute to 'baselines' by monitoring the ES under the proposed intervention. Most of the projects in this category are those that target biodiversity as an ES, for example MULTAGRI, AGFORWARD. They are also predominantly focussed on agricultural land, and as such, there is an implicit inclusion of soil ES, though this is not often examined.

Some projects can be characterised as decision making and policy research, i.e. seeking to aid in the promotion of 'successful' ES management. These projects often seek to design tools to aid in decision making around land use, for example LandSFACTS. Projects may also propose a range of policy responses to promote the uptake of ES management initiatives, or to prevent the damage of ES. A subset in this category are those that explicitly pursue payments for ES through the valuation of ES. This category is dominated by projects that do not have soil ES as an explicit focus.

This mini-review has highlighted a research gap in creating policy and management for soil ES. Research projects that focus on soil ES are primarily concerned with establishing a baseline to understand and characterise such ES. In this way, soil research is less well developed (Vihervaara *et al.*, 2010). However, promising baseline knowledge is being created in order to develop management and policy approaches. This baseline is being further supplemented by research that examines soil ES as intermediary services to end services such as biodiversity. These projects implicitly include soil ES and in doing so often contribute to understanding the status and baselines of such services. In addition, by tying soil into other services that are tangible and of popular concern, soil research can benefit from the interdisciplinary, interconnected nature of ES.

13.7 Adapted soil functions and ecosystem services framework for RECARE

Although many ES frameworks have evolved over time as presented in the above sections, choosing an appropriate framework for the purposes of RECARE remains challenging. RECARE aims to assess the various effects on soil functions and ES caused by soil threats as well as prevention/remediation measures, and more over has the objective to do so at various spatial scales. It plans to make use of the ES concept to communicate with local stakeholders in order to identify the most beneficial land management measures and with national and European policy makers to identify trade-offs and win-win situations resulting from and/or impacted by European policies. The framework thus needs to reflect/respect the specific contributions of soils to ES and also distinguish changes in ES due to soil management and policies impacting on soil, while at the same time be simple and robust for practical application with stakeholders at various levels. It should serve the needs of those work packages that make use of the ES concept, especially within the following tasks:

Task 2.3: *Soil functions and ecosystem services*
Task 3.3: *Development of a harmonized universal methodology* to assess the state of soil degradation and conservation
Task 6.3: *Quantitative assessment of effectiveness of the WP5-selected measure:* input data for the assessment of soil functions and ecosystem services performed in WP7.1
Task 7.1: *Impact assessment on ecosystem services*
Task 4.3: *Stakeholder valuation of ecosystem services*
Task 8.2: *Upscale Case Study results to European level using modelling.*

The activities and outcomes of these tasks need to refer to one common ES framework and thus an agreed terminology in order to truly build on each other and produce sound results. For example, WP6 requires a selection of soil threat indicators identified in WP2/WP3 in order to assess the effects of the implemented remediation measures. WP7/WP8 will then build on that work and create meaningful composite indicators in order to get a comprehensive appraisal of the prevention/remediation impact on the various soil functions and ES.

From the review of ES frameworks in section 13.3.2 it becomes evident that none of the existing frameworks fully suits these requirements of RECARE. We see the following three major challenges for working with and thus adapting the ES framework within the RECARE project:

- The need to link ES to soils as well as to Sustainable Land Management (SLM)
- Use the framework together with stakeholders in order to assess and value the services provided by and changed through SLM (in order to mitigate soil threats)
- Be simple but scientifically correct.

We consider these combined challenges as the research gap which we aim to close as much as possible. We have therefore adapted existing ES frameworks, mainly the one from Braat and de Groot (2012) with elements from more soil-oriented recent suggestions such as Dominati *et al.* (2014), while trying to introduce a consistency of terms understandable by stakeholders. With this, we are in line with suggestions from authors like Bouma, opting for soil scientists to become more effective in transdisciplinary approaches, such as to achieve the UN Sustainable Development Goals (SG's) (Bouma, 2014). In RECARE, we suggest thus to use the adapted ES framework as presented in Figure. 4. We have used the following elements from existing frameworks:

- MEA (2005): major categories of ES
- TEEB (2010): subcategories of ES, but adapted and simplified
- Haines-Young and Potschin (2010): cascade model
- Braat and de Groot (2012): main model structure and feedback loops in TEEB model
- CICES (2013): only indirectly. The idea is to translate TEEB into CICES, (see Maes *et al.*, 2013)
- SmartSOIL (Glenk *et al.*, 2012): soil processes, benefits
- Van Oudenhoven *et al.* (2012): land management, driving forces, societal response
- Dominati *et al.* (2014): Natural capital with inherent and manageable properties of soil; external drivers as 'other driving forces', degradation processes as 'soil threats'

Similar to many ES frameworks the RECARE framework distinguishes between an ecosystem and human well-being part. As the RECARE project is on soil threats, this is the starting point on the ecosystem part of the framework. Soil threats affect natural capital such as soil, water, vegetation, air and animals, and are in turn influenced by those. Within the natural capital, the RECARE framework focuses especially on soil and its properties, classified in inherent and manageable properties. The natural capital then enables and underpins soil processes, while at the same time being affected by those. Soil processes finally are the ecosystem's capacity to provide services, thus they support the provision of soil functions and ES. ES may be utilized to produce benefits for individuals and human society. Those benefits are explicitly or implicitly valued by individuals and human society. The values put to those benefits influence policy and decision-making and thus lead to a societal response. Individual (e.g. farmers') and societal decision making and policy determine land management and other (human) driving forces, which again affect soil threats and natural capital.

For example soil erosion (soil threat) leads amongst others to reduced soil organic matter content in the topsoil (natural capital), which affects soil organic matter cycling (soil process). This may result in a decreased production of biomass (soil function and ES) and thereby poor crops harvest (benefit). The loss in crop harvest is negatively valued by human society thus ideally leads to a stronger legislation to protect soil against erosion.

The RECARE ES framework presented here is still a draft and will further be developed based on feedback from RECARE partners and other contributors.

The RECARE framework also relates to the DPSIR framework (Smeets and Weterings, 1999), by showing the driving forces (*driver*) impacting on land management as the pressure on soil resources, manifested through soil threats (*pressure*). These change the conditions of the natural capital (*status*) and leads to impacts on ES (*impact 1*) and human well-being (*impact 2*). In response to both of these, society either changes its policy and decision making, or land users directly adapt their land management (*response*). See also Müller and Burkhard (2012) who suggest a similar link of the ES and DPSIR framework within an indicator-based perspective. In order to improve ES with SLM, the services need to be "manageable" for the stakeholders. A small study in Australia assessed farmers' perceived ability to manage ES (Smith and Sullivan, 2014). Only soil health and shade/shelter were indicated as being highly manageable, with high convergence in views. While shade/shelter was a specific issue of the area, soil health was the only ES where farmers indicated being highly vulnerable to its loss, while at the same time being able to influence it themselves.

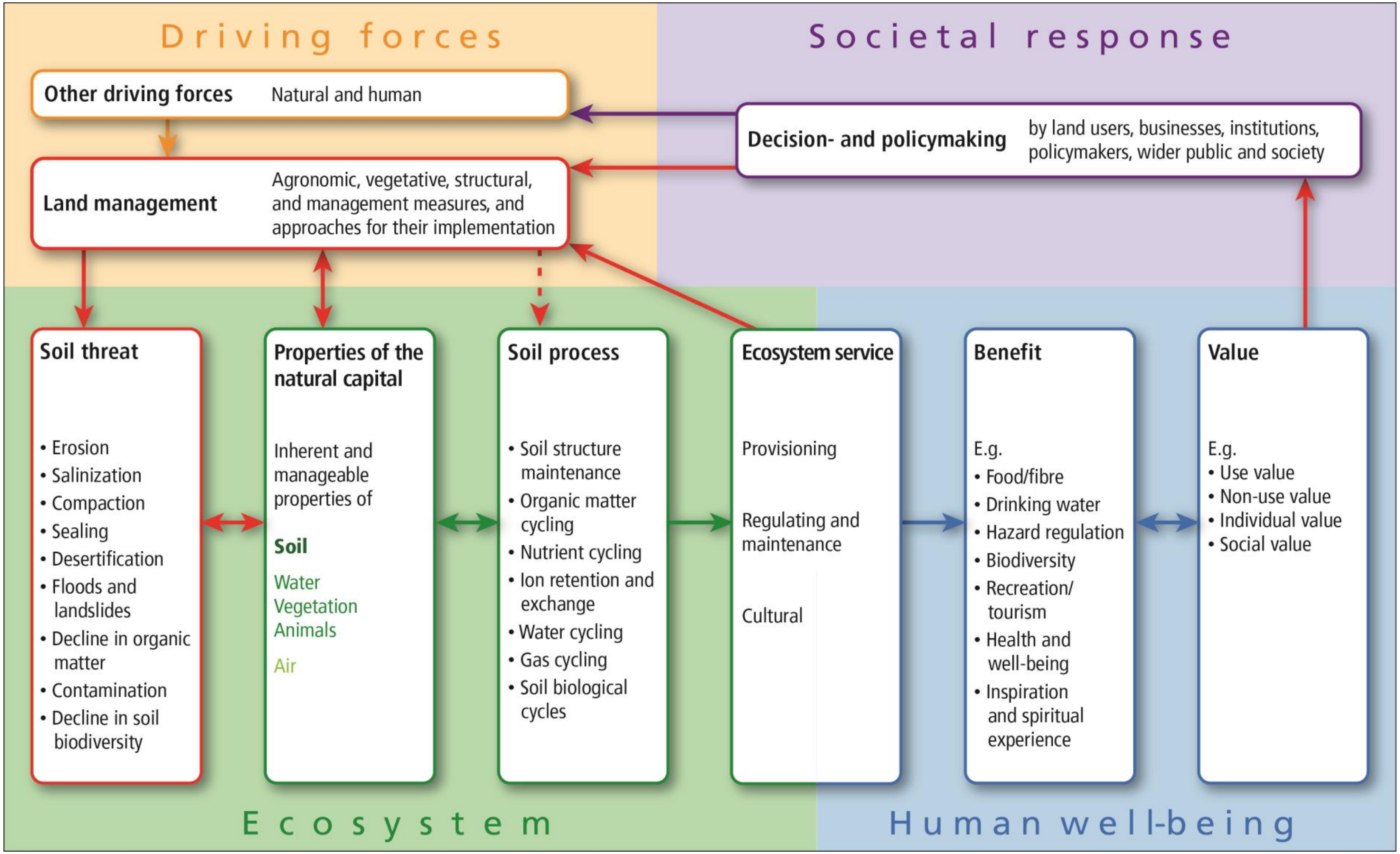

Figure 13.4: Proposed ES Framework for RECARE.

To measure the desired and achieved improvements in ES and thus in their underlying soil functions, indicators need to be identified. The previous chapters of this review present these indicators for each soil threat separately. Effects of soil threats and remediation measures are thus captured by key soil properties as well as through bio-physical (e.g. reduced soil loss) and socio-economic (e.g. reduced workload) impact indicators. In order to use such indicators in RECARE, it should be possible to associate the changes in soil functions to impacts of prevention/remediation measures (SLM). This requires the indicators to be sensitive enough to small changes, but still sufficiently robust to proof the change and associate it to SLM.

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14 SYNERGY

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14.1 The state of soil degradation in Europe

The chapters on soil threats give an overview of the geographically extent of the soil threat and in some cases on its severness. In addition to European wide information (where available), some chapters highlighted regional studies on soil threats.

Soil erosion by water identifies three regions in Europe with a different status of the threat: a southern zone with severe risk, a northern loess zone with moderate risk and an eastern zone with an overlap of both of these zones. However, the authors also recognize that within these zones, hot spots occur. *Soil erosion by wind* occurs mainly in the northern parts of Germany, eastern Netherlands, eastern England and the Iberian Peninsula. The authors indicate that a comprehensive knowledge about where and when wind erosion occurs in Europe is lacking. *Decline in SOM in peatsoils* is a major degradation process in northern Europe, whereas *decline in SOM in mineral* soils is a European wide degradation process. Based on the calculation of the Relative Normalized Density, risk of *soil compaction* proves to be most severe in northern and central Europe. *Soil sealing,* unsurprisingly, occurs in the densely populated areas of Europe, with a focus on central and west Europe. For *soil contamination*, we used the identified number of contaminated sites per country to visualize the spread of contamination (Panagos *et al.*, 2013). For emerging pollutants, no geographical reference is yet known. The regional spread of soil contamination through pesticides and herbicides is also not known, though figures about herbicide applications at the European level are available. *Soil salinization* mainly occurs in the southern part of Europe, and partly in the Balkan region. Parts of central, eastern and southern Europe are sensitive to the risl of *desertification*, based on a mapping exercise using soil quality, climate and vegetation parameters. *Flooding* has been reported along the major rivers in Europe, whereas risks of *landslides* are mainly localized based on topography (mountain areas). We constructed overlay maps of Europe presenting the localization of each threat for 10-km^2 cells. The maps show areas of low (Fig. 14.1), low and moderate accumulated (Fig 14.2), as well as low, moderate and high accumulated (Fig. 14.3) levels of soil threats. Weighting was done by giving the low, moderate and high threshold values a weighing factor of 1, 2 and 3 respectively. These numbers were summarized for each grid.

Included in the maps are erosion by water (PESERA) (t h^{-1} yr^{-1}), landslide susceptibility, biodiversity functions (risk), wind erosion susceptibility, carbon emissions from peat soil (ton per country), topsoil organic carbon in mineral soil (%), susceptibility to compaction, salinization (% of area), degree of soil sealing, sensitivity to desertification, flood damage potential (Purchasing Power Parities, PPPs) and contamination. The latter is based on the number of identified contaminated sites per country (Panagos *et al.*, 2013). Mineral soils were delimited by low organic carbon content of < 12%. Threshold values for the different soil threat levels were defined for all threat categories (Table 14.1) and summarized for each 10-km^2 grid. Organic carbon losses for peat soil and contaminated soils are included countrywide, since information on these are given at this scale.

The underlying soil threat maps originate mainly from the European soil portal (European Commission - Joint Research Centre), with the exception for soil sealing, desertification and flooding. These where gathered from the CORINE Land Cover Database (European Environment Agency), the DISMED Project (European Environment Agency) and the Floods Portal (European Commission - Joint Research Centre). An overview of the sources for the maps is given in Table 14.2.

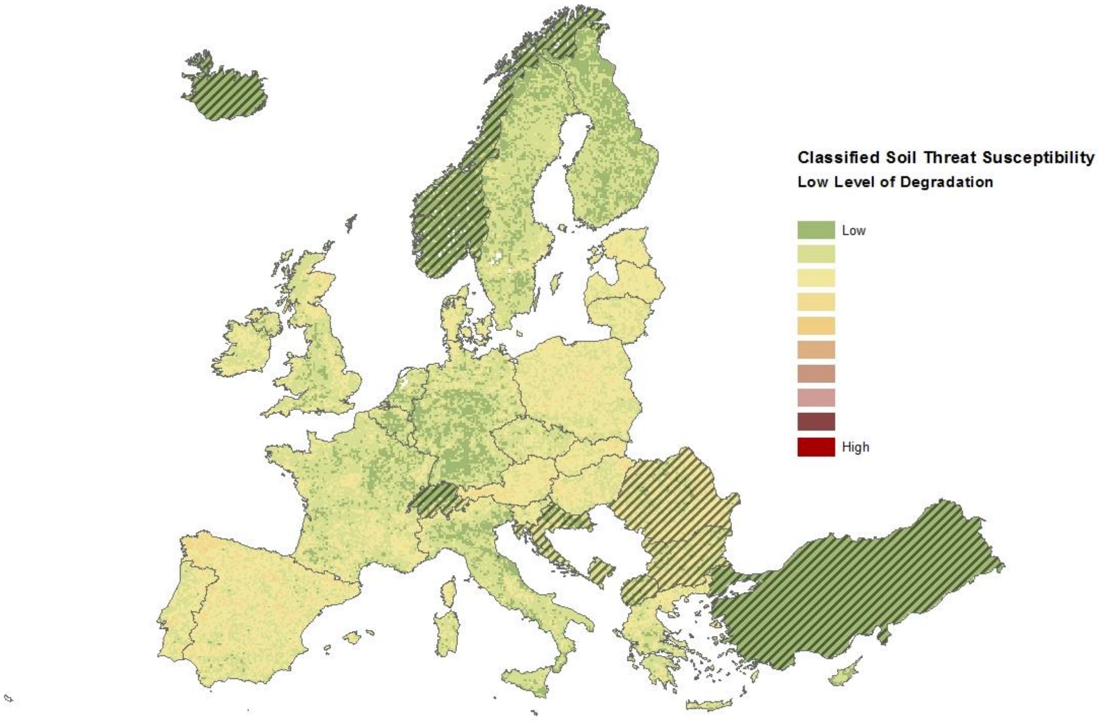

Figure 14.1: Soil threat map of Europe for the low category of degradation. For the shaded areas, not all threats are mapped.

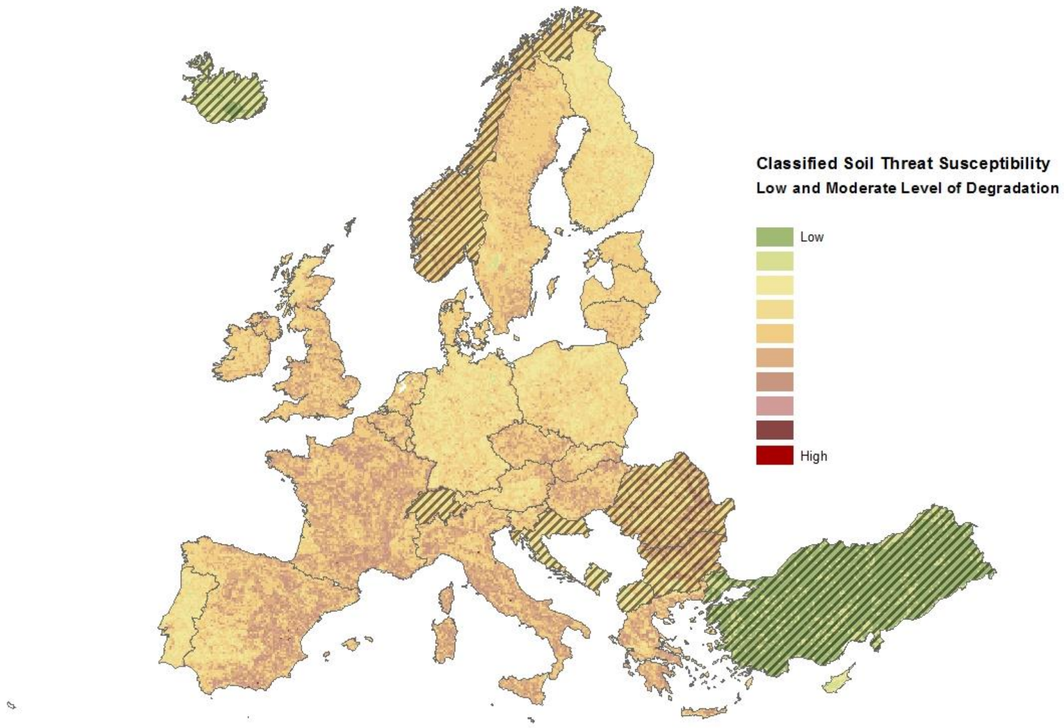

Figure 14.2: Soil threat map of Europe, summarized for the low (weighing coefficient 1) and moderate (weighing coefficient 2) category of degradation. For the shaded areas, not all threats are mapped.

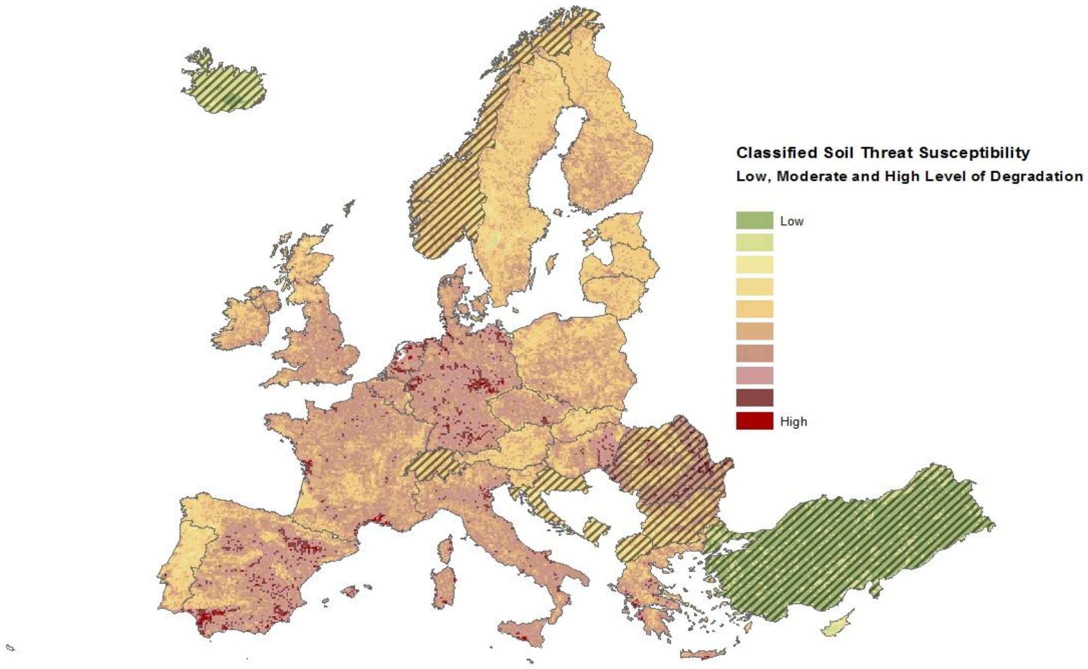

Figure 14.3: Soil threat map of Europe summarized for the low (weighing coefficient 1), moderate (weighing coefficient 2) and high (weighing coefficient 3) category of degradation. For the shaded areas, not all threats are mapped.

Table 14.1: Threshold values for the low, moderate and high classes of soil threats

	Erosion by Water (t/h/yr)	**Landslides Susceptibility [a]**	**Wind Erosion Susceptibility [b]**	**Organic Carbon (%) in mineral soils**	**C emissions from peat soil (ton C per country)**	**Susceptibility to Compaction[c]**
Low	1 - 2	Very low-Low	Very low-Low	>3	1 - 500000	Low
Moderate	3 - 10	Moderate	Moderate	>1 - <=3	500001 - 5000000	Medium
High	>10	High-Very high	High	>0 - <=1	>5000000	High-Very high
Comments	Excluding Norway, Sweden, Turkey, Iceland, Switzerland, Montenegro, Macedonia and Croatia	Excluding Turkey, Iceland, Switzerland, Croatia, Makedonia and Montenegro	Excluding Turkey and Iceland	Excluding Turkey and Iceland	All countries included	All countries included
	Salinization (% of area)[d]	**Degree of Soil sealing (%)**	**Sensitivity to Desertification[e]**	**Flood damage potential (Purchasing Power Parities, PPPs)**	**Identified number of contaminated sites**	**Biodiversity functions (risk)**
Low	Potentially salt affected area	>0 - 29	Very low-Low	>0 - <1 000 000	1-1000	0.200 - 0.249
Moderate	Sodic <50 % and Saline <50%	30 - 79	Low to moderate	1 000 000 - 10 000 000	1001-10000	0.250 - 0.3
High	Sodic >50 % and Saline >50%	80 - 100	High to very high	>10 000 000	> 10000	> 0.3
Comments	Excluding Turkey	All countries included	Only Spain, Portugal, south of France, Italy and Greece	Excluding Norway, Turkey, Iceland, Switzerland and Montenegro	Excluding Turkey, Bulgaria and Portugal	Excluding Norway, Iceland, Turkey, Balkan

[a] Values already defined for Landslide Susceptibility (Fig. 11.3)

[b] Values already defined for Wind Erosion Susceptibility (Fig. 3.6)

[c] Values already defined for Compaction (http://eusoils.jrc.ec.europa.eu/library/themes/compaction/Data.html)

[d] Values already defined in map (see http://eusoils.jrc.ec.europa.eu/library/themes/Salinization/Data.html)

[e] Values already defined for Sensitivity to Desertification (Fig. 10.1)

Table 14.2: Sources for the soil threat status calculations

Soil Erosion by water– PESERA: http://eusoils.jrc.ec.europa.eu/ESDB_Archive/pesera/pesera_data.html
Wind Erosion: http://eusoils.jrc.ec.europa.eu/library/themes/erosion/winderosion/
Soil Organic Carbon in peat soils: http://www.wetlands.org/Portals/0/publications/Report/The%20Global%20Peatland%20CO2%20Picture_web%20Aug%202010.pdf
Soil Organic Carbon in mineral soils: http://eusoils.jrc.ec.europa.eu/ESDB_Archive/octop/octop_data.html
Soil Compaction: http://eusoils.jrc.ec.europa.eu/library/themes/compaction/Data.html
Soil Salinization: http://eusoils.jrc.ec.europa.eu/library/themes/Salinization/Data.html
Landslides: http://eusoils.jrc.ec.europa.eu/library/themes/LandSlides/index.html#ELSUS
Soil contamination: http://www.hindawi.com/journals/jeph/2013/158764/
Soil Sealing: http://www.eea.europa.eu/data-and-maps/figures/eea-fast-track-service-precursor
Desertification: http://www.eea.europa.eu/data-and-maps/figures/sensitivity-to-desertification-in-the-northern-mediterranean
Flooding: http://www.floods.jrc.ec.europa.eu
Biodiversity: Source is not yet published, classified information so far.

14.2 The main drivers impact on soil threats

The drivers of climate, policy and human activity have different levels of influence and importance for the various soil threats. For some of the soil threats, like water erosion or flooding and landslides, climate is the most important driver. For threats like sealing and contamination, human activities are the most important drivers. For other threats, a combination of climate and human activities is important.

14.2.1 Climate drivers

Climate can be an active direct driver for the soil threat (influence of temperature, precipitation, wind), but climate can also be an indirect driver, influencing factors important for development of the soil threat. Some of the chapters describe the indirect effects of climate on the soil threat e.g. the chapter about water erosion. A future change in climate can change the conditions for development of the soil threats. This report describes the influence of current climate, but for some of the threats, examples of expected effects of future climate changes are also given (e.g wind erosion chapter). A brief summary of the influence of the climate driver is given here and Table 14.3 illustrates the importance of climate as a driver for the different threats.

Water erosion

Climate, particularly rainfall, is the primary, direct driver of soil erosion by water. Rainfall is a main agent of detachment of soil particles and a source of surface runoff for detachment and transport of eroded material. In cold climate regions, freezing-thawing cycles can also play a key role in detachment and snow melt can be an important additional source of runoff. The erosivity of rainfall is related to the kinetic energy. Large variations occur between and within individual rain storms depending on their origins in terms of synoptic weather conditions (e.g. convectional vs frontal rain) and on wind speed, also influencing the runoff generation. The rainfall-runoff response of soils can be divided in two main runoff generating processes: infiltration-excess overland flow occurs when rainfall intensity exceeds a soil's infiltration capacity; and saturation overland flow occurs when a soil's water storage capacity has been exceeded, typically due to prolonged antecedent rainfall.

Climate can affect soil erosion by water *indirectly*, through its impacts on soil properties, soil cover (natural vegetation/crops) and interactions between these impacts. Soil properties strongly determine a soil's infiltration and storage capacity and thereby its hydrological response. The indirect role of climate can be illustrated by examples: i) the importance of dry spells in the formation of a structural surface crust or in the appearance and severity of soil water repellency ii) freezing and thawing can influence soil properties by increased erodibility and iii) protective plant cover against rainsplash in semi-arid and arid regions can decrease with increasing aridity.

Wind erosion
Climate affects wind erosion by detachment of particles and transport, depending both on the occurrence of wind and precipitation (dry/wet soils). Climatic change can also have a direct impact on wind erosion if it results in stronger or more frequent winds. Climate change can have indirect impacts on wind erosion by influencing plant cover, soil moisture, snow cover and the growing season (plant cover). Reduced precipitation, producing dry conditions for plant cover will increase the risk of wind erosion. Both for water and wind erosion, climate can influence plant cover and thereby have an indirect effect on the erosion processes.

Decline in organic matter in mineral soils
In natural ecosystems, climate is the main driver from the effects of temperature, moisture and solar radiation. Sensitivity of net primary production (NPP) to moisture availability is higher than that of decomposition rates, while the opposite is observed in the case of temperature (Post, 2006). Soil organic matter (SOM) is positively correlated with precipitation and negatively with temperature, explaining the general pattern of decline from northern to southern Europe. Baldock and Nelson (2000) placed land use and management at the top of the ranking of soil-forming factors of SOM content: management>climate>biota >topography=parent material >time. Over long periods, the SOM content varies mainly due to climatic, geological and soil forming factors, but for short periods, vegetation disturbances and land use changes affect the storage (Batjes, 2006).

Decline in organic matter in peat soils
Ciais *et al.* (2010) estimated the C balance of European (EU-25) croplands over the last two decades, and found that it followed the NPP trend, which, in turn, was mainly driven by technological changes (>90%), rather than by climatic and atmospheric CO_2 concentration (<10%). Technological developments have the potential for controlling the C balance but there are uncertainties about the effects of climate change on SOC content. Wu *et al.* (2011) found the expected responses of the C stocks to warming and altered precipitation (i.e. soil respiration was increased by warming and increased precipitation and reduced by decreased precipitation) but, at the same time, that the interactive effects tended to be smaller than the additive single-factor effects. Climate change can have a major impact on peatsoil degradation and increase of CO_2 emissions, due to the increase of decomposition rate by the temperature rise, and by the more frequent occurrence of long periods with extreme drought.

Soil compaction
For compaction the main driver discussed in the chapter is related to the 'disturbing agent' / the machinery exerting mechanical stresses to the soil, with no focus on the soil / the 'system' threathened (OECD, 2003; Schjønning *et al.*, 2015). However, climate changes may also be regarded as a driver of soil compaction because the soil's ability to withstand mechanical stresses decreases with increases in soil water content (e.g. Arvidsson *et al.*, 2003). Scenarios indicate significant changes in the amount and pattern of precipitation for a range of regions in Europe (Olesen *et al.*, 2011). The mean annual precipitation increases in northern Europe and decreases in the South. But the change in precipitation varies substantially from season to season and across regions. There is a projected increase in winter precipitation in northern and central Europe, whereas there is a substantial decrease in summer precipitation in southern and central Europe, and to a lesser extent in northern Europe (Olesen *et al.*, 2011). These changes will affect the number of trafficable days (Gut *et al.*, 2015), which may become critically low for some cropping systems, for example, sugar beet harvesting in Northern Europe (Arvidsson *et al.*, 2003). This illustrates that a combination of climate and human factors can play an important role for the risk of compaction.

Soil sealing and contamination
For the soil threats, soil sealing and contamination human activities and policies are considered more important than climate as drivers.

Floods and landslides
Climate and climate change control precipitation and snowmelt (frequency, intensity and magnitude, seasonality, cyclonality) and their impacts locally and regionally, and are the most important direct/external drivers for landslides and flooding (e.g., Iverson, 2000; Crosta & Frattini, 2003).

The spatio-temporal variability of rainfall, can significantly affect flooding and trigger landslides, and lead to great variability in responses and uncertainty in their prediction (Paschalis *et al.*, 2014). Van den Besselaar *et al.* (2013) showed that the frequency of extreme events is increasing in all regions for all the seasons for both 1 and 5 days events. The Northern part of Europe is generally more affected than Southern Europe as the winter months show the highest rate of change in the frequency of rainfall events, indicating an increase of flood and landslide risks. Pre-conditions of hydrological patterns, such as the snow water equivalent, need to be better investigated in order to improve the understanding of the effect of a catchment's hydrological conditions for flood formation. Snow melt can also have a primary role in the triggering of landslides, especially when coupled with rainfall events. In the case of thick snow cover and unfavourable weather conditions (sudden rise of temperature) the melted snow water equivalent can considerably increase the amount of water that can infiltrate, increasing the pore pressure in the soil leading to landslide activity (Bíl and Müller, 2008). Changes in winter precipitation can change flooding risk and seasonal flood patterns. In regions where the snowmelt driven floods are the largest flooding risk - then reduced snowmelt can reduce the chances of spring flooding. Instead, the risk of rainfall driven flooding can increase and seasonal patterns will change.

Desertification
For desertification there is a strong link between climate (high temperature, low precipitation) and the loss of soil quality. Climate change, with warmer weather, has the potential to drive soils towards desertification. Climate change is likely to drive the boundaries of the arid, semi-arid and sub-humid areas in the Euro-Mediterranean region northwards (Gao and Giorgi, 2008), thereby expanding the area that is potentially susceptible to desertification. The impact of climate, however, is seen to interact strongly with movements of population and human activities. Wild fire provides another form of direct influence on the soil and vegetation system. Fire occurs naturally, and the risk of fire increases strongly with temperature. Fire ignition occurs naturally through lightning strikes, but its frequency is much greater where people have access to a fire-prone area, and accidentally or deliberately start fires. This illustrates that drivers can be a combination of climate and human factors.

Salinisation
Saline soils have developed in most arid regions, where climate is the determining driver as evapotranspiration contributes steadily to the formation of saline soils and lack of rainfall impedes consistent flushing. As a result, the surface layers continuously accumulate water soluble salts found in both the upper and underlying layers, and the circulating solution present in the latter rises by capillarity as a consequence of t to the evaporation. This fact is very important in Mediterranean regions in which evaporation reaches 8-10 mm day-l. In the rainy season, precipitation may flush and refresh soil bodies to some degree. Finally, wind in coastal areas can blow moderate amounts of salts inland (Geeson *et al.*, 2003; Jones *et al.*, 2012; Salama *et al.*, 1999).

Soil Biodiversity
Climate change is considered a potentially important factor in driving future soil biodiversity decline (Suárez *et al.*, 2002). Generally, soil organisms have a relatively wide tolerance tof temperatures variations, and the warming (or cooling) of soils which are buffered diurnally and seasonally (Tibbett & Cairney 2007) means that the direct effect of temperature changes are unlikely to be a key factor in itself. It is the global ecosystem-scale effects ton other abiotic aspects of soil ecosystems that are likely to cause the greatest pressure on soil biodiversity. Climate change leading to flooding and subsequent anoxia and compaction, loss of organic matter through enhanced oxidation, and prolonged periods of drought (in typically un-droughted landscapes) are the drivers of biodiversity loss in soil. Many of these factors link with, and may be compounded by, local and regional land management practices.

For Europe, the main pressures have been recognised for the three levels of biodiversity: ecosystem, species and gene (Jeffery, 2010). At the level of ecosystems, the main pressures were thought to derive from land use change, overuse and exploitation, a change of climatic and hydrological regime and change of geochemical properties.

Table 14.3: Importance of climate as a driver for each soil threat as identified in the different chapters. High importance, low importance or a combination of climate and human drivers.

Threat	Climate high importance	Combination of climate and human activity	Climate low importance
Water erosion	●		
Wind erosion	●		
Decline in SOM -peatsoil	●		
Decline in SOM mineral soils	● (long term)	● (short term) Management most important	
Compaction		●	
Sealing			●
Contamination		● Main driver human activity	●
Salinization	●		
Desertification	●		
Flood and landslides	●		
Biodiversity	● when climate affect ecosystem	● At ecosystem level-combination of climate and human activities	

14.2.2 Policy drivers

Policy drivers directly or indirectly affect different soil threats by making a particular human activity possible, or by prohibiting it, or by making it more or less attractive to the landowners and land users, as well as more broadly by driving changes in land use, incentivising overexploitation of resources. The mechanism by which a driver affects a soil threat through land use and management can vary, and a detailed overview of these mechanisms is beyond the scope of this report. Some of the individual chapters outlined examples of these mechanisms. The integrated impact assesment to be conducted in WPs 8 and 9 of RECARE will examine the causal links in detail in order to evaluate the impact of policies and to assess where there are opportunities for improved policy intervention, while also considering how the policy drivers interact with socio-economic and climate drivers.

While inadequate policies can put significant pressure on land resources, policies can also provide incentives and opportunities for resource protection. In Table 14.4, the key policy areas are listed, and the type of impact (positive or negative) in relation to the soil threats is indicated. For detail, please refer to individual chapters.

Although the various policies and their instruments can have very different impacts on soil threats, there are some general conclusions that can be drawn from assessing their direct and indirect assessments (EEA, 2015):

The *Common Agricultural Policy (CAP)* has historically been and continues to remain the key funding source for rural land management in the EU. Historically, the CAP was a driver behind specialisation and intensification of agricultural production by providing payments to farmers which were coupled to the production levels (i.e. payments per tonne of commodities) and which directly incentivized farmers to increase production levels through specialization and increased application of inputs, as well as by reclaiming productive or potentially productive areas (such as through drainage of peatlands). Recently, on the other hand, CAP has also seen the integration of various mechanisms which aim to safeguard or protect soil resources, such as the Good Agricultural and Environmental Conditions (GAECs), which has a positive impact on maintaining SOM as well as soil structure, and which helps to reduce soil erosion. The current CAP includes a range of instruments impacting on the land use and management of agricultural areas that either positively or negatively and directly or indirectly impact on soil. A more detailed analysis to assess the impacts of the various instruments on the range of soil threats would be beneficial and will be carried out during the RECARE project.

Energy and Climate policies impact on land and soil in two main ways: through investment in energy infrastructure, impacting on land take and hence soil sealing, and through increased use of renewable energies and biofuels, which are likely to increase agricultural intensification leading to loss of soil organic matter and a reduction of soil water retention. On the other hand, bio-energy production might also posively impact the soil by mitigating soil erosion.

Environmental policies are likely to mitigate soil threats e.g. through improved soil management, land rehabilitation, green infrastructure development or limitation of urban sprawl, thus impacting on various threats amongst which loss of organic matter, loss of soil biodiversity and soil sealing. Water management policies generally also have a positive impact on soil, by reducing fertilizer use and improving manure management (Nitrates Directive), and through reduction of pressure from agriculture, restoration of rivers and ecosystems and stimulation of sustainable land use, including flood plain restoration (e.g. Water Framework Directive and Floods Directive). These directives are likely to impact on a range of threats, including flooding and land slides and soil organic matter.

Transport policies, on the other hand, are likely to have negative impacts on soil, although this is most prominent for soil sealing, as urban sprawl and land fragmentation are commonly indirect effects of infrastructure development. However, instruments stimulating sustainable urban transport can have a very positive impact on soil. *Cohesion Policy* can lead to similar issues as transport policies when funds are used for infrastructure investment. Alternatively, when funds are used for investment in biodiversity, nature protection, green infrastructure, or regeneration of brownfields, they can have a very positive impact on mitigating a range of threats, amongst which are soil erosion, soil organic matter or contamination.

In addition to the above-mentioned sectoral policies, Strategic Environmental Assessments and Environmental Impact Assessments also impact on soil threats. Generally these will be favourable to mitigating soil threats as they will bring negative impacts on soil into the decision space, whenever included in the assessments.

Besides the directives, Table 14.4 also shows some examples of strategies, guidelines and roadmaps, which are not legally binding to the same extent as the directives, but are rather aimed at providing guidance in developing strategic directions as well as providing examples of good practice.

Table 14.4: Summary Overview of links between policy areas and soil threats, + (green) indicates the policy is likely to mitigate the threat, - (red) that is is likely to worsen it, +/- (orange) that the impact can be positive or negative depending on the instrument used within the policy and its implementation.

Policy	Abbreviation	Soil erosion by water	Soil erosion by wind	Decline of OM in peatlands	Decline of OM in mineral soils	Soil compaction	Soil sealing	Soil contamination	Soil salinization	Desertification	Flooding and landslides	Decline in soil biodiversity
Common Agricultural Policy	CAP	+/-	+/-	+/-	+/-	+/-		+/-	+/-	+/-	+/-	+/-
Nitrates directive	ND	+	+	+	+			+/-				+
Water Framework Directive	WFD	+/-		+/-	+/-			+	+		+	+
Floods Directive	FD						+/-				+	
Habitats / Birds Directives	HD, BD	+	+	+	+	+	+	+		+	+	+
Renewable Energy Directive	RED	+	+	-	-	-						+/-
Industrial Emissions Directive, Sewage Sludge Directive, Environmental Liability Directive, Landfill Directive, Waste Incineration Directive	IED, SSD, ELD, WID, Landfill Dir.							+				+
Directives on Environmental Impact Assessment and Strategic Environmental Assessment	EIA, SEA	+	+	+	+	+	+	+	+	+	+	+
Kyoto Protocol, Emissions Trading Scheme Directive and Effort Sharing Decision	ETS, ESD			+				+/-	+/-			
Biocidal Products Regulation	BPR							+				+
Structural Policy and Cohesion Policy	CP	+/-	+/-	+/-	+/-		+/-	+			+/-	+/-
7th Environment Action Programme	7EAP	+	+	+	+				+	+		
Soil Thematic Strategy	STS	+	+	+	+	+	+	+	+	+	+	+
Forest Strategy		+	+								+	
Roadmap to a Resource Efficient Europe		+	+	+	+		+					
A Blueprint to Safeguard Europe's Water Resources		+		+							+	
Guidelines on best practice to limit, mitigate or compensate soil sealing							+					
Roadmap to a Single European Transport Area, Adaptation Strategy, Europe 2020		+	+				+/-				+	

14.2.3 Socio-economic drivers

Socio-economic drivers directly or indirectly affect different soil threats and there is a strong link with the policy-drivers (14.2.2). As for the political drivers, the mechanism by which a driver affects a soil threat through land use and management can vary, and a detailed overview of these mechanisms is beyond the scope of this report. Some of the individual chapters outlined examples of these mechanisms. The integrated impact assesment to be conducted in WPs 8 and 9 of RECARE will examine the causal links in detail in order to evaluate the impact of socio-economics, while also considering how the policy drivers interact with socio-economic and climate drivers.

Based on the different chapters, the following socio-economic drivers are identified having a direct or indirect effect on soil threats by initiating human activities and responses on the driver. This list is based on the preliminairy inventory presented in the chpaters, and will be extended and examined in more detail in WP's 7, 8 and 9 of the RECARE project.

Population growth leading to pressures to produce more food resulting in agricultural intensification. In addition, population growth leading to pressures on land use e.g. urban growth, mining, and tourism growth with impacts on soil (e.g. soil sealing, contamination, salinization). Some areas, particularly southern Mediterranean are experiencing rural depopulation (due to poverty, lifestyle choices) resulting in land abandonment and soil degradation (e.g. collapse of terraces).

Consumer demands (food consumption patterns) resulting in retailer contract specifications leading to inappropriate management practices. For example, the harvesting of high value vegetable crops in inappropriate weather to meet supermarket contract demands resulting in soil compaction.

The driver for mechanization in agriculture (labour costs) is the need to replace expensive labour with efficient and hence cost-effective machinery. The pressure with respect to soil compaction is caused by (frequent) traffic with heavy machinery. Related to this are the technological developments. More powerful machinery means cultivation moving higher up the slope, leading to increased erosion. Heavier machinery is leading to compaction. For some Eastern European countries and for regions characterized by small farm units within Western Europe, the loads applied to the soil may be lower than estimated above. The rural development in these regions, including land purchase in Eastern Europe by farmers from other countries, implies that big machinery is also on its way into such areas.

Driving forces of soil sealing refer to the need for new housing, business locations and road infrastructure related to economic development of cities. Most social and economic activities depend on the construction, maintenance and existence of sealed areas and developed land. Soil consumption has considerable consequences for society and economy.

The cost/price squeeze (*macro-economic factors*) resulting in pressures for economies of scale. This has resulted in increased specialization, decline in mixed farms, farming in larger blocks, all of which has a detrimental impact on soil.

Where land is farmed on short-medium term contracts there is a lack of incentive for the long term planning that is required to prevent soil degradation (land tenure).

The socio-cultural drivers that influence behaviour are important drivers but can be very context-specific and therefore difficult to measure at an EU level. Influence occurs at different levels:

1) Personal/family beliefs and values as to how soil should be managed “this is how we/ the family have always done it”,
2) Behaviour (social norms) influenced by a particular reference group e.g. farming peers, co-operative group, community,
3) Societal influence - meeting expectations of society – how soil is valued by society.

Advisory services (knowledge and information exchange) can directly influence soil management practices. Quality of soil advice is very variable across Europe. There can be a lack of soil management advice from free state-advisory services and some commercial agricultural advice can conflict with advice on soil management

Some other specific, human drivers are: deliberate setting of wildfires, industrial activities, manufacturing processes and tourism and increasing demands for water resources (salinization).

14.3 Interaction between the soil threats

Based on the information from each chapter on the impact of the individual soil threat on other soil threats, we have derived an interaction table that shows the effects among soil threats. We asked the authors of each chapter to determine the effects of the specific soil threat on other soil threats, so a one-way analysis. The results of these are reflected in the pie charts given in each chapter. These relationships are partly based on expert knowledge and partly on the literature review. The impact is expressed in qualitative terms in four categories: no, low, moderate and large effects. The interactions presented in Table 14.5 show the result of merging the given impacts into one matrix table. The impact can both be negative (i.e. worsen the state of the soil threat based on the other soil threat) and positive (the state of the soil threat increases based on the impact of another soil threat). The latter is only recognized for the soil sealing effect on contamination. As is stated in chapter 7.5, soil sealing itself prevents dispersion of the contaminants and is one of the technical methods for inactivation of contaminants inland. One can argue that several soil threats have a positive effect on other threats, but this is not recognized in the chapters.

Some chapters indicate that other soil threats influence the specific soil threat. In the desertification chapter (10), it is stated that soil erosion by water and wind, and salinization have been recognized as key threats for desertification. However, in the chapter on water erosion this interaction is not mentioned, reflecting 'no influence' in Table 14.5. Since soil erosion by water leads to loss of organic material, it also (indirectly) has a strongly negative influence on desertification. In the contamination chapter (8), it is stated that contaminants can indirectly affect the quality of organic matter in soils as they influence the biological activity and therefore indirectly decomposition, mineralization and humification (Baath 1989). Similar to this, salinity (section 9.7) affects various mechanisms of vegetation growth and reproduction, causing symptoms similar to those of water deficiency regardless of nutrient availability (Hu and Schmidhalter, 2005). The subsequent loss of vegetation cover enhances the loss of organic matter, erosion, and desertification. These indirect effects are not identified in the table, but have been recognized in the text of the different chapters.

As is stated in chapter 3.6, wind erosion is linked with contamination. This is explained by wind erosion being able to transport fertilisers, herbicides, and pesticides, as well as pathogens, such as for example those causing Q-fever. It is also responsible for part of the fine dust that is in the atmosphere. According to Kuhlman *et al.* (2010), the fine dust that is created by wind erosion can have a major impact on human health. This shows that the contamination effect of wind erosion has a direct consequence for human health, and less direct effect for soil contamination.

In chapter 4.6, a specific process of peat soils in arable agriculture is described. Peat soils are vulnerable to severe drying of the topsoil and result in severe hydrophobia making the soil less suitable for agriculture and very prone to water erosion and especially wind erosion. This is an important property, and influences the severity of water and wind erosion, though the decline of SOM in peat soils itself has less effect on this phenomenon. Chapter 4.6 concludes that degraded peat soils in arable agriculture or in overgrazed grasslands are vulnerable to water and wind erosion. Water erosion is especially a problem in overgrazed blanket peats. Wind erosion is a serious problem on peat soils in arable agriculture. This interaction is also recognized in Table 14.4, where water and wind erosion have an effect on SOM decline in peat soils.

Urbanization (par. 7.6) usually increases the background contents of contaminants in the soil (e.g. trace elements or polycyclic aromatic hydrocarbons) which are not necessarily exceeding risk levels in soil. Soil contamination might appear locally as a direct result of urbanization: construction work, landfills, waste management or industrial activities. Soil sealing itself prevents dispersion of the contaminants and is one of the technical methods for inactivation of contaminants in land. This positive effect of contamination is not included in the pie chart (Fig 7.4), but included in the interaction Table 14.5).

As is described in the flooding and landslides chapter (par. 11.5), in order to understand the interactions of flooding and landslides and other soil threats on different spatial and temporal scales, more detailed knowledge of the risks of the contradictory impacts for mitigating measures is needed. Actions to prevent erosion on slopes may reduce flood risks but, in turn, they may counteract threats to downstream, where channel erosion may be amplified. In the flooding and landslides chapter (11), the effect of landslides and flooding on other soil threats are separated. We have combined these two again in table 14.5, in order to synchronize this with the definitions of the RECARE project on soil threats.

Table 14.5 shows clearly that decline in soil biodiversity is affected by most soil threats. As is stated in 12.6, it can, in turn, have effects on other soil threats. Most of these effects are, however, very poorly understood. To a lesser extent, also soil erosion by water is affected by several other soil threats. Remarkably, none of the soil threats has an effect on soil sealing, according to the different chapters. On the other hand, soil sealing does affect a number of soil threats (water erosion, compaction, contamination, flooding and landslides and soil biodiversity). Declining SOM in peat soils has a minor effect on other soil threats, but it does have a large effect on loss of soil biodiversity.

We have to bear in mind that these interactions reflect the perception of the authors of the individual chapters, (as already mentioned), based on the literature research and their own assessment. There is a need for further research to quantify the interactions between the soil threats. A more comprehensive approach is needed to understand all links and interactions of soil threats over space and time. To sum up, the information presented in this report on the interactions between the soil threats is important for the RECARE project in helping to look for suitable measures for preventing, and remediating the degradation of soils in Europe. The large knowledge gap is evidently the lack of understanding on the interactions between the soil threats. We have tried to present this issue for discussion by synthesizing all information given in the different chapters on the effect of one soil threat on all others. By constructing a matrix table from this information, a first approach is made to understand and describe interrelations between the soil threats. During the course of the RECARE project we will, together with the project partners, update the information on interaction between the soil threats.

14.4 Methods/procedures to assess soil degradation using key soil properties

One of the main objectives of WP2 is to provide a base for RECARE's data collection and methods that can be used to assess the soil degradation/threats prevailing in the case study sites. To achieve this objective, an extensive literature review was carried out regarding indicators and methods used to monitor soil degradation trends across Europe. There is available information on indicators and methods for soil degradation assessment in Europe for some soil threats, as reported by Huber *et al.*, (2008); van Beek *et al.*, (2010) and OECD (2013). However, a standard and harmonized methodology to monitor a set of indicators for a given soil property that represents the soil threat is lacking at the European scale (www.recare-project.eu). This section of the report provides a synthesis of information on key indicators, methods/models/procedures applied to monitor the indicators along with a list of references.

14.4.1 List of key indicators

The EU-funded ENVASSO project has identified a number of indicators for most of the soil threats identified in this report (Huber *et al.*, 2008). Out of this, some of the top three (TOP3) indicators of soil threats identified by ENVASSO project are adopted in this report. In addition, new sets of indicators are proposed for those soil threats that were not addressed before and those that were merged together. For instance, a list of indicators and/or proxy indicators are suggested for soil erosion by wind, decline of OM in peat soils, decline of OM in mineral soils and a separate set of indicators for flooding. These indicators have been developed by taking into account the following key issues:

- methodological soundness and data availability,
- measurable and sensitivity to changes,
- policy-relevance and utility for users, and
- geographical coverage of the indicators.

Table 14.5: Interactions between soil threats. Size of the dots indicates the impact: low, moderate and large for small medium and large dots respectively.

Soil threat	Water erosion	Wind erosion	SOM decline peat soils	SOM decline mineral soils	Compac-tion	Sealing	Conta-mination	Saliniza-tion	Desertifi-cation	Flooding and landslides	Bio-diversity decline
Water erosion			⬤	⬤			•		⬤	●	●
Wind erosion			●	⬤			•	•	⬤		●
SOM decline peat soils	•	•								●	⬤
SOM decline mineral soils	⬤	⬤			●				⬤		⬤
Compaction	⬤									●	⬤
Sealing	•				•		• / •			⬤	⬤
Contamination	●	●									⬤
Salinization	●	●	●	●			⬤		⬤		⬤
Desertification		⬤		⬤				⬤			⬤
Flooding and landslides	⬤			⬤	⬤		⬤	●			⬤
Biodiversity decline											

During the selection of indicators, time-variant soil properties have been given particular attention. Time-variant soil properties, such as organic carbon, soil depth, pH and salt contents are common parameters required to assess soil degradation across many of the soil threats for e.g. soil erosion by water and/or wind, decline in OM, salinization and desertification.

Table 14.6 presents a list of key and/or proxy indicators for the soil threats identified by this RECARE report along with the ENVASSO project. It is noteworthy to mention here that the identification and development of relevant indicators for each soil threat is an ongoing process. At a later stage of the RECARE project, the list of indicators will be refined and updated, if deemed necessary.

In the process of selection of indicators, the ENVASSO working group did not include a number of indicators, such as total carbon stocks up to 1 m depth, SOM content up to 1 m depth, SOM molecules size/weight, SOM stratification ratio, dissolved organic carbon (DOC) to total SOC ratio, soil respiration rate and chemical composition of organic matter. According to the report by Huber *et al.*, (2008), the indicators were not selected because of their poor geographical coverage, a lack of existing data, a lack of scientific consensus on methodological issues and/or lack of sufficiently robust methods. Nonetheless, it is increasingly accepted that carbon at greater soil depths should be accounted for in future assessments, since it contributes to more than half of the global soil carbon stock and its response to land use change can be equated to that of the top layer 30 cm (Schmidt *et al.*, 2011).

The chapter on soil erosion by water (in this report) has suggested making use of the European Environmental Agency (EEA, 2000) report which has identified a list of agri-environmental indicators of soil erosion by water that were considered relevant to pan-European policy making. Based on a critical review by Gobin *et al.* (2004), the authors suggested focusing on two indicators of the state of soil erosion i.e. area affected by soil erosion (in km^2) and extent of area affected by soil erosion (in %). But, the magnitude of soil erosion or sediment delivery (in tons) was considered as the combined indicator of state and impact. For desertification (chapter 10), the most complete lists of indicators available is that of the one developed by EU FP5 DesertLinks project (DIS4ME, 2004).

Table 14.7 presents a classification of indicators of the soil threats in terms of driver, pressure, state, impact, response (DPSIR) and the effectiveness of each indicator in terms of time and spatial scale. Many of the indicators listed in the table are state indicators and a few are either driver, pressure, impact and/or response indicators. The state indicators are able to show the state of soil degradation in the short term or long term. Indicators like soil loss by water or wind can be measured and evaluated in at least two growing seasons or rainfall years. Indicators such as peat stocks in large area/volume, can only be evaluated in the long term. Impact and response indicators (for e.g. different mitigation measures against soil degradation) are only powerful enough to detect soil degradation/conservation trends after several years of implementation since soil formation and development takes a considerably long time.

Table 14.6: List of key and/or proxy indicators for soil threats identified by RECARE and ENVASSO

Soil threat	RECARE (This study, 2015)	ENVASSO (Huber *et al.*, 2008)
Soil erosion by water	• area affected by soil erosion (km^2) and/or extent of area affected by soil erosion (%) • magnitude of soil erosion/deposition or sediment delivery (tons)	• estimated soil loss by rill, inter-rill and sheet erosion ($t\ ha^{-1}\ yr^{-1}$)
Soil erosion by wind	• measured soil loss by wind ($t\ ha^{-1}\ yr^{-1}$) • annual/periodic estimates of wind erosion • soils' susceptibility to wind erosion *Proxy indicators* • soil resistance (Ohms) • surface roughness (%) • wind velocity ($km\ hr^{-1}$) • soil moisture content (%) • soil cover (%, ha)	• estimated soil loss by wind ($t\ ha^{-1}\ yr^{-1}$)
Decline in OM in peat soils	• peat stocks (Mt) *Proxy indicators* • water table (m) • soil moisture content (%) • (soil) temperature (°C) • vegetation type (species)	• peat stocks (Mt)
Decline in OM in mineral soils	• total carbon stocks to 1 m depth (($t\ ha^{-1}$) • clay/SOC • TOP2 indicators by ENVASSO	• topsoil organic carbon content (%, $g\ kg^{-1}$) • topsoil organic carbon stocks ($t\ ha^{-1}$)
Soil compaction	• relative Normalized Density, • air-filled pore volume (%) • penetration resistance (Mpa)	• soil density ($g\ cm^{-3}$) • air-filled pore volume (%) • vulnerability to compaction (classes)
Soil sealing	• sealed area (ha, %) • transition index (TI) • sealed to green areas ratio	• sealed area (ha, %) • land take (Corine Land Cover, CLC) • new settlement area established on previously developed land (%)
Soil contamination	TOP3 indicators by ENVASSO	• heavy metal contents in soils (%) • critical load exceedance by sulphur and nitrogen (%) • progress in management of contaminated sites (%)
Soil salinization	TOP3 indicators by ENVASSO	• the salt profile • Exchangeable Sodium Percentage (ESP) • potential salt sources
Desertification	TOP3 indicators by ENVASSO	• land area at risk of desertification (ha) • land area burnt by forest fires (ha) • soil organic carbon content in desertified areas (%, $g\ kg^{-1}$)
Flooding	• seasonality, magnitude and frequency of precipitation/rainfall intensity • extent of inundated area (ha) • flood frequency (number per year) • loss of crops due to inundation of fields (ha, Euro)	• The threat has not been addressed
Landslides	TOP3 indicators by ENVASSO	• occurrence of landslide activity (ha, km^2 affected per ha or km^2); • volume/weight of displaced material (m^3, km^3, ton of displaced material); • landslide hazard assessment (variable)
Decline in soil biodiversity	TOP3 indicators by ENVASSO	• earthworms diversity & fresh biomass (number m^{-2}, g fresh weight m^{-2}) • Collembola diversity (number m^{-2}, g fresh weight m^{-2}) • microbial respiration ($g\ CO_2\ kg^{-1}$ soil)

Table 14.7: Summary of indicator descriptions: DPSIR type and effectiveness in time and spatial scale.

Indicators	unit	DPSIR type	Time scale *	Spatial scale
soil loss	t ha^{-1} yr^{-1}	state	short term	plot
peat stocks	Mt	state	long term	point, plot, national
topsoil organic carbon content	%, g kg^{-1}	state	long term	point
topsoil organic carbon stocks	t ha^{-1}	state	long term	point
clay/SOC				
soil density	g cm^{-3}	pressure	short term	point
air-filled pore volume	%	state	short term	point
sealed area	ha, %	impact	long term	national/continental
land take	CLC	impact	long term	national
new settlement area established on previously developed land	%	impact/response	long term	national
heavy metal contents in soils	%	state	long term	plot/catchment
critical load exceedance by S & N	%	state	long term	plot/catchment
progress in management of contaminated sites	%	impact/response	long term	national
the salt profile	-	state	long term	point
Exchangeable Sodium Percentage (ESP)	%	state	long term	point/plot
potential salt sources	-	drivers	long term	catchment
land area at risk of desertification	ha	impact	long term	national/continental
land area burnt by forest fires	ha	impact	short term	national/continental
SOC content in desertified areas	%, g kg^{-1}	impact	short term	national/continental
rainfall intensity	mm yr-1	drivers	short term	catchment
extent of inundated area	ha	impact	short term	catchment
flood frequency	number per year	drivers	medium term	catchment
loss of crops due to inundation of fields	ha, Euro	impact	short term	plot
occurrence of landslide activity	ha affected per ha	impact	long term	catchment
volume/weight of displaced material	m^3 of displaced material;	impact	long term	catchment
landslide hazard assessment	variable	impact	long term	national/continental
earthworms diversity & fresh biomass	no. m^{-2}, g fresh weight m^{-2}	state	long term	Point/plot
Collembola diversity	no. m^{-2}, g fresh weight m^{-2}	state	long term	point/plot
microbial respiration	g CO_2 kg^{-1} soil	drivers	long term	point/plot

* Short term refers to less than 2 years, medium term: 2-5 years and long term more than 5 years.

14.4.2 Methods/procedures

The methods/models for each TOP3 indicators that could be used to assess the different threats to soils are presented in Tables 14.8 to 14.13. The purpose of each indicator and the corresponding methods are described briefly. A list of references is also given in the last column to provide more information on the applications of the methods and/or models in the field or under laboratory conditions and materials required to apply the methodologies, including sampling procedures, data collection and analysis. The choice of the methods/models depends on several factors, among others, the type of indicator, cost, data quality and resources available. However, RECARE aims to develop a standardized and harmonized methodology/procedure that can monitor and/or assess the soil degradation trends across Europe regardless of spatial differences. In fact, some of the methods/models have been verified and validated in different climatic zones and are universally applicable, such as the erosion micro-plots/pins, rainfall simulators, standard laboratory analysis and field sampling procedures and measurements. However, a few methods may be tested in the case study sites of the RECARE project for further validation purposes.

Table 14.8: Soil erosion by water: key indicators, purpose of the indicator, methods and corresponding references.

Indicators	Purpose	Methods	References
soil loss by water erosion	measure/estimate transport of soil particles by rainsplash/splash erosion	– splash boards as well as funnels and cups of various designs <15-20 cm Ø	Morgan (2005); Jones *et al.* (2008)
		– portable rainfall simulators	Iserloh *et al.*, (2013); Jones *et al.* (2008)
	measure/estimate transport of soil particles by sheet flow/inter-rill erosion	– micro-plots, field rainfall simulators	Morgan (2005); Jones *et al.* (2008)
	measure transport of soil particles by sheet and concentrated overland flow	– large-enough plots ("Wischmeier" plot) typically >10 m long	Morgan (2005); Jones *et al.* (2008)
	produce erosion risk map	– eRUSLE model	Bosco *et al.* (2014)
	determine soil erosion risk	– OECD assessment	OECD (2013)
	measure sediment yield	– sediment yield data	Vanmaercke *et al.* (2012)
	produce erosion risk map	– erosion plot data	Cerdan *et al.* (2010)
	predict soil erosion risk	– PESERA model predictions	Kirkby *et al.* (2004)
magnitude of sediment delivery	measure transport of soil particles beyond the hillslope	sediment yield = streamflow's suspended sediment concentration × discharge	Vanmaercke *et al.* (2012)
area affected by soil erosion and/or deposition	determine status of cumulative soil erosion	– cross-sectional area of the rills/gullies across a slope – mapping erosion features using aerial photogrammetry, 3-D laser scanning & satellite imagery	Morgan (2005)
	measure changes in ground level	– sequential DTMs, erosion pins, erosion bridge	Morgan (2005)
		– sediment pins	Tesfai and Sterk (2002)
	determine patterns of soil erosion/deposition in a landscape over periods	– concentrations of radioactive isotope tracers in soil profiles (e.g. Cs-137, magnetic iron oxides)	Guzman *et al.* (2013); Morgan (2005)

Table 14.9: Soil erosion by wind: key indicators, purpose of the indicator, methods and corresponding references.

Indicators	Purpose	Methods/models	References
soil loss by wind	– measure depth of soil removed, – measure transport rates on erosion plots	– Erosion pins (with 50 cm long and 5 mm Ø)	Riksen en Goossens (2007)
	– quantify sediment load – measure sediment concentrations	– wind erosion plots: circular shape – sediment traps	Hessel et al, (2011); Toy *et al.*, (2002)
	– determine soil erodibility, soil roughness, climate, field length and vegetation cover	– WEQ (Wind Erosion Equation)	Woodruff & Siddoway (1965)
	– measure soil loss/deposition	– WEPS (Wind Erosion Prediction System)	Tatarko & Wagner (2002); Hagen (2001)
	– create wind erosion model	–TEAM (Texas Tech Erosion Analysis Model)	Gregory & Darwish (2001, 2002)

	– simulate modules for wind, wind erosivity, soil moisture, soil erodibility, soil roughness and land use.	– WEELS (Wind Erosion on European Light Soils)	Warren (2002)
annual/periodic estimates of wind erosion	– estimate wind, erodibility, surface crust, roughness, ground cover	–RWEQ (Revised Wind Erosion Equation)	Zobeck *et al.* (2001)
soils' susceptibility to wind erosion	– determine wind-erodible fraction of soil	– Index of Land Susceptibility to Wind Erosion (ILSWE)	Borrelli *et al.* (2014a, b); Fryrear *et al.* (2000); Fryrear *et al.* (1994)

Table 14.10: Decline of OM in peat and mineral soils: key indicators, purpose of the indicator, methods and corresponding references.

Indicators	Purpose	Methods/procedures/equations	References
decline of OM in peat soils			
peat stocks	– measure amount of C in peat soils	$P_S = P_A \times P_D \times 10^{-4} \times D_b$ where P_{Stock} is peat stock in Mt; P_{Area} is peat area in km^2; P_{Depth} is peat depth in m; D_b is bulk density in t m^{-3} (t m^{-3})	Jones *et al.* (2008)
	– measure/estimate direct CO_2 emissions	closed gas chamber	Koskinen *et al.* (2014); Duran and Kucharik, (2013); Venterea and Parkin, (2012); Pedersen *et al.* (2010)
		micro-meteorological measurements using eddy-covariance techniques	Jacobs *et al.* (2007); Aubinet *et al.* (2000, 2003)
	– identify vegetation type	mapping of vegetation types characterized by the presence and absence of species groups indicative for specific water level classes.	Couwenberg *et al.* (2011)
	– estimate loss of OM and GHG emissions	SWAP-ANIMO to simulate peat land of CO_2, CH_4 and N_2O, soil subsidence and nutrient loading of surface waters	Hendriks *et al.* (2008)
decline of OM in mineral soils			
clay/SOC	– describe interaction b/n SOM & mineral particles	clay/SOC	Dexter *et al.* (2008)
topsoil organic carbon content	– measure SOC content	Dry or wet combustions	Jones *et al.* (2008) Islam (2006)
topsoil organic carbon stocks	– measure bulk density	BD = oven-dried weight of soil/ volume of soil	Schrumpf *et al.* (2011)
	– estimate organic carbon stocks	– SOC models such as CENTURY – Roth-C – Tier 3 approach	Stockmann *et al.* (2013) Farina *et al.* (2013) IPCC (2006)

Table 14.11: Soil compaction and soil sealing: key indicators, purpose of the indicator, methods and corresponding references.

Indicators	Purpose	Methods/procedures/equations	References

Soil compaction			
soil density	– measure soil mechanical strength	cone penetrometer	Herrick & Jones (2002)
	– determine Relative Normalized Density (RND)	RND = $\sigma/\sigma_{critical}$ = σ/1.6 (if clay contents <16.7 %w/w) *σ is bulk density* RND = $\sigma/\sigma_{critical}$ = σ/(1.75 – 0.009×*Clay*) (if clay contents ≥16.7 %w/w), where σ is actual bulk density (g cm^{-3}).	van den Akker and Hoogland (2011)
air-filled pore volume	– calculate air capacity	AFPV = TPS – θ_v where θ_v is volumetric soil water content at 5kPa and TPS is total pore space TPS = 1 – (Db/Dp) × 100	van den Akker (2008); Smith and Thomasson (1982); Hall *et al.* (1977, p.6-18)
Soil sealing			
sealed area	– determine permeability to water, gases and substances	– remote sensing imagery including aerial photographs, – cadastral method	Tóth *et al.* (2013); Jones *et al.* (2008)
	– produce municipal (planning) maps, cadastre maps	Corine Land Cover (CLC) data	http://www.eea.europa.eu/publications/COR0-landcover
	– determine permeability of the soil sealed	LUCAS spatial point data sets	http://eusoils.jrc.ec.europa.eu/projects/Lucas/
	– locate urban settlement locations	GHSL layer	http://ghslsys.jrc.ec.europa.eu/
	– determine annual land take	Soil Sealing Layer of Europe	Prokop *et al.* (2011); http://www.eea.europa.eu/data-and-maps/data/eea-fast-track-service-precursor-on-land-monitoring-degree-of-soil-sealing; Verzandvoort *et al.* (2010)
	– predict future land take	forecasting soil sealing using – LUMP/CLUE – LUMOCAP/Metronamica modelling framework.	JRC (2014); van Delden *et al.* (2011)
transition index (TI)	– determine soil classes	$TI = \frac{\%\text{ of soil class 'n' in new buil}}{\%\text{ of soil class 'n' in new whole u}}$	Siebielec *et al.* (2010)
sealed to green areas ratio	n.n.	n.n.	n.n.

Table 14.12: Soil contamination and soil salinization: key indicators, purpose of the indicator, methods and corresponding references.

Indicators	Purpose	Methods/procedures/equations	References
Contamination			

Indicators	Purpose	Methods/procedures/equations	References
heavy metal contents in soils	measure arsenic, cadmium, chromium, copper, mercury, nickel, lead and zinc contents in topsoils	Flame and electrothermal atomic absorption spectrometric	Jones *et al.* (2008); Lado *et al.* (2008)
critical load exceedance by S & N load	determine Sulphur & Nitrogen loads	procedure of calculating critical loads and their exceedances is given by ICP M&M	ICP M&M; www.icpmapping.org; www.rivm.nl/cce
progress in management of contaminated sites	characterize and assess soil contaminated areas	The indicator corresponds to the EEA corset indicator CSI015, further information can be found on the EEA website	http://themes.eea.europa.eu/IMS/ISpecs/ISpec ification20041007131746/IAssessment11 52619898983/view_content
Salinization			
salt profile	– measure Total Dissolved Solids – measure Electrical Conductivity (EC)	salinity sensors and sampling electromagnetic induction, remote sensing and geographic information systems	Metternicht & Zinck (2003)
Exchangeable sodium percentage	– determine Exchangeable Na^+ – determine cation concentrations – measure pH	ISO protocol cation concentration analyses pH meter	Shahid *et al.* (2013); Jones et al (2008)
potential salt sources	– determine water and salinity stress in agricultural fields	water absorption bands in the SWIR (short-wave infrared wavelength bands) and NIR (near infrared wavelength bands)	Zhang *et al.* (2011); Leone *et al.*, (2007); Poss *et al.* (2006); Ceccato *et al.* (2001)
	– measure EC of irrigation water, groundwater and seepage water and calculate SAR (Sodium Adsorption Ratio)	EC meter $SAR = \frac{Na^+}{\sqrt{\frac{1}{2}(Ca^{2+}Mg^{2+})}}$	Shahid *et al.*, (2013); van Beek and Tóth, (2012); Jones et al (2008)
	calculate Leaching Requirement (LR)	$LR = \frac{D_{DW}}{D_{IW}} \approx \frac{W_{FC}}{W_{SP}} \cdot \frac{EC_{IW}}{EC_e}$ D: amount of water (mm year^{-1}), w: water content by weight, ECe : soil salinity Subscripts DW, IW, FC, and SP denote drainage water, irrigation water, field capacity of soil	van Beek & Tóth (2012); Corwin *et al.* (2007)

Table 14.13: Desertification, flooding, landslides & decline in soil biodiversity: key indicators, purpose of the indicator, methods and corresponding references.

Indicators	Purpose	Methods/procedures/equations	References
Desertification			
land area at risk of desertification	– determine indices of soil quality, climate quality, vegetation quality and management quality	High resolution field survey maps; ARC GIS; MEDALUS model	Jones et al (2008) Kosmas, *et al.* (1999) Farajzadeh & Egbal (2007)
land area burnt by forest fires	– assess land damage due to forest fire	European Forest Fire Information System (EFFIS) 12 'rapid damage assessment' tool for forest fires	http://effis.jrc.ec.europa.eu/wmi/viewer.html Jones et al (2008)
soil organic carbon content in desertified areas	– measure SOC content	Dry or wet combustions	Jones *et al.* (2008) Islam (2006)
Flooding			
precipitation/rain-fall intensity	– analyze flood generation potential of soils at hill slopes and catchment scales	statistical analysis of precipitation measurements	n.n.
extent of inundated area	– potential area of soil degradation due to floods	flood zone mapping	n.n.
flood frequency	– quantitative estimate of natural hazards	statistical analyses	n.n.
loss of crops due to inundation of fields	– estimate economic losses due to floods	questionnaires, surveys	n.n.
Landslides			
occurrence of landslide activity	– produce landslides distribution map	High-resolution field survey, ARC GIS, GPS device, remote sensing/aerial photographs	Fressard *et al.* (2014); Guzzetti *et al.* (2005); Pack *et al.* (1998)
volume or mass of displaced material			
landslide hazard assessment	– detect landslides at catchment or hillslope scale	various hydrologic models	
Decline in soil biodiversity			
earthworms diversity	– determine earthworms/ collemboia diversity based on soil descriptions (depth, pH,	– Soil type should follow WRB 2006 classification (ftp://ftp.fao.org/agl/agll/docs/wsrr103e.pdf), –) Land management, land use and vegetation type should follow FAO	Jeffery *et al.* (2010); Jones et al (2008); van Straalen (1998)
collemboia diversity			

	nutrient) and site descriptions (climate, land use, vegetation)	2006 classification (ftp://ftp.fao.org/agl/agll/docs/guidel_soil_descr.pdf)	
microbial respiration (substrate induced)	– measuring CO_2 respiration responses from soil	Multiple substrate induced respiration	Degens & Harris (1997) Campbell *et al.*, 2003

14.5 Impacts of soil threats on soil functions & ES

Based on the information given in the chapters about the soil threats, we constructed Table 14.14, where the effects of the soil threats on identified soil functions are presented. The effects are classified into three categories: low, medium and large. The classification is, as far as possible, taken from the chapters, but where these were not given, we tried to basis it on an interpretation of the chapter text and our own interpretation.

Most of the soil functions are affected by soil erosion by water, whereas only biomass production and filtering functions are identified as being affected by soil erosion by wind. For the latter, an indirect effect is identified through its negative effect on soil structure and texture. This in turn can have an effect on soil functions, mainly on production. It could also be argued that the offsite effects of wind erosion, as listed in the chapter, can have effects on some of the soil functions (like burial of archaeological artefacts), but this effect has not been identified and accounted for in table 14.14. Most of the effects of decline in SOM of peatsoils on soil functions are described for situations where the peat layer totally oxidates. By oxidation and mineralization of N, an important supply of nutrients becomes available for biomass production (a positive effect). On the other hand, when all peat is lost the underlying mineral soil is most frequently less fertile. The total disappearance of the peat layer will in general lead to a strong decrease in soil biodiversity. In contrast with the oxidation of peatsoils, the decline in SOM in mineral soils leads to a negative impact on biomass production, because of the loss of the available pool of nutrients in the soil. It is also recognized that, due to a decline in SOM in mineral soils, the risk of soil compaction increases, leading to an even further reduction in biomass production. Decline in SOM in mineral soils has negligible effects on the soil function's 'physical bases, 'raw materials' and 'cultural heritage'. For soil compaction, only the effect on biomass production and filtering function is described. This is justified by explaining the effect of compaction on the pore system, affecting the mentioned soil functions. Though not mentioned in the chapter on soil compaction, we also believe that biodiversity is negatively influenced by soil compaction. The negative impact of soil sealing on biomass production is among others caused by the fact that most of the productive soils are found in sub-urban areas at the borders of urban agglomerations, which are prevailingly used for agriculture. In general, cultural heritage is negatively influenced by soil sealing, but some construction work might help to discover buried records of natural or human history. As is stated in the chapter on soil sealing, the negative effects can be partly mitigated through the use of partially permeable layers and the presence of green or blue spaces in urban areas. Soil sealing is considered a driving force for the extraction of raw materials. Obviously, the diversity of the soil organisms at different scales is strongly affected by soil contamination. Indirectly, soil contamination affects the storage and filtering capacity of soils by its effect on limiting the biodegradation of the organic matter. Desertification affects all soil functions, with the strongest impact on biomass production, biodiversity, and storage and filtering functions. The most obvious recognised impact by flooding and landslides is on biomass production. In the short term, floods and landslides will affect food production negatively, whereas in the longer term (and especially for landslides), this can lead to a rejuvenation of soils. Floods and landslides can affect soil as a platform for physical basis indirectly, where infrastructure is damaged by floods or landslides. The statement in the chapter on soil biodiversity that '…the soil biota are essential to provide most of the ecosystems…' is clearly presented in Table 14.14 by classifying the impact of soil biodiversity on the the soil functions in the large category.

Table 14.14 shows that the soil functions 'biomass production', 'storage and filtering' and 'gene pool' are most affected by the different soil threats. This has an effect on the ecosystem services as is described in Chapter 13. In Figure 13.4, the soil-based ecosystem services are listed in the proposed framework for the

RECARE project. We present a first approach in classifying the effects of the soil threats on soil functions, but this is not the final output. During the course of the RECARE project, more insight into these interactions will become clear. As stated in Chapter 13 for the RECARE ES framework, the relationships between the soil threats and soil functions is also still a draft and will be further developed based on feedback from RECARE partners and other contributors. The major challenge of the work is on integrating Tables 14.5 and 14.14, to analyse the interactions between the soil threats and in what way they interact with soil functions.

Table 14.14: Soil threats impact on soil functions, categorized in classes low, medium and large reflected by the size of the dots. Red means negative effect, green positive.

	Biomass production	Storing/filtering/transforming	Gene pool (biodiversity)	Physical basis	Raw materials	Cultural heritage
Water erosion	●		⬤	⬤	●	•
Wind erosion	●	•				
SOM decline peat	• / ○	⬤	⬤		•	⬤
SOM decline mineral	⬤	⬤	●			
Compaction	⬤	⬤	•			
Sealing	⬤	⬤	⬤	◯	○	● / ◦
Contamination	⬤	●	⬤			
Salinization	⬤	•	⬤	•		•
Desertification	⬤	⬤	⬤	•	•	•
Landslides and flooding	● / ◦	•	•	●	•	•
Biodiversity decline	⬤	⬤	⬤	⬤	⬤	⬤

14.6 Results in RECARE perspective

This section assesses the implications of the results of this report by focusing on the three main objectives of WP2 of RECARE project.

Objective 1: To achieve an improved overview of existing information on soil degradation at the European scale.
After a rigorous review and analysis of the information available in the literature pertaining to soil degradation for each soil threat in Europe (for e.g. EC, 2012; Jones *et al.*, 2012), this report has presented updated information on the concepts and definitions of the soil threats, processes of soil degradation occurring in wind erosion, water erosion, decline of OM in peat and minerals soils, compaction, sealing, contamination, salinization, desertification, flooding and landslides and loss of biodiversity in soils. Moreover, it has identified a list of knowledge gaps on soil degradation in Europe (Table 14.15). The report has also produced maps that show the level of soil degradation (defined as low, medium and high) that covers a large part of Europe using NUTS-level 3 areas. These maps give a general overview of the current status of soil degradation levels at the European scale, except for EEA countries, some Balkan countries and Turkey. The influence of each soil threat on other soil threats, their interactions, and the interactions of soil threats on the six soil functions and associated soil-based ES has been described in qualitative terms in the report. The report has presented a proposal on how to develop an operational framework of the ES concept for the RECARE project. The RECARE ES framework presented is still a draft and will be further developed based on feedback from RECARE partners and other contributors.

Despite the above mentioned results, there is still a large uncertainty and lack of quantitative information on, for example, the interactions between the soil threats and the influence of soil threats on soil functions and ES. There is a need for further research on these issues in order to achieve an improved overview of existing information on soil degradation at the European scale.

Objective 2: To assess the influence of climate and human activities upon regulating key soil properties, soil functions and ecosystem services.
Climate and human activities are one of the soil forming factors in addition to topography, vegetation, parent material, and time. Adverse climatic conditions and inappropriate human activities on land use can lead to loss of soil quality and as a consequence to degradation of soil properties. In this report, the influence of climate as a direct and/or indirect driver to the soil threats has been reviewed and discussed. For more information, one can refer to the synergy chapter 14.2, which deals with climate and human drivers, including policies to soil threats. Table 14.3 shows that for most of the soil threats, climate is an important driver. In the proposed ES Framework for RECARE (Fig. 13.4), climate as a driver is not specifically mentioned. Here, the driver is defined as a 'natural driving force', with 'geology' as an example. Evidently, when it comes to the influence of climate on key soil properties, soil functions and ES, the information presented in this report is very general. This might be due to a lack of information and data in the literature.

The human drivers and socioeconomic pressures on the soil threats have been reviewed and discussed in the report. These drivers entail policy interventions, population growth, urbanization, industrialization, technological development and others, which are possible causes for number of soil threats. For instance, the expansion of tourism and agricultural intensification through irrigation along the coastal lines are two of the causes for salinization in Europe. Wildfires result in the loss of OM in peat as well as mineral soils and declining of biodiversity in soils. The policy issue will be dealt with in WP9 of the RECARE project. Despite the lack of information in the literature regarding the influence of policies on soil threats (for e.g flooding and landslides), background information is provided for WP9 to carry out an in-depth analysis of the policy effects on soil functions and ES.

Objective 3: To provide a base for RECARE's data collection and methods in the Case Study sites.
Each soil threat chapter provides information on a list of key indicators and methods to assess the indicators. Previous studies on indicators and methods to assess soil threats in Europe were reviewed and the information was collated and synthesized. A range of indicators used to assess the soil threats has been presented in tables and described in each chapter. Out of these, TOP3 indicators for each soil threat were developed by adopting some of the TOP3 indicators identified by the ENVASSO project (Huber *et al.*, 2008). In addition, suggestions made from this study by the authors were added to the list. The indicators are presented in Tables 14.8 to 14.13. In WP5 of the RECARE project, the list can be used to select the most

suitable indicators to monitor a given soil property that determines the prevailing soil threat in an area by taking into consideration resources availability, the data quality required and other relevant factors.

Each chapter of the soil threats has reviewed various literature pertaining to methods of indicators and their applications. The methods, models and procedures commonly used to measure or estimate the TOP3 indicators in each soil threat are presented in Tables 14.8 to 14.13. A list of references are also given in order to provide more information on how to apply the methods in the field or laboratory, collect samples and also analyze and interpret the data. WP6 can choose which methods are most applicable to assess which indicator and which soil threat to monitor in which period. This report provides a basis for RECARE's data collection and methods at the Case Study sites. WP3 of the RECARE project will develop standardized and harmonized procedures that can be applied across Europe so that the results from various areas in Europe are comparable and the information easily shared.

Table 14.15: List of knowledge gaps on the soil threats as extracted from the chapters on soil threats.

Soil threats	Knowledge gaps/research needs
Soil erosion by water	• Lack of harmonization on which methods/models to use over which spatial and temporal scale.
Soil erosion by wind	• Lack of comprehensive knowledge where and when wind erosion occurs in Europe and the intensity of erosion that poses a threat to agricultural productivity
Decline in OM in peat soils	• Lack of standard definition of peat soils and calculation method of CO_2 emissions from peat soils
Decline in OM in mineral soils	• Overestimation of SOC levels in the topsoils • Lack of accurate SOC estimations and the lack of tools to conduct scenario analyses, especially for agricultural soils.
Compaction	• few measured data on subsoil compaction across Europe
Sealing	• Inconsistence on land take rate data at European level due to different methodologies applied by the countries.
Contamination	• In light of the potential impact of heavy metals on aquatic life and human health, the lack of knowledge regarding their behavior in the environment and the deficiency in analytical and sampling techniques, action is urgently required.
Salinization	• Systematic data on soil salinization trends across Europe are not available
Desertification	• There is a lack of standardized procedures for desertification assessment and lack of integrated maps for desertification in Europe. An integrated framework is needed to enable meaningful, repeatable and comparable assessment of desertification (Vogt *et al.* 2011).
Flooding and landslides	• lack of information regarding the impacts of EU policies towards flooding and landslides
Decline in biodiversity	• no universal method that provides an overall measure of soil biological health (Black *et al.* 2011; Ritz *et al.* 2009)

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Annex I

ES and soil related projects in Europe and their characteristics

Project Name	Funder	Full title	ead Institutio	cosystem ser	Website	Baseline		Managemen	Policy		
						Monitoring/	Impacts	Managemen	Decision sup	Valuation	Policy appro
MULTAGRI	ERA-NET RURAGRI	Rural development throug	Lund Univers	Mechanisms	None	yes	yes	yes	yes	yes	yes
AGFORWARD	FP7	Agroforestry that will enha	Cranfield Uni	Aims include	http://www.a	yes	yes	yes	no	no	yes
SOIL SERVICE	FP7	Conflicting demands of lar	Lund Univers	Understand h	http://www4.	yes	yes	yes	yes	no	no
LIBERATION	FP7	Linking farmland Biodivers	ALTERRA	Range of wor	http://www.f	yes	yes	yes	no	yes	yes
STEPS	FP7	Status and Trends of Europ	Reading Univ	WP5: Empiric	http://www.s	yes	yes	yes	no	no	yes
FuturES	Research Centre (Leupha	Futures of Ecosystem Servi	Leuphana Un	Research gro	http://www.l	yes	yes	yes	yes	yes	yes
SoilTrEc	FP7	Soil Transformations in Eu	University of	examine soil	www.soiltrec	yes	yes	yes	?	no	no
PLUREL	FP6	Peri-urban Land Use Relati	Copenhage U	Develop strat	www.plurel.n	yes	yes	yes	yes	no	no
UK National Ecosystem As	National consortium			Enable identi	http://uknea.	yes	yes	yes	no	yes	yes
TEEB	Large consortium of donc	The Economics of Ecosyste	TEEB	Draw attentio	www.teebwe	yes	yes	yes	yes	yes	yes
Remedinal, Madrid	La Comunidad de Madrid	Restauracion ecologica en	?	To improve b	http://www.r	Yes	yes	yes	no	no	no
LandSFACTS	Funded by various grants	Landscape scale functiona	The Macaula	a modelling t	http://www.r	no	yes	no	yes	no	no
Ecosystem Services Partn	Foundation for Sustainab	The Ecosystem Services Pa	Environmenta	Worldwide ne	http://www.e	yes	yes	yes	yes	yes	yes
Environmental Change Ne	NERC	Environmental Change Net	The Centre fo	Long term mo	http://www.e	yes	yes	yes	no	no	no
EPSRC grant EP/F007604/	EPSRC	An evidence based method	Loughboroug	Examining im	http://gow.ep	yes	yes	yes	no	no	no
UKPopNet Linking biodive	NERC	The UK Population Biodive	?	Funded/parti	http://www.r	yes	yes	?	?	?	?
MOUNTLAND	Swiss grant? ETH	Prioritization for adaption t	ETH, Zurich	Provide mana	http://www.c	yes	yes	yes	no	no	yes
NOMIRACLE 003956	FP6	Novel Methods for integrat	JRC	Developing ri	http://www.p	no	yes	no	no	no	no
EcoFINDERS	FP7	?	Aarhus Unive	Objective to	http://ecofind	yes	yes	no	no	yes	yes
RUBICODE (036890)	FP6	Rationalising Biodiversity (	?	Biodiversity f	http://www.r	yes	yes	yes	no	no	no
BiodivERsA (ERA Net)	FP7 Era Net		INRA	Network fo 2	http://www.biodiversa.org/2						
SENSOR	FP6	Tools for Environmental, Sc	Leibniz centr	Impact asses	http://www.i	yes	yes	yes	yes	no	no
Forest Trends	Charity		Forest Trends	Various initia	http://www.f	no	no	yes	no	yes	yes
REGKLAM (BMBF 01LR080	Germany Federal Researc	Regionales Klimaanpassun	Leibniz Institu	Module three	http://www.r	yes	yes	yes	no	no	no
EcoChange	FP6	Biodiversity and ecosysten	CNRS	Develop futu	http://www.e	yes	yes	yes	no	yes	no
Academy of Finland 11038	Academy of Finland		The Finnish F	Understandin	http://www.r	no	yes	no	no	no	yes
Greenhance	Academy of Finland	Enhancing urban biodivers	University of	How to enha	http://www.h	yes	yes	yes	no	?	no
Jena Experiment	DFG	Exploring mechanisms und	University of	Long term mu	http://www.t	yes	yes	yes	no	no	no
ECOMIC-RMQS project (F	ANR	?	INRA	Soil biodivers	http://prodin	yes	yes	yes	no	no	no
EUROPEAT	FP6?	Tools and scenarios for su	Wgeningen?	To understan	http://levis.s	yes	yes	yes	no	no	no
Soil Infrastructure, Interfa	Denmark research counci	Soil Infrastructure, Interfac	Aarhus Unive	Explore how s	https://djfext	yes	yes	yes	no	no	no
OPENLOC	Autonomous Province of T	Innovation policy and its e	University of	Aims to defin	http://www.c	no	no	no	no	yes	yes
CONNECT	?	?	Institute for E	Relationship	http://www.c	yes	yes	yes	yes	yes	yes
OPERAs	FP7	Ecosystem Science for Poli	University of	developing e	http://www.c	yes	yes	yes	yes	yes	yes
VOLANTE	FP7	Visions of Land Use Transi	Alterra	Examining pr	http://www.v	no	no	no	no	yes	yes
REGSUS (Finland)	Academy of Finland	Regional Sustainability - ec	University of	The use of in	http://users.u	yes	yes	yes	yes	yes	no
CLIMES	Academy of Finland	Climate change impacts on	Finnish Envir	Spatial mode	http://www.a	no	no	no	yes	yes	yes
EuroDiversity AgriPopes	European Science Founda	Agricultural Policy Induced	Swedish Univ	Examine ecos	http://www.a	yes	yes	yes	no	no	no

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