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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.

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CONTENTS

1 INTRODUCTION

Soil is one of our most important natural resources that provides us with vital goods and services to sustain life. Nevertheless, soils functions are threatened by a wide range of processes and a number of soil threats have been identified in Europe (Van Camp. *et al.*, 2004; Bellamy *et al.*, 2005; Funk and Reuter, 2006; Tóth et al, 2008; De la Rosa *et al.*, 2009). Although there is a large body of knowledge available on soil threats in Europe, the complexity and functioning of soil systems and their interaction with human activities, climate change, and ecosystem services (ES), is still not fully understood. This is due to: (i) the knowledge on soil threats in Europe is scattered over numerous and diverse publications, thus hindering an integrated approach; (ii) existing reports or guidelines with regard to soil threats are rather qualitative or descriptive and do not allow selection of effective prevention, mitigation and restoration measures (Jeferry *et al.*, 2010; EC, 2012); and (iii) even the existing scientific knowledge on soil degradation is not sufficiently linked to land management measures and is not sufficiently implemented by the end users (Bouma, 2010). To address these issues, the RECARE project (*Preventing and Remediating Degradation of Soils in Europe through Land Care*) was launched in November 2013. The RECARE project aims at the development of effective prevention, remediation and restoration measures using an innovative trans-disciplinary approach, actively integrating and advancing knowledge of stakeholders and scientists in 17 Case Studies, covering a range of soil threats in different bio-physical and socio-economic environments across Europe. The project consists of 11 work packages (WP).

This report presents the result of WP2 of the RECARE project. One of the objectives of WP2 (*Base for RECARE data collection and methods*) is to provide an improved overview of existing information on soil threats and degradation at the European scale. The report is written by a group of experts from the RECARE team, coordinated by Bioforsk. In total, 60 persons were included in the process of writing, reviewing and editing the report. Eleven soil threats were identified for the report. These soil threats are soil erosion by water, soil erosion by wind, decline of organic matter (OM) in peat, decline of OM in minerals soils, soil compaction, soil sealing, soil contamination, soil salinization, desertification, flooding and landslides and decline in soil biodiversity. In the review process, the WP2 team organised a workshop on soil threats and ecosystem services to improve our understanding and knowledge on the concepts and applications of ES and soil functions. Most of the authors/co-authors of the respective soil threats have attended the workshop.

This report provides comprehensive, thematic information on the major soil threats of Europe with due attention given to the Driving force-Pressure-State-Impact-Response to soil threats. The report is organized into 14 chapters and extended annexes. Each chapter on a specific soil threat consists of seven sections annexed with references, and gives short descriptions on the following:

- definition of the soil threat and processes involved
- state of the soil degradation,
- drivers/pressures (including climate, human activities, policies),
- key indicators of the soil threat,
- methods to assess the soil threat,
- effects of the soil threat on other soil threats,
- effects of the soil threat on soil functions.

Chapter 1 gives general background information on the content of the report. Chapters 2 to 12 deal with each soil threat identified in this project. Chapter 13 describes the concepts of soil functions, frameworks of ecosystem services, measuring, monitoring, and mapping ES, valuing ES, analysis of the operationalization of the soil ES concept in European research projects, and adapted soil function and ecosystem services framework for RECARE. The last chapter 14 provides a synthesis of the report with regard to (i) the impact of the main drivers (climate, human, policy) on soil threats, (ii) state of the soil degradation in Europe, (iii) list of key indicators, (iv) methods/procedures to assess soil degradation in each soil threat using key soil properties, (v) interaction between the soil threats, (vi) effects of soil threats on soil functions & ES and finally (vii) concludes with the implications from RECARE perspectives.

We hope this report will be a valuable document, which can be useful to the scientific community in general (including researchers, students, and scholars), senior managers, and policy makers working with agriculture, water, food security issues and development agencies.

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2 SOIL EROSION BY WATER

Jacob Keizer, Hakan Djuma and Volker Prasuhn

2.1 *Description of soil erosion by water*

Soil erosion in general can be defined as a three-phase process that consists of: (i) the detachment of individual soil particles from the soil mass; (ii) their subsequent transport by an erosive agent; and, ultimately, (iii) their deposition when the erosive agent lacks sufficient energy for further transport (Morgan, 2005). In the case of soil erosion by water, both rainsplash and water running over the soil surface detach and then move the detached particles, but rainsplash is the most important detaching agent whereas running water is the principal transporting agent. The transport of soil particles resulting from the direct impact of falling raindrops is designated as rainsplash erosion, while the transport of soil particles by running water is commonly divided into interrill and rill erosion. Interrill erosion then refers to water running as a shallow sheet ("overland flow") and removing a relative uniform thickness of soil, whereas rill erosion refers to water running as concentrated flow and removing soil by "digging out" channels of increasing deepness and/or width. In turn, rill erosion is generally divided into rill and gully erosion depending on channel dimensions. A cross-sectional area of at least 1 ft^2 (Poesen, 2003) is a widely recognized criterion to distinguish gullies from rills. Poesen (2003) calculated that at larger scales around 80% of detachment/soil loss comes from gullies.

Not only water running over the soil surface as described above but also water moving laterally through the soil matrix in downslope direction ("interflow") can detach and transport soil particles, including as concentrated flow in macro-pores or subsurface pipes (Morgan, 2005). These subsurface erosion processes mainly occur in peatlands (Holden, 2005) as well as in areas where man-made subsurface drainage systems have been installed (Russel *et al.*, 2001).

Soil erosion appears to have been recognized by mankind since the early civilizations of China and the Mediterranean Basin (Morgan, 2005). Nonetheless, scientific research into soil erosion did not gain impetus till the 1920s and 1930s, with Hugh Hammond Bennett leading the soil conservation movement in the USA. In Western Europe, by contrast, the importance of soil erosion only started to be duly recognized from the 1970s onwards.

2.2 *State of the soil erosion by water*

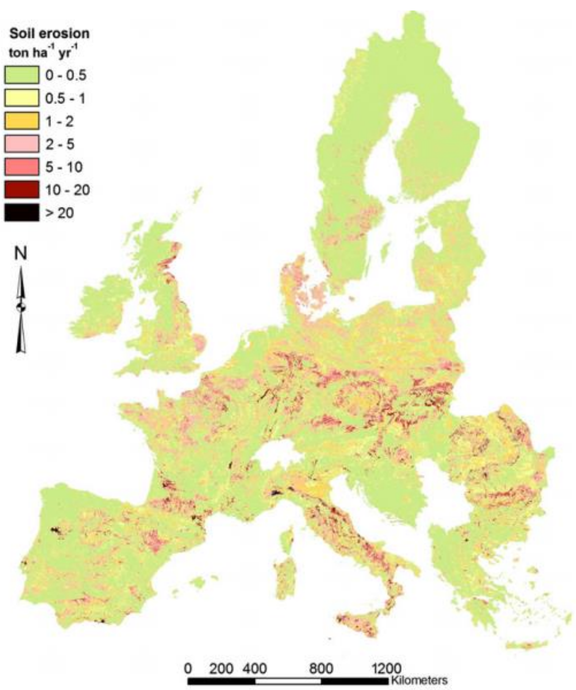

Figure 2.1: Maps of the risk of soil erosion by water across Europe based on erosion plot data (Cerdan et al., 2010).

The work done by Boardman & Poesen (2006) is so far the most comprehensive research into the extent, seriousness and impacts of soil erosion in Europe. It involved erosion experts from 33 European countries who collaborated to compile and analyse existing data and information at the national scale and/or from typical case studies, with a strong emphasis on field observations and measurements. These erosion data, however, are not always directly comparable, as there is a lack of harmonization among the different European countries on which methods, approaches and models to use over which spatial and temporal scales. Furthermore, European countries differ markedly in the amount of erosion data they have. Because of this lack of harmonised measurement data, soil erosion risk has frequently been used as a surrogate indicator in national as well as European-wide risk assessments. Risk assessment involves the identification of the risk and the quantification of the exposure to that risk (Jones *et al.*, 2004; Grimm *et al.*, 2002).

The risk of erosion by water has been assessed at the European scale using various models and expert-based approaches. The most recent attempts are those of Kirkby *et al.* (2004): applying the PESERA model, Cerdan *et al.* (2010): based on erosion plot data, Vanmaercke *et al.* (2012): based on sediment yield data, the OECD (2013), and Bosco *et al.* (2014): applying the eRUSLE model, whereas Grimm *et al.* (2002) and Jones *et al.* (2004) provided a list of earlier approaches and the corresponding maps. The former are described in more detail underneath, starting with those based on measurement data and with data collected over the smallest spatial extent. In addition, the latest OECD is briefly presented.

Table 2.1: Overwiev of recent estimates of the risk of soil erosion by water in European countries.

source	Cerdan *et al.*, 2010	Kirkby *et al.*, 2004	Bosco *et al.*, 2014	Panagos *et al.*, 2014	OECD 2013
indicator	Country-wise mean soil loss risk	Country-wise mean soil loss risk	Country-wise mean soil loss risk	Country-wise mean soil loss risk	area with mean soil loss risk > 11 ton ha^{-1} yr^{-1}
unit	ton ha^{-1} yr^{-1}	ton ha^{-1} yr^{-1}	ton ha^{-1} yr^{-1}	ton ha^{-1} yr^{-1}	%
estimates based on	erosion plots	PESERA	eRUSLE	EIONET	OECD
Austria	1.6	0.5	4.8	0.7	3
Belgium	1.4	1.1	2.3	3.7	9
Bulgaria	1.9	0.6	2.2	1.9	
Czech Republic	2.6	1.3			4
Denmark	2.6	2.3			
Finland	0.2				0
France	1.5	1.6			4
Germany	1.9	0.9	2.7	1.4	
Greece	0.8	5.8			20
Hungary	1.0	0.4			25
Ireland	0.5	0.2			
Italy	1.0	3.1	7.4	6.6	30
Latvia	1.3	0.1			
Lithuania	1.0	0.3			
Luxembourg	1.3	0.5			25
Netherlands	0.4	0.1	0.6	0.3	0
Norway	0.2				3
Poland	1.5	0.7	1.6	1.5	29
Portugal	1.2	4.6			
Romania	1.8	0.4			
Slovakia	3.2	1.3	2.3	1.0	55
Slovenia	1.2	0.9			38
Spain	1.0	2.4			28
Turkey	0.3				39
United Kingdom	0.9	0.3			17

(i) Erosion plot data (Figure 2.1): Cerdan *et al.* (2010) compiled data from 81 experimental sites in 19 countries, amounting to a total of 2,741 plot-years, and calculated mean inter-rill and rill erosion rates for the area in Europe covered by the CORINE database. The authors used correction factors for topography and soil properties to extrapolate the plot data to the European scale, producing a map with a 100 m resolution. The estimated inter-rill plus rill erosion rates were, on average, 1.2 ton ha-1 yr-1 for the whole CORINE-covered area and 3.6 ton ha-1 yr-1 for the arable lands within that area. These estimates were much lower than earlier estimates, as these earlier figures involved erroneous extrapolation of local plot measurements. Erosion rates were comparatively high (2–10 ton ha-1 yr-1) in the hilly loess areas of Western and Central Europe, and revealed marked spatial variation in the Mediterranean Zone, being high in many areas in Italy (Apennine slopes and Sicily) as well as in some areas in Spain (southern part of the Guadalquivir basin and the area around Zaragoza). Erosion rates also varied strongly for Europe as a whole, as 70% of the total erosion originated from 15 % of the territory. At the country level, the highest mean erosion rates were predicted for Slovakia, Denmark, Czech Republic and Italy (Table 2.1). Erosion rates further differed markedly between land covers. The highest rates

(ii) were estimated for vineyards (17.4 ton ha-1 yr-1), arable lands (3.6 ton ha-1 yr-1) and orchards (3 ton ha-1 yr-1), respectively, whereas all other land uses revealed mean values well below 1 ton ha-1 yr-1.

(ii) Sediment yield data: Vanmaercke *et al.* (2012) compiled annual sediment yield data for 1,794 catchments in Europe, which corresponded to at least 29,203 catchment-years of observations. They compared these data with annual erosion rates (n = 777) from runoff plots located at 187 study sites that were relatively well spread across Europe as well as with the above-mentioned map produced by Cerdan *et al.* (2010) and the PESERA map produced by Kirkby *et al.* (2004). The authors found that the sediment yield data and the runoff plot data indicated significantly higher soil loss rates than the two maps, even though sediment yields do not take into account that large proportions of eroded sediment may be deposited before reaching the catchment outlet. To the authors, this clearly demonstrated the importance of erosion processes other than inter-rill and rill erosion for catchment-scale sediment yields, in particular gully erosion, channel erosion, mass movements, and glacial erosion. These findings were later confirmed by De Vente *et al.* (2013). Thus, soil erosion by water is only one possible source of the sediments that leave a catchment outlet, as a result caution is needed when comparing soil erosion rates and sediment yields (Verheijen *et al.*, 2009).

(iii) PESERA model predictions (Figure 2.2): the Pan-European Soil Erosion Risk Assessment (PESERA) model is a process-based and spatially distributed model that was developed to estimate the risk of soil erosion by water across Europe (Kirkby *et al.*, 2004). The PESERA results were also selected by the OECD as basis for its agri-environmental indicator of soil erosion (IRENA fact sheet No. 23; EEA, 2006). According to PESERA, about 105 million ha or 17% of the total land area of Europe (excluding Russia) is subject to some degree of soil erosion risk. Furthermore, Europe can be divided in three zones w here erosion risk is significant: (i) a southern zone characterised by a severe risk of erosion by water; (ii) a northern loess zone with a moderate risk; and (iii) an eastern zone where the two prior zones overlap. Within all three zones, however, hot spots of soil erosion risk do occur. At the country level, Greece, Italy, Portugal, Italy and Spain stand out with the highest mean annual rates of soil erosion risk (Table 2.1). Spain is the country with the largest area subject to a high erosion risk, comprising southern and western Spain and covering 44% of the country's territory. Portugal ranks second, with one-third of its territory country revealing a high erosion risk. In Central and Eastern Europe, soil erosion risk is most widespread in Bulgaria and Slovakia, affecting some 40% of the territory of both countries.

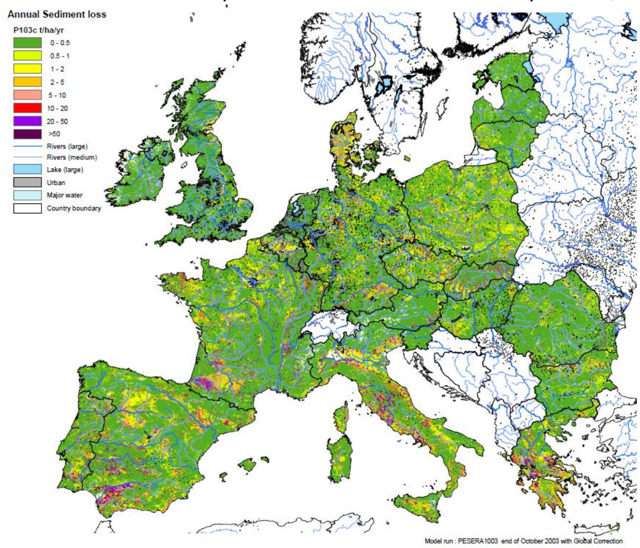

Figure 2.2: Maps of the risk of soil erosion by water across Europe based on PESERA model predictions (Kirkby et al., 2004).

(iv) eRUSLE model predictions (Figure 2.3): Bosco *et al.* (2014) presented a new, extended version of the Revised Universal Soil Loss Equation (RUSLE). The authors validated their eRUSLE predictions through comparison with national datasets as well as based on expert judgement. The eRUSLE results indicated that 130 million ha in the EU-27 countries are at risk of being affected by soil erosion by water and that this risk is moderate to high for about 14 % of the European territory. Almost 20% was subjected to soil loss in excess of 10 ton ha^{-1} yr^{-1} (EEA, 2012). Soil erosion rates exceeding 11 ton ha^{-1} yr^{-1}, defined as moderate to severe erosion by the OECD, were foreseen to affect just over 7% (= 115,410 km^2) of the cultivated lands (arable and permanent cropland) in the EU-24 (excluding Greece, Cyprus and Malta) (Jones *et al.*, 2012). The average rate of soil erosion by water across the EU-27 (excluding CY, GR and MT) was estimated at 2.76 ton ha^{-1} yr^{-1}; rates were higher in the EU-15 (3.1 ton ha^{-1} yr^{-1}) than in the EU-12 (1.7 ton ha^{-1} yr^{-1}), probably as the EU-15 includes the Mediterranean area where overall erosion rates were higher.

(v) OECD assessment: the OECD assessed soil erosion risk through questionnaires to the experts of the individual countries, using a standard table linking erosion risk to erosion rates (OECD, 2013). The OECD's table classifies soil erosion risk into five categories ranging from tolerable (< 6 ton ha^{-1} yr^{-1}) to severe erosion (> 33 ton ha^{-1} yr^{-1}). However, not all countries employed the class limits proposed by the OECD and, in particular, various countries use lower upper thresholds for the class of tolerable soil erosion. During the period 1990-2010, nine of the 20 European OECD member countries had more than 20% of their agricultural lands exposed to a moderate to severe erosion risk. These nine countries were Slovak Republic, Turkey, Slovenia, Italy, Poland, Spain, Luxembourg, Hungary and Greece (Table 2.1).

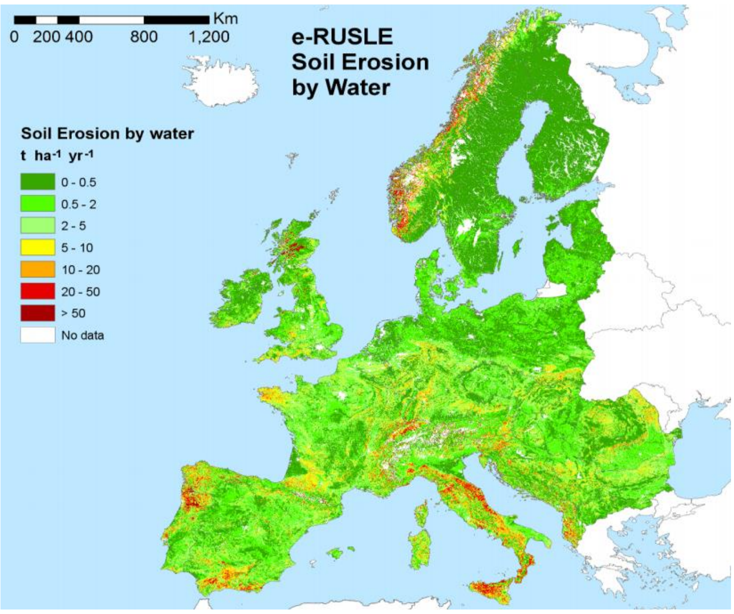

Figure 2.3: Maps of the risk of soil erosion by water across Europe based on eRUSLE model predictions (Bosco et al., 2014).

The results of the various erosion risk models and approaches that have been applied at the European-scale differ quite considerably. This relates to differences in modelling approaches, differences in model input data and their quality as well as to differences in the models' spatial and temporal resolutions. Model input data lacking sufficient quality and/or spatial resolution can result in substantial errors and uncertainties in model predictions. Also the geographical extents of the model-based assessments differ, depending on the countries that are being considered as European.

The fundamental importance of model input data is well illustrated by the following three examples.

Example 1: Panagos *et al.* (2014a) presented new values for the soil erodibility factor K based on a pan-European harmonised soil data set and taking into account stoniness. The countries with the lowest mean value for the K-factor (< 0.025 ton ha^{-1} h^{-1} ha^{-1} MJ^{-1} mm^{-1}) ranged from Portugal, Ireland, Denmark, Greece, Netherlands, United Kingdom, Estonia to Finland, whereas the countries with the highest mean value for the K-factor (> 0.032 ton ha^{-1} h^{-1} ha^{-1} MJ^{-1} mm^{-1}) included Belgium, Luxembourg, Czech Republic, Hungary and Slovakia.

Example 2: Panagos *et al.* (2014b) compared soil losses predicted by PESERA with the plot data compiled by Cerdan *et al.* (2010) as well as with the national data from eight countries collected through EIONET-SOIL. Overall, the PESERA figures were not only lower than the mean soil losses of the EIONET data set but also than those of the Cerdan *et al.* (2010) data set, except in the case of Italy.

Example 3: Hessel *et al.* (2014) applied the MESALES model (Modèle d'Evaluation Spatiale de l'ALéa Erosion des Sols) to three geographical areas using two distinct soil data bases. This resulted in noticeable differences in soil erosion risk, in spite of the fact that the risk estimates were on a semi-quantitative scale ranging from very low to very high.

Model assessment is often constrained by a lack of measurement data with the necessary spatial resolution, so that it is often impossible to determine which of the models is performing best. Nonetheless, it is widely recognised that a model such as RUSLE tends to overestimate soil losses. Furthermore, model-based estimates can be expected to overestimate soil erosion risk, since soil conservation measures are not taken into account. This is first and foremost due to the absence of EU-wide data on the application of practices such as sequential cropping, reduced tillage and strip tillage. By contrast, plot-based studies and models such as PESERA and eRUSLE may underestimate soil erosion rates, since they assess inter-rill and rill erosion but not (ephemeral) gully erosion. The existing measurements of gully erosion rates mainly concern the Mediterranean region of Europe. They revealed a huge variation, with figures ranging from 1 to 455 ton ha^{-1}

yr^{-1}, depending on rainfall and site conditions. Ephemeral gullies at four sites in Belgium were estimated to produce medium-term soil losses between 3.2 and 8.9 ton ha^{-1} yr^{-1} (Verheijen *et al.*, 2009).

The thresholds above which soil erosion should be regarded as a serious problem continue to be controversial, including because soil formation processes and rates seem to differ substantially across Europe (Bosco *et al.*, 2014). Nonetheless, direct measurements of soil formation rates are very scarce. Soil formation rates (by weathering) in Europe under current conditions are estimated to vary in between 0.3 and 1.2 ton ha^{-1} yr^{-1} (Verheijen *et al.*, 2009). At such slow rates of soil formation, soil losses exceeding 1 ton ha^{-1} yr^{-1} can be considered irreversible and unsustainable within a time span of 50-100 years (Jones *et al.*, 2004; Verheijen *et al.*, 2009). Soil losses ranging from 5 to 20 ton ha^{-1} yr^{-1} can have serious impacts, both at the site where the soil is lost and off-site in downstream flood zones and aquatic habitats. Soil losses of 20 to 40 ton ha^{-1} yr^{-1} by individual storms with a return interval of two or three years are measured regularly in Europe, whereas extreme rainfall events have been found to produce soil losses exceeding 100 ton ha^{-1} yr^{-1}. Such large soil losses can have catastrophic on-site effects as well as serious off-site consequences (Grimm *et al.*, 2002).

An alternative approach to modelling soil erosion by water is to represent the role of running water in an explicit manner, predicting the generation of runoff as well as its detachment and/or transport capacity. The blueprint for this approach was presented as early as 1969, by Meyer & Wischmeier (1969), but it was not implemented until more than a decade later, in the semi-empirical model of Morgan, Morgan and Finney (MMF); Morgan *et al.* (1984) as well as in the bulk of the physically-based models (e.g. CREAMS by Knisel, (1980), WEPP by Nearing *et al.* (1989), EUROSEM by Morgan *et al.* (1998) and PESERA by Kirkby *et al.* (2004). Due to their large data-demands, such models are difficult to apply at EU-scale.

2.3 Drivers/pressures

The factors controlling soil erosion are commonly divided into:
(i) erosivity of the erosive agent or its capacity to detach and transport soil particles;
(ii) erodibility of the soil or the inverse of the soil's resistance against the detachment and transport of its particles;
(iii) plant and litter cover; and
(iv) slope of the terrain (Morgan, 2005).

In the case of soil erosion by water, erosivity typically focuses on the detaching power of raindrops, ignoring that of running water, whereas erodibility usually refers not just to the soil's resistance to rainsplash and running water but also to the likelihood that water will actually be running over the soil surface. This conceptual framework is intimately linked to the Universal Soil Loss Equation (USLE) by Wischmeier & Smith (1978) which estimates annual soil losses from plots, fields and hillslopes as the product of the four above-mentioned factors. USLE does, in fact, include a fifth multiplication factor but specifically to predict how effective land management practices such as contouring and bench terracing are to reduce soil losses.

Climate drivers: Climate and, in particular, rainfall is the primary driver of soil erosion by water. Rainfall is not only the main agent of detachment of soil particles but also the principal source of water running over the soil surface (Morgan, 2005). In cold climate regions, however, also freezing-thawing cycles can play a key role in detachment, while snow melt can be an important additional source of runoff. The erosivity of rainfall is typically related to the kinetic energy of the raindrops striking the soil surface and, as such, calculated as a function of the intensity and duration of a rainfall event as well as of the mass, diameter and velocity of the raindrops. The measurement of these raindrop characteristics has long posed considerable challenges, at least till the development of disdrometers. Therefore, the kinetic energy of rainfall is typically estimated based on its relationship with rainfall intensity, often using relationships that are adjusted to local climate conditions. Nonetheless, these local relationships can reveal marked variations between and within individual rain storms, especially depending on their origins in terms of synoptic weather conditions (e.g. convectional vs frontal rain) and on wind speeds. The rainfall-runoff response of soils is typically explained as a function of the two main runoff generating processes. Infiltration-excess overland flow occurs when rainfall intensity exceeds a soil's so-called infiltration capacity or, in other words, the rate at which a soil can take in water that has accumulated at its surface. By contrast, saturation overland flow occurs when a soil's water storage capacity has been exceeded, typically due to prolonged antecedent rainfall.

Climate also affects soil erosion by water indirectly, through its impacts on soil properties, soil cover by (whether of (semi-natural) vegetation or of croplands and sown pastures) as well as interactions between these impacts. For instance soil properties strongly determine a soil's infiltration and storage capacities and, thus, its hydrological response. This includes properties that tend to be time-invariant such as soil texture, soil depth and the presence of impermeable layers as well as properties that vary markedly in time such as the presence of a surface crust, soil aggregate stability, soil water repellency or groundwater level. In the case of soil hydrological properties, the indirect role of climate is well-illustrated by the importance of dry spells in the formation of a structural surface crust or in the appearance and severity soil water repellency. In the case of soil properties determining erodibility, the indirect role of climate can be exemplified by the marked increase that freezing and thawing can produce (Coote *et al.*, 1988). In the case of plant cover, the indirect role of climate is perhaps most obvious in semi-arid and arid regions, where the protective cover provided by plants against rainsplash tends to decrease with increasing aridity.

Human drivers: Arguably, human activities have become the most important driver of soil erosion by water in modern times and places, especially those witnessing strong increases in population and/or rapid advances in slope- and landscape-engineering capabilities. The concept of a new geological age – the Anthropocene – has become a topic of serious debate (Zalasiewicz *et al.*, 2011), including based on the observed and modelled impacts of humans on sediment flux (Syvitski and Kettner, 2011). The paramount importance of human activities in soil erosion by water is also evidenced by the commonly-made distinction between "natural" (or "geological") erosion rates and human-induced, "accelerated" erosion rates (Verheijen *et al.*, 2009). In turn, the concept of accelerated erosion is closely linked to that of tolerable soil erosion, as (changes in) land management are implied in avoiding to exceed "any actual soil erosion rate at which a deterioration or loss of one or more soil functions does not occur" (Verheijen *et al.*, 2009).

Human activities can accelerate soil erosion by water in a wide variety of ways but always in an indirect manner, by provoking changes in especially the first three of the erosion-controlling factors listed at the beginning of the present section. Some examples will follow to illustrate this for each of these three factors. Rainfall erosivity is expected to increase under likely climate change scenarios, especially in Mediterranean climate regions as autumn rainfall events become more intense. Erosivity of surface runoff can be enhanced by ploughing, leading to concentration of overland flow in furrows and to reduction of micro-topographic variations and, thereby, of the resistance to flow. Overland flow generation can be enhanced by compaction of the topsoil in the wheel tracks of heavy machinery, provoking a reduction in infiltration capacity. Soil erodibility can be enhanced by ploughing, both directly by destroying soil aggregates and indirectly by reducing soil organic matter content and, thereby, the formation of new aggregates. Soil cover will typically be less in croplands than in the original vegetation, leading to an overall reduction in the protection of the soil surface against rainsplash as well as in the resistance to overland flow.

In contrast, human activities can also reduce accelerated and even natural rates of soil erosion by water, through so-called soil conservation techniques (Morgan, 2005). Soil conservation techniques can be divided in three groups: agronomic, vegetative, structural and management. In a nutshell, agronomic measures target plant cover, soil management measures aim at soil erodibility and infiltration capacity, and mechanical measures are directed towards terrain shape and drainage network, often involving engineering solutions. Typical examples of these three measures are mulching with organic residues, contour-tillage and terracing, respectively. Bench terraces have existed for over 2000 years and from ancient civilizations across the globe.

Socio-economic-politics drivers: land use and land management, which are influenced by the socio-economy and policy, can have an important impact on soil erosion by water (Schwilch *et al.*, 2012). Nonetheless, more detailed assessments of the impacts of specific socio-economic factors and of past and present agricultural, forestry and soil conservation legislation and plans seem to be lacking. Such assessments also have a requisite that is typically lacking: adequate erosion monitoring schemes. For example, the common-grounds in the mountains of Portugal were afforested on a large scale by the "Estado Novo" ("New State") following the 1930s, among others on the official grounds of preventing the silting-up of rivers and the truncation of soil profiles (Estevão, 1983). The effectiveness of this afforestation plan in terms of erosion reduction, however, cannot be easily quantified, as no erosion measurements were carried out before and after afforestation and/or to compare afforested and non-afforested lands.

With respect to political drivers, one notable example is the Norwegian political decision to subsidise farmers who levelled their fields in the 1970's. Land levelling causes very high erosion in Norway (Lundekvam *et al.*, 2003) resulting in new guidelines and regulations prohibiting land levelling without specific permission.

2.4 Key indicators of soil erosion by water

The European Environmental Agency (EEA, 2000) used the driving force–pressure–state–impact-response (DPSIR) framework (which also underpins the approach by RECARE) to identify a list of agri-environmental indicators of soil erosion by water that were considered relevant to pan-European policy making. This section will focus on the two indicators of the state of soil erosion and the combined indicator of state and impact, i.e. area affected by soil erosion (in km^2), extent of area affected by soil erosion (in %), and magnitude of soil erosion or sediment delivery (in tons), respectively.

Gobin *et al.* (2004) critically reviewed the EEA indicators in terms of policy relevance and utility, analytical soundness (including data availability) and measurability, and based on this analysis, provided recommendations. In relation to the two state indications, the authors recommended the implementation of a combined measurement-modelling-expert approach, considering that: (i) erosion measurements by themselves are unsuitable for European-wide assessments but indispensable to validate model-based predictions of the risk of erosion across Europe under present environmental and land-cover/use conditions; (ii) expert knowledge is required for verification of regional-scale assessments of actual erosion risk. In relation to the combined state-impact indicator, Gobin *et al.* (2004) stressed that measurements of sediment yield at catchment outlets or of sediment deposition in lakes/reservoirs provide at best an indirect validation of catchment-scale model predictions. The main reasons according to Gobin *et al.* (2004) are that the origin(s) of the sediments are mostly uncertain (e.g. due to riverbank or channel erosion) and that sediment yield/deposition data typically lack the required accuracy. In addition, sediment delivery ratios (i.e. the proportions of the sediments eroded from the land surface that discharges into a river) are estimated to vary widely, from less than 5 to 90% (Walling 1983).

Measurement-based indicators of soil erosion by water can be divided into two broad classes, those referring to the actual transport process of soil particles and therefore expressed as ton ha^{-1} yr^{-1} (or equivalent unit); and those related to changes in land surface or soil characteristics resulting from soil erosion (Morgan, 2005). The transport process indicators encompass the amount of particles transported by rainsplash (splash erosion) or running water, as sheet flow (inter-rill erosion) and/or concentrated flow (rill and gully erosion). The changes in land surface or soil characteristics resulting from soil erosion indicators include differences in contents of radioactive tracers, differences in ground levels, altered soil profiles (e.g. truncated profiles without A-horizon), and the presence/extent of so-called erosion features such as pedestals, rills, gullies and recently deposited sediments. They reflect cumulative erosion processes and, thus, require a well-defined time basis to be converted into the same measurement units as the former indicators. For example in the case of rills, this can be achieved by measuring their extent and dimensions at regular intervals, in combination with measurements of the bulk density of the removed soil.

2.5 Methods to assess the status of soil erosion by water

Although models are indispensable for assessing the status of soil erosion by water for larger areas and for larger time frames (both past and future), this section will be limited to the methods used for measuring erosion as such. Measurements are a prerequisite for the validation of model predictions.

The transport of soil particles by rainsplash (splash erosion) can be measured in the field by splash boards as well as by funnels and cups of various designs, which are typically less than 15-20 cm in diameter (Morgan, 2005). Rainsplash can be measured under natural rainfall conditions as well as artificial rainfall conditions, for which a wide range of portable rainfall simulators can be employed (e.g. Iserloh *et al.*, 2013).

The transport of soil particles by sheet flow (inter-rill erosion) can be measured in the field by using plots that are sufficiently small to avoid that the overland flow occurs as concentrated flow. Arguably, inter-rill erosion has mainly been studied by means of field rainfall simulators, applying rainfall with typically high intensities but low kinetic energies to bounded plots of small dimensions rarely exceeding 1 m^2. Nonetheless, these so-called micro-plots have also been employed to measure inter-rill erosion under natural rainfall conditions, including for assessing the representativeness of the results obtained under simulated rainfall conditions.

The transport of soil particles by combined sheet and concentrated overland flow (inter-rill + rill erosion) can be measured in the field by using appropriately sized plots, typically more than 10 m long (Morgan, 2005). A widely-used approach is the so-called "Wischmeier" plot. It is standard 22 m long and 1.8 m wide, bounded by sheets of, for example, metal that stick out 150-200 mm above the soil surface, has a collecting through or gutter at the bottom end where the runoff, with its sediments, is channelled into one or more collecting tanks, depending on runoff volumes. Nonetheless, bounded plots of other designs and especially smaller dimensions have also been frequently used. An alternative design consists of unbounded plots, avoiding edge effects and possible sediment exhaustion but introducing uncertainty about the contributing area. Unbounded plots such as Gerlach troughs and sediment fences have especially been used for measuring runoff and/or sediment losses at larger spatial scales such as agricultural fields, permanent crop or tree plantations or entire hillslopes, i.e. including by gully erosion.

The transport of soil particles beyond the hillslope scale can be measured at the outlets of catchments with a hydrometric station, where typically water level is recorded continuously and sediment yield is estimated by multiplying the streamflow's suspended sediment concentration by discharge. Water level recordings are converted to discharge estimates based on the stage-discharge curve at the catchment outlet, which, in turn, is derived from discharge measurements that should ideally cover the full range of water levels. Hydraulic structures such as weirs and flumes can greatly reduce the need for repeating discharge measurements, especially if the channel section at the outlet is subject to marked changes. The suspended sediment concentration can be determined through runoff samples collected throughout runoff events, either manually or by means of one or more automatic samplers, or through turbidity recordings. While turbidity sensors have the advantage of providing continuous estimates of suspended sediment concentration, the quality of these estimates does depend critically on the relationship between the two parameters.

The status of cumulative soil erosion can be described through a survey of (selected) erosion features, mapping either their presence/absence or their extent and dimensions. These features can include pedestals (evidencing rainsplash erosion), soil profile characteristics (e.g. truncated profiles without A-horizon or profiles with buried A-horizons), rills, gullies and sediment depositions. A simple method of estimating the cumulative volume of soil removed by rill or gully erosion on a slope is to determine the cross-sectional area of the rills/gullies along a series of transects of 20-100 m long across the slope (Morgan, 2005). The bulk density of the removed soil is then needed to estimate the sediment losses by weight. A similar approach can be used to estimate the volume and weight of sediments recently deposited on hillslopes or at footslopes, measuring their length, width, depth, and bulk density. More precise estimates of the volume of removed soil/deposited sediments can be obtained by classical topographic methods, also depending on the dimensions involved. This is of particular relevance in the context of repeated surveys. Terrestrial and aerial photogrammetry, terrestrial and airborne 3-D laser scanning as well as satellite imagery can equally be useful for (repeated) mapping of erosion features, as long as the precision of the resulting digital terrain models (DTM) match the dimensions of the features and the changes therein. A dense cover of high-stature vegetation can, in this respect, be a constraining factor.

Changes in ground level can be estimated not only from sequential DTMs, as mentioned above, but also through erosion pins as well as by means of an erosion bridge (Morgan, 2005). Typically, erosion pins are installed in large numbers, and the distance between the pin's head and a washer (originally placed at the soil surface) measured at regular intervals. An erosion bridge is a device that allows the repeated measurement of the distance to the soil surface from a fixed height at fixed points along a fixed transect. Sediment pins have been used to measure the sedimentation rate in the irrigated fields particularly in spate irrigation systems where farmers divert flood water that contains soils and nutrients to adjacent irrigable fields (Tesfai and Sterk, 2002). Also changes in the level of sediments in ponds, reservoirs and lakes can be used to estimate sedimentation rates at the catchment scale. Besides sedimentation rates, the efficiency to trap these sediments must be estimated to arrive at sediment yields. Trap efficiency is particulary difficult to measure with sufficient accuracy to avoid large uncertainties in the resulting sediment yields (Morgan, 2005).

Differences in the concentrations of radioactive isotope tracers in soil profiles can provide not only qualitative information on the patterns of soil erosion/deposition in a landscape over time depending on the decay rate of the isotope, but also estimates of soil erosion rates when combined with conversion models such as the proportional approach or the mass balance model (Morgan, 2005). The most commonly radioactive isotope tracer in erosion studied has been ceasium-137. Among innovative tracers, magnetic iron oxides attached to soil particles deserve special mention as they can be measured easily, cheaply, and directly in the field (Guzman *et al.*, 2013).

2.6 Effects of soil erosion by water on other soil threats

Soil erosion by water can have an important impact on other soil threats especially for decline in soil organic matter (SOM), flooding risk, and decline in soil biodiversity.

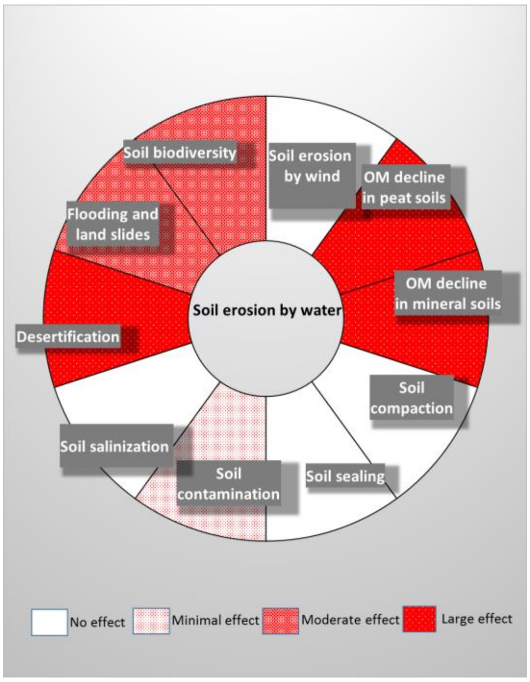

Figure 2.4: Effects of soil erosion by water on other soil threats. Red is negative

Soil erosion by water can reduce the SOM stocks of soils directly as well as indirectly. The direct effect involves the transport, by running water, of organic compounds in dissolved and especially particulate form, aggregated to mineral soil particles. This effect can be especially relevant in recently burnt areas to the extent that an ash layer is present and, arguably, that it resulted from the burning of the litter layer as opposed to the above-ground standing biomass. In the case of mineral soils, the reduction in SOM stock is further aggravated when inter-rill erosion is the predominant process, as organic matter contents tend to decrease with soil depth. The indirect effect of soil erosion results in denudation of the upper soil layer, exposing of SOM at greater soil depths to conditions propitious to its decomposition.

Soil erosion by water can also enhance flooding risk directly as well as indirectly. The same overland flow that causes soil erosion will typically constitute an important component of the hydrological response of catchments during flooding events. This is especially true for flash floods associated to infiltration-excess overland flow and less so for regional-scale floods associated to saturation overland flow due to a larger baseflow component. Flooding risk can further be increased by the silting-up of the channel network resulting from the deposition of soil eroded during prior erosion events. The sediment load carried by the water also increases the volume of the flood, and that results in larger damage off-site effects of erosion.

Soil erosion by water is often regarded as one of the most intense and widespread desertification processes (e.g. Rubio & Bochet, 1998; Vanmaercke, *et al.*, 2011). This has lead to the use of various desertification indicators that are related to soil erosion.

Eroded sediment can contain contaminants (agricultural or other) that cause contamination downstream where the sediment is deposited. Furthermore, sediment itself is also considered contamination by some. Situation becomes critical if highly contaminated sites are eroded, such as mine-spills.

Soil erosion by water can result in direct losses of soil biodiversity through the removal of soil flora, fauna and micro-organisms in the running water. This has been demonstrated for nematodes as well as seeds. Soil erosion by water can also lead to losses in soil biodiversity in an indirect manner, by changing the environmental conditions of the soil habitat, for example in terms of SOM as mentioned above.

2.7 Effects of soil erosion by water on soil functions

Soil erosion by water can affect the soil function of food and other biomass production both directly and indirectly. Possible direct effects are the removal of seeds by runoff and damage to above- and below ground plant organs. Possible indirect effects can be related to plant growth itself such as reduced rooting space for support, reduced available soil water and reduced soil nutrient pool, or to land management operations such as the removal of recently applied agrichemicals and additional efforts required to fill-up rills or circumvent gullies.

Soil erosion by water can have negative consequences for a soil's capacity for storage, filtering, buffering and transformation. In the case of storage and buffering, these consequences would seem to depend fundamentally on the net reduction of soil depth or, in other words, the difference between soil loss and soil accretion relative to the total soil stock. In the case of filtering and transformation, however, the impacts of soil erosion would seem to depend first and foremost on how important the soil layer that is being eroded is for the respective filtering or transformation process.

Soil erosion by water can be expected to have important implications for the soil function of biological habitat and, possibly, also that of gene pool if the removal of an organism by runoff is significant in terms of its existing population. The habitat effect would seem to depend strongly on the degree to which the organism depends on the topsoil for its habitat.

Soil erosion by water and especially gully erosion can affect the soil function of physical heritage by the resulting changes in the aspect of the landscape. It can also affect the function of cultural heritage through the removal and re-deposition of archeological artifacts as well as through the burial of archeological artifacts under sediments eroded upslope or upstream.

Soil erosion by water can have major consequences for a soil's function as a platform for man-made structures, either through the removal of the soil underneath these structures or through the deposition of eroded sediment again or on these structures.

Soil erosion by water can play an important role in the provision of raw materials. This is well-illustrated by sands accumulated in river beds which are exploited for civil construction purposes.

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3 SOIL EROSION BY WIND

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3.1 *Description of soil erosion by wind*

Soil erosion by wind is a serious environmental problem (Lal, 1994) causing severe soil degradation in arid, semi-arid and agricultural areas (Woodruff and Siddoway, 1965; Kalma *et al.*, 1988). It is estimated that ca. 28% of the global land area that experiences land degradation suffers from wind-driven soil erosion process (Oldeman, 1994). A total land area of 549 million ha is potentially affected by wind erosion, of which 296 million ha could be severely affected (Lal, 2001).

The movement of soil occurs when forces exerted by wind overcome the gravitational and cohesive forces of soil particles on the surface of the ground (Bagnold, 1941), and the surface is mostly devoid of vegetation, stones or snow (Shao, 2008).

Thus, wind erosion occurs where; 1) the soil is loose, finely divided and dry; 2) where the soil is smooth and bare; and 3) wind is strong. These conditions are more likely to be met in arid regions, but are not restricted to those regions. Funk *et al.* (2002), for example, report that wind erosion is a serious problem in the north-eastern parts of Germany because the months of highest wind erosivity (March & April) coincides with seedbed preparation for crops like sugar beet and maize. The basic factors that control wind erosion are wind speed, soil characteristics and vegetation conditions (Fryrear & Bilbro, 1998).

Wind erosion has always occurred as a natural land-forming process (Livingstone and Warren, 1996). Extensive aeolian deposits from past geologic eras prove that this is not a recent phenomenon (Skidmore, 1994; Haase *et al.*, 2007). However, today the rates of wind erosion are locally accelerated by inappropriate land management (e.g. leaving cultivated lands fallow for extended periods of time, overgrazing rangeland pastures and, to a lesser extent, over-harvesting vegetation (Leys, 1999)). In agricultural lands, soil erosion by wind mainly results in the removal of the finest and most biological active part of the soil richest in organic matter and nutrients (Funk and Reuter, 2006). Repeated exposure to wind erosion can have permanent effects on agricultural soil degradation, making it difficult to maintain favourable soil conditions in the long run (Jönsson, 1994).

3.2 *State of the soil erosion by wind*

Land degradation due to wind erosion is also an European phenomenon (Warren, 2003) which locally affects the semi-arid areas of the Mediterranean region (Gomes *et al.*, 2003; Lopez *et al.*, 1998; Moreno Brotons *et al.*, 2009) as well as the temperate climate areas of the northern European countries (Bärring *et al.*, 2003; De Ploey, 1986; Eppink and Spaan, 1989; Goossens *et al.*, 2001). In Northwestern Europe, wind erosion mostly occurs on large open fields with sandy soil, in particular under dry conditions when soil cover is low (in spring). Figure 3.1 gives an example of wind erosion in the Netherlands. Wind erosion is also a problem in Iceland due to its volcanic soils, combined with low vegetation cover and strong winds.

The unwise use and management of land, together with intensive crop cultivation, increasing mechanisation, increased field sizes, and removal of hedges, exacerbate the effects of wind erosion in the already most sensitive agricultural areas in Europe (Warren, 2003; Riksen et al, 2003; Funk and Reuter, 2006).

Today, wind erosion is a serious problem in many parts of northern Germany, eastern Netherlands, eastern England and the Iberian Peninsula. Estimates of the extent of wind erosion range from 10 to 42 million ha of Europe's total land area, with around 1 million ha being categorized as severely affected (Lal, 1994; EEA, 2003). Recent work in eastern England reported mean wind erosion rates of 0.1-2.0 ton ha^{-1} yr^{-1} (Chappell and Warren, 2003), though severe events are known to erode much more than 10 ton ha^{-1} yr^{-1} (Böhner *et al.*, 2003). In a similar study, Goossens *et al.* (2001) found values of around 9.5 ton ha^{-1} yr^{-1} for arable fields in Lower Saxony, Germany. Breshears *et al.* (2003) researched the relative importance of soil erosion by wind and water in a Mediterranean ecosystem and found that wind erosion exceeded water erosion in scrubland (around 55 ton ha^{-1} yr^{-1}) and forest (0.62 ton ha^{-1} yr^{-1}) sites but not in grasslands (5.5 ton ha^{-1} yr^{-1}).

However, the current state of the art in erosion research lacks a comprehensive knowledge about where and when wind erosion occurs in Europe, and the intensity of erosion that poses a threat to agricultural productivity. Recent investigations within the framework of EU projects (Wind Erosion on European Light Soils

(WEELS) and Wind Erosion and Loss of Soil Nutrients in Semi-Arid Spain (WELSONS; Warren, 2003) provide reasons to suggest that the areas potentially affected by wind erosion may be more widely spread than previously reported by the European Environment Agency (EEA, 1998). Further studies have shown that the areas previously reported as being only slightly affected by wind erosion (EEA, 1998) are currently undergoing severe erosion processes (Böhner *et al.*, 2003; Gomes *et al.*, 2003). These findings indicate that the European-wide assessment of the distribution and severity of wind erosion provided by the EEA (1998) may no longer be representative. The lack of research, particularly at the landscape to regional scales, prevents national and European institutions from taking actions aimed at an effective mitigating of land degradation.

Figure 3.1: Wind erosion in the Veenkolonien, Netherlands, spring 2013 (photo: Allard Hans Roest and Lidy Roest).

To gain a better understanding of the geographical distribution of wind erosion processes in Europe, in early 2014, the JRC proposed an integrated mapping approach to estimate soil susceptibility to wind erosion (Borrelli *et al.*, 2014a). The wind-erodible fraction of soil (EF) is one of the key parameters for estimating the susceptibility of soil to wind erosion (Fryrear *et al.*, 1994; Fryrear *et al.*, 2000). It was computed for 18,730 geo-referenced topsoil samples (from the Land Use/Land Cover Area frame statistical Survey – LUCAS - dataset). The prediction of the spatial distribution of the EF (Figure 1) and a soil surface crust index drew on a series of related but independent covariates, using a digital soil mapping approach (Cubist-rule-based model to calculate the regression, and Multilevel B-Splines to spatially interpolate the Cubist residuals) (Goovaerts, 1998). The spatial interpolation showed a good performance with an overall R2 of 0.89 (in fitting). Spatial patterns of the soils' susceptibility to wind erosion in line with the state of the art in the literature were archived. Regional control areas (i.e., Lower Saxony and Hungary) (Figure 3.2) showed encouraging results, and indicated that the proposed map can be suitable for national and regional investigations of spatial variability and analyses of soil susceptibility to wind erosion.

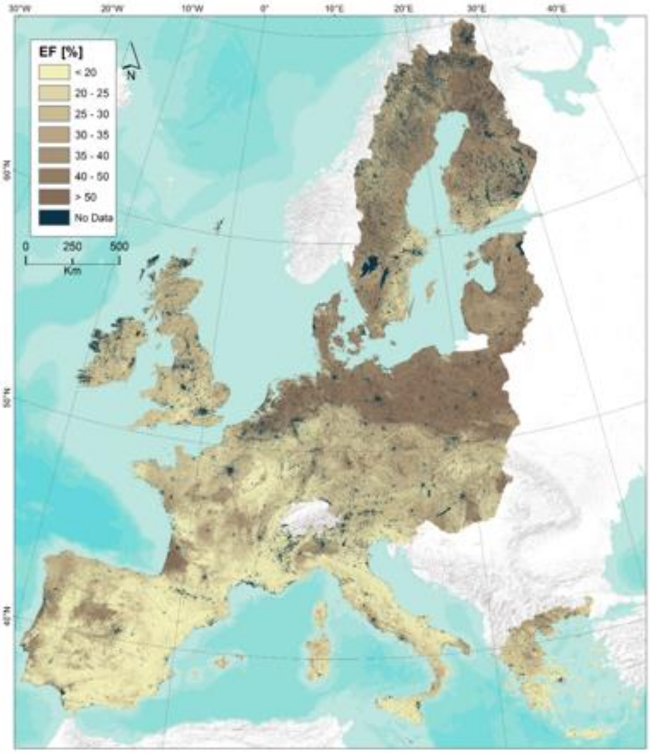

Figure 3.2: Map of wind erosion susceptibility of European soils (500mspatial resolution) based on the estimation of the wind-erodible fraction of soil (EF) (Chepil, 1941; Fryrear et al., 2000). The geographical extent of this study includes 25 member states of the European Union. Bulgaria, Croatia and Romania were not included, as the LUCAS-Topsoil database currently does not include them. Non-erodible surfaces (such as lakes, glaciers, bare rocks and urban areas) were described as No Data.

The resulting erodible fraction (EF) values ranged from 3.6% to 69.0%, with a mean value of 30% (σ 10.6%). According to the erodibility classification proposed by Shiyatyi (1965), which has been adopted for European contexts by López *et al.* (2007), 81.3% (EF b 40%) and 13.8% (EF ≥ 40% and b50%) of the investigated area are characterised by slight and moderate erodibility, respectively, whereas 4.9% are characterised by high erodibility (EF ≥ 50%). As can be inferred from Figure 3.2, the distribution of the spatial wind-erodible fraction patterns suggests a division of the European surface into three regions: i) a north region mostly dominated by the highest EF values, ii) a central eastern region with average EF values interspersed with some high/low spots, and iii) the Mediterranean area, which has mainly low wind-erodible fraction values.

Later, in the second half of 2014 the JRC carried out a preliminary pan-European assessment that delineates the spatial patterns of land susceptibility to wind erosion, and lays the groundwork for future modelling activities. An Index of Land Susceptibility to Wind Erosion (ILSWE) (Borrelli *et al.*, 2014b) was created by combining spatiotemporal variations of the most influential wind erosion factors (i.e. climatic erosivity, soil erodibility, vegetation cover and landscape roughness) (Figure 3.3). The sensitivity of each input factor was ranked according to fuzzy logic techniques. State-of-the-art findings within the literature on soil erodibility and land susceptibility were used to evaluate

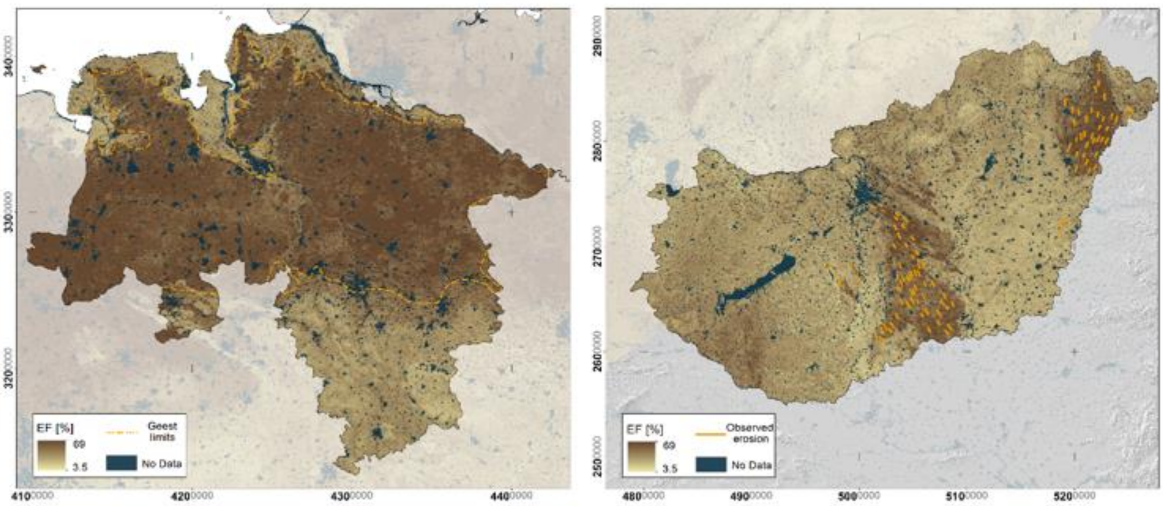

Figure 3.3: Comparison of the predicted wind erosion susceptibility of soil (background raster image) with regional observations (represented with yellowish lines). a) The Geest area in Lower Saxony. This area mainly consists of glacial moraines and sand plains, forming light sandy soils largely endangered by wind erosion (Capelle, 1990; Gross and Schäfer, 2004, among others). b) Area affected by wind erosion in Hungary according to Stefanovits and Várallyay (1992).

the outcomes of the proposed modelling activity. Results show that the approach is suitable for integrating wind erosion information and environmental factors. Within the 34 European countries under investigation, moderate and high levels of land susceptibility to wind erosion were predicted, ranging from 25.8 to 13.0 M ha respectively (corresponding to 5.3 and 2.9 % of total area). New insights into the geography of wind erosion susceptibility in Europe were obtained (Figure 3.4 and Figure 3.5), and provide a solid basis for further investigations into the spatial variability and susceptibility of la nd to wind erosion across Europe.

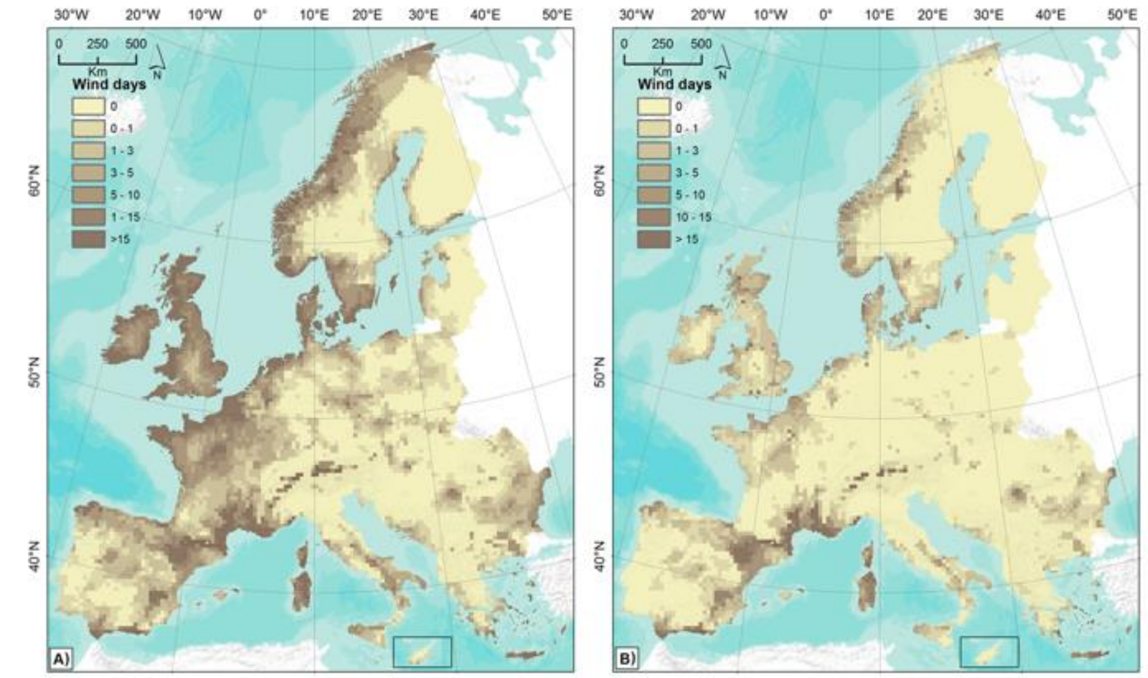

Figure 3.4: Assessment of land susceptibility to wind erosion-workflow.

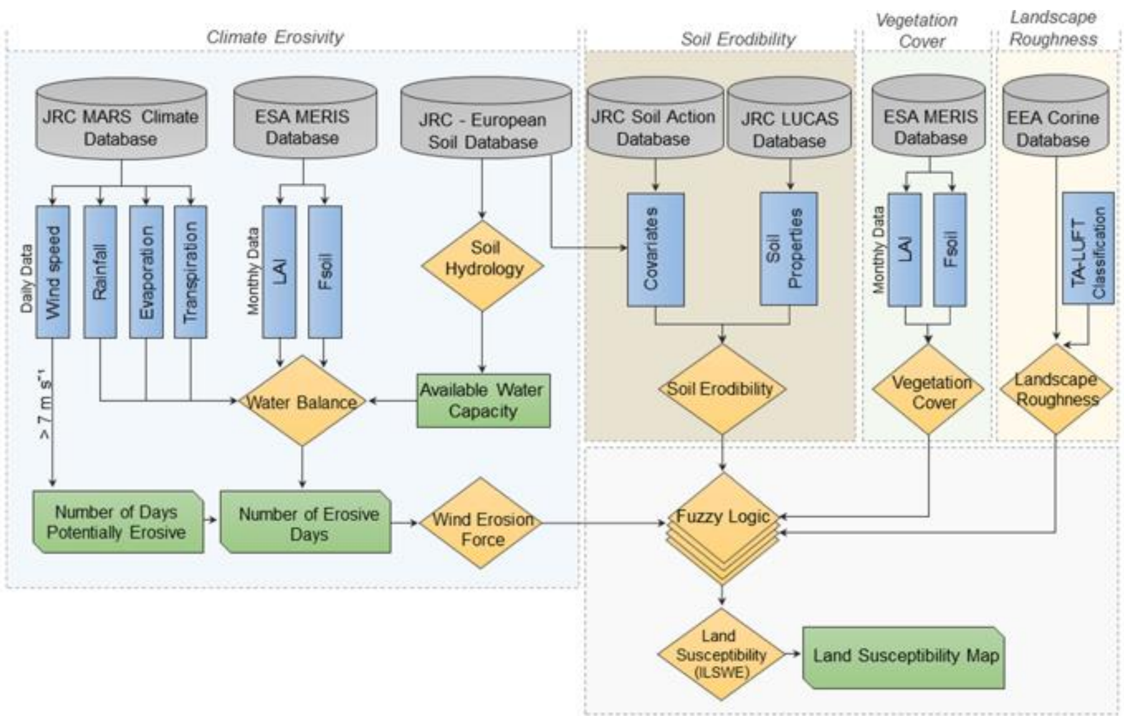

Figure 3.5: Number of erosive days. A) Spatial distribution of potentially erosive days (wind-speed threshold assumed as 7 m s^{-1}); B) spatial distribution of erosive days corrected according the proposed topsoil moisture content.

Figure 3.6 presents the Index of Land Susceptibility to Wind Erosion (ILSWE) for 36 European countries using climate data for the period 1981-2010 and the land cover condition of 2012. The modelling outcomes were ranked into five classes using the *quantile classification* method. Approximately 78.5% of the land surface under investigation showed no usceptibility to wind erosion. The portion of the studied area with low and very low susceptibility accounted for 13.3%, whereas moderate susceptibility was reported for 5.3% (ca. 25.8Mha). For the remaining 2.9% (ca. 13Mha) of the studied area, high land susceptibility to wind erosion was modelled.

The results show that regions susceptible to wind erosion occur in most of the countries observed. Nevertheless, areas potentially affected by high erosion levels appear only in specific regions. In the Mediterranean area, susceptibility is high to moderate along the south-west coast of Spain (in the Spanish communities of Aragón, Castilla-La Mancha and Cataluña), in the Gulf of Lion (i.e. the French metropolitan region of Languedoc-Roussillon and Provence-Alpes-Côte d'Azur) and on the Italian, French and Greek islands. In northern Europe, the most highly susceptible regions are found along the coastal area, i.e. in Nord-Pas-de-Calais and Normandy in France; and parts of Northern Netherlands. In the United Kingdom, some of the most susceptible areas were estimated in south-western England and Scotland. Large parts of Denmark, particularly in the western sector of the peninsula and in the eastern archipelago, also show high susceptibility values. The region of Scania is the area with the highest susceptibility in Sweden. Severe susceptibility was also modelled along the Romanian and Bulgarian coasts and in the lowlands surrounding the Carpathian Mountains. For the more continental areas, the results show high susceptibility in the Pyrenean and Alpine regions, central Spain and north-eastern Serbia. Few hotspots were identified along the coasts of Germany and Poland, in central France and central and southern Italy. The sectors of the study region that tend to have consistently low susceptibility values are the Baltic States, Finland, Slovenia, Portugal, southern Germany and Ireland.

The cross-check results of ILSWE show that the areas that were predicted as susceptible to wind erosion coincide with the reference locations reported in the literature. The overall accuracy was 95.5% with a Kappa Index of Agreement (KIA) of 0.910. Accordingly, 109 (69.7%) of the 156 locations reported in literature were classified as being moderately/highly susceptible while another 13 (8.4%) fell into areas defined as having low susceptibility (more details in Borrelli *et al.*, 2014b). Another 27 (17.4%) of the literature sites fell into areas classified as being very lowly susceptible. Considering that, quantitative measures of wind erosion are

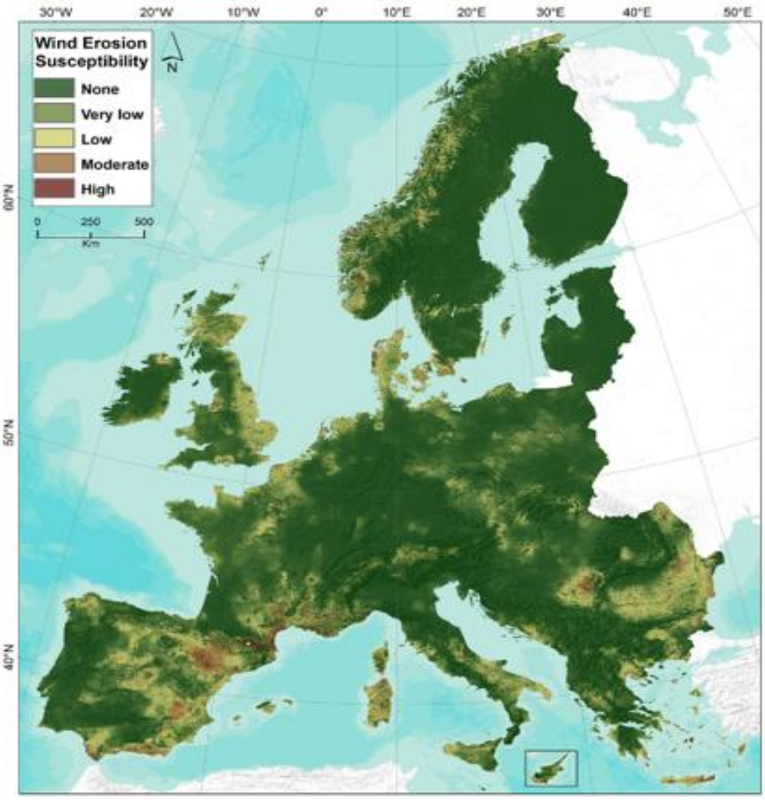

Figure 3.6: Index of Land Susceptibility to Wind Erosion (ILSWE) predicted for 36 European countries.

not available for most parts of Europe, and that the findings in the literature are heterogeneous in the scales and methods and are often limited to qualitative descriptions, the results obtained in this preliminary investigation show a good agreement with local and regional studies. Particularly encouraging results were observed from comparisons with studies carried out in Scania (Bärring *et al.*, 2003), western Denmark (Veihe *et al.*, 2003), north-eastern Germany (Funk and Voelker, 1998), East Anglia (Warren, 2003), Greece (Kosmas *et al.*, 2006), Spain (Gomes *et al.*, 2003) and Austria (Strauss and Klaghofer, 2006).

3.3 Drivers/pressures

It is generally accepted that wind erosion occurs when three conditions are present: the wind is strong enough, the soil surface is susceptible enough, and there is no surface protection from crops, residues or snow cover (Shao and Leslie, 1997). Under these conditions, the magnitude of an erosive event is governed by the eroding capacity of the wind and the inherent potential of the land to be eroded (Fryrear *et al.*, 2000). Various drivers cause changes in the conditions that influence wind erosion.

Climatic change can have a direct impact on wind erosion if it results in stronger or more frequent winds. However, climatic change will also have indirect impacts as it also influences other factors that are relevant for wind erosion. For example, it has an impact on plant cover, soil moisture, snow cover and growing seasons.

Human activities, in particular land use and land management, have a major impact on soil cover and can influence soil properties. Compared to natural vegetation, soil cover in arable land is less constant and most of the year it is lower. Arable fields are especially vulnerable to wind erosion in spring, when the seedbed has been prepared but plant cover is still too low to adequately protect the soil from wind erosion. As indicated earlier, the degree to which arable land is susceptible depends on: field size, level of mechanisation, intensiveness of cultivation, presence or absence of barriers that slow down the wind, such as tree belts. According to Riksen et al (2003), these factors have all contributed to increased wind erosion in Europe since the 1950s. Land management activities also have an impact on soil structure and on various soil properties, such as organic matter content, aggregate stability and cohesion.

Land use and land management are also influenced by socio-economic and political factors. For example, EU policies have an effect on the choices that farmers make regarding land use and crop. Furthermore, some potential measures against wind erosion have been prohibited. For example, in the Veenkolonien in the Netherlands it used to be common practice to apply liquid manure to the soil surface (Hessel et al, 2011) when strong winds were predicted under dry conditions. This measure was useful to decrease wind erosion rates, but was prohibited because of the N emissions associated with the application of the manure. Another example of the influence of this kind of driver is that of economics, which influences the type of measures used against wind erosion perhaps more than the physical effectiveness of such measures. For example, for crops that have low margins, such as potato and sugar beet in the Netherlands, only cheap measures against wind erosion are affordable from an economic point of view. For flower bulbs, the margins are higher and more expensive measures are therefore affordable.

3.4 Key indicators for soil erosionby wind

Soil erosion by wind is a complex geomorphic process governed by a large number of variables (Shao, 2008). Field-scale models such as the Wind Erosion Prediction System (WEPS – Wagner, 1996) employ up to some tens of parameters to predict soil loss. A pan-European assessment of land susceptibility to wind erosion calls for a simplified and more practical approach (Zobeck *et al.*, 2000; Funk and Reuter, 2006). Therefore, a limited number of key parameters, which can express the complex interactions between the variables controlling wind erosion, should be considered

In the context of the ENVASSO project (Huber *et al.*, 2008), a working group identified two key indicators for wind erosion (Table 3.1). The indicators address the main question “What is the current status of wind erosion in Europe? This indicator cannot be used in isolation because of the scarcity of experimental plots. However, it can play an important part in future calibration and validation of estimated soil loss from wind erosion models. Therefore, this indicator of ‘measured soil loss by wind erosion’ should be considered for future model validation and quantification of uncertainties. The resources allocated to measuring wind erosion will need to be increased significantly if this approach is to prove successful.

Table 3.1: Wind erosion indicators.

Indicators	Advantages	Disadvantages
Estimated soil loss by wind erosion (t/ha/yr)	• accurate estimates of soil loss by wind are needed to implement soil protection measures • processes leading to wind erosion are well researched and understood.	• Wind erosion is even more difficult to estimate than water erosion • modelling errors are potentially large • There are far fewer data on wind strength and direction in Europe than there are rainfall data • Wind erosion has been measured at even fewer sites in Europe than water erosion.
Measured soil loss by wind erosion (t/ha/yr)	• Methods for measuring soil loss by wind from field plots are well documented	• Experimental errors are generally larger than for water erosion measurement • There are very few monitoring sites in Member States and they tend to be located only where wind erosion is active

Several factors influence the amount of material that can be eroded, and might thus qualify as indicators. The first is soil resistance, which depends on grain size and on stability of aggregates. As a result of weak bonding between particles, sandy soils are more susceptible to wind erosion than fine textured soils. A second is surface roughness, which usually decreases wind erosion rates. A third factor is climate, which determines wind velocity and moisture content. Only dry soil can be detached by wind, and moisture also has indirect effects due to its effect on plant growth. However, rainfall can also destroy aggregates and smooth the soil, so that it becomes more susceptible to wind erosion. On the other hand, rainfall can seal the soil, resulting in increased resistance to wind erosion (Toy *et al.*, 2002). Further, topography, and especially length of exposed area, both play a role. Finally, vegetation has a large influence on wind erosion rates. Vegetation is an effective protection against wind erosion because it causes zero plane displacement and because vegetation barriers decrease wind velocity. Vegetation can also reduce erodibility of the soil, e.g. through roots and increased organic matter content (Toy *et al.*, 2002).

3.5 Methods to assess the status of soil erosion by wind

Wind erosion can be assessed through measurement/monitoring and through modelling.

3.5.1 Measurement

Wind erosion can be measured in wind tunnels and in the field. The use of wind tunnels permits the control of particular conditions, which makes them suitable for studying certain aspects of wind erosion. However, they cannot capture all factors that are active in the field. In the field, wind erosion can be determined by measuring the depth of soil removed (using e.g. erosion pins), or by determining the transport rates on erosion plots.

Riksen en Goossens (2007) used 50 cm long erosion pins with 5 mm diameter to measure wind erosion in the Kootwijkerzand, which is an area with shifting sands in the Netherlands. Length of the pins was measured every week. Contrary to water erosion plots, which are usually rectangular, wind erosion plots are often circular because wind can come from any direction. Furthermore, the air entering the measurement plot will already carry a certain sediment load (Toy *et al.*, 2002), which is not the case for properly designed water erosion plots. Therefore, multiple measurement equipment is needed to be able to measure sediment load of wind coming from any direction both when it enters and when it leaves the measurement plot. Furthermore, sediment concentrations must be measured at multiple height since there is a vertical distribution of sediment load.

Windblown sediment can be collected using traps, which may be placed horizontally (to measure total creep and saltation load) or vertically (to measure saltation and suspension load as a function of height). Vertical traps include mechanisms to rotate the traps into the wind. Figure 3.7 shows and example of wind erosion measurements in the Netherlands.

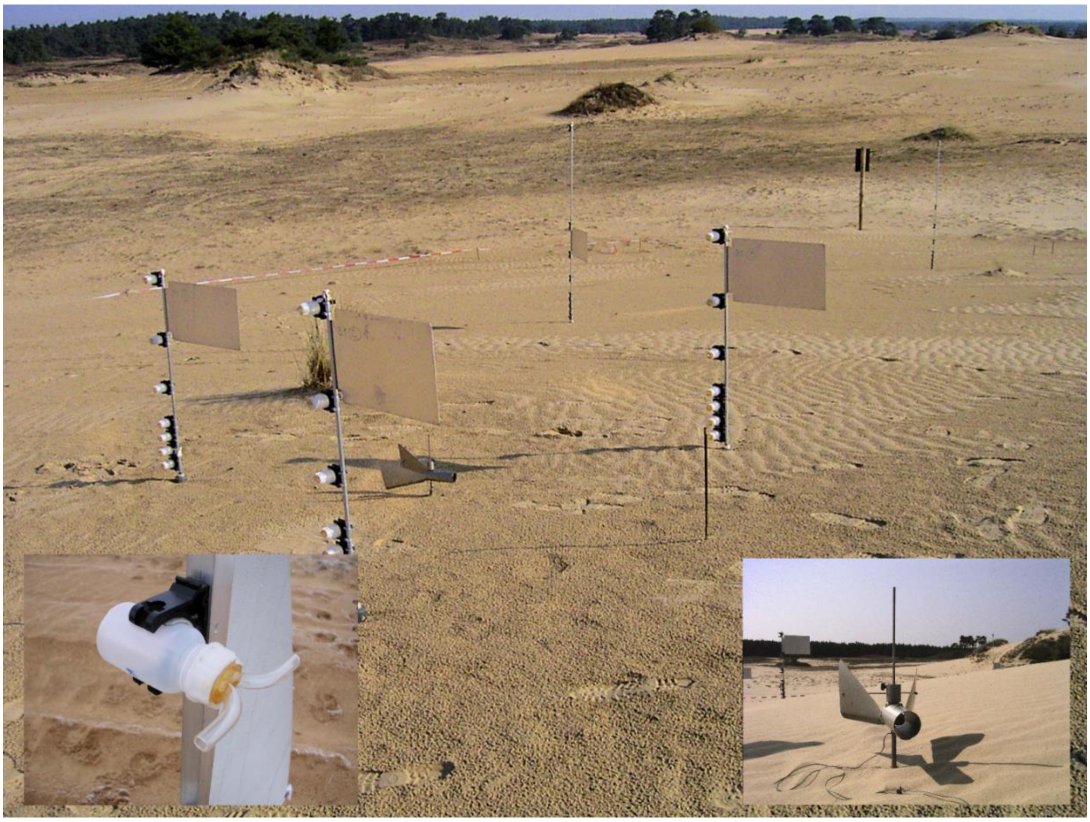

Figure 3.7: Examples of equipment used to measure wind erosion in the Kootwijkerzand, Netherlands (Hessel et al, 2011).

3.5.2 Modelling

Development of wind erosion models has not been as prolific as that of water erosion models. The first model to appear was WEQ (Wind Erosion Equation) (Woodruff & Siddoway, 1965). It has similarities to the USLE in that it too calculates erosion as a function of a number of factors that are supposed to influence erosion. In the case of WEQ those factors are: soil erodibility, soil roughness, climate, field length and vegetation cover. Unlike USLE, erosion cannot be calculated as a multiplication of these factors because several of the factors interact. Van Pelt & Zobeck (2002) found that WEQ generally under predicted wind erosion by about 50% for discrete periods of several months. However, predictions were much improved after calibration for local conditions.

RWEQ (Revised Wind Erosion Equation) is an empirical model that makes annual or period estimates of wind erosion and based on a single event wind erosion model (Zobeck *et al.*, 2001). Like WEQ, it uses a number of factors: wind, erodibility, surface crust, roughness and ground cover. Zobeck *et al.* (2001) validated RWEQ for single events and found that RWEQ has the tendency to underestimate transport capacity and soil loss and to overestimate critical field length. However, significant relationships were found between observed and predicted transport capacity and soil loss, showing the potential of RWEQ. RWEQ is currently being used to determine if residue left on the field is enough to provide sufficient protection, or if additional measures are necessary (Fryrear & Bilbro, 1998).

WEPS (Wind Erosion Prediction System) is more process based than RWEQ and therefore requires additional input parameters. WEPS uses a daily time step and operates on field scale (Wagner & Tatarko, 2001). It consists of several sub models and simulates spatial and temporal variability of soil, crop, residue conditions and soil loss/deposition. It gives separate predictions for saltation/creep, suspension and PM10 (Hagen, 2001; Tatarko & Wagner, 2002). WEPS can also be used for single event (i.e. days) by using the erosion sub model as a standalone model (Tatarko & Wagner, 2002). Hagen (2001) validated the model for 24 storms on a cropland field at Big Spring, Texas, and obtained an R-squared of 0.65 between observed and predicted soil loss. Funk *et al.* (2002) validated the model for 21 events in north-eastern Germany and obtained an R-squared above 0.9. WEPS was developed independently of WEPP, but some commonality between these models is being sought (Fox *et al.*, 2001).

Gregory & Darwish (2001, 2002) developed the TEAM (Texas Tech Erosion Analysis Model) in an attempt to create a wind erosion model that is applicable to all environments in which wind erosion occurs. According to them, the USDA models (WEQ and WEPS) are only suitable for agricultural land and not for deserts and mine tailings. TEAM is a single event process-based model, although the windows version can at present only handle long-term climatic input.

WEELS (Wind Erosion on European Light Soils) predicts the spatial distribution of wind erosion, using a modular structure. There are modules for wind, wind erosivity, soil moisture, soil erodibility, soil roughness and land use. Measures against wind erosion can be simulated using a reduction factors, as in WEQ en RWEQ.

The latest modelling effort on wind erosion in Europe was made by Borrelli et al (2014a, b) and was described earlier in this chapter. It is based on the combination of the most influential parameters, i.e. climate (wind, rainfall, evaporation), soil characteristics (sand, silt, clay, $CaCO_3$, organic matter, water-retention capacity, soil moisture) and land use (land use, percent of vegetation cover, landscape roughness). The spatial and temporal variability of these factors were appropriately defined through Geographic Information System (GIS) analyses. Harmonised datasets and a unified methodology were employed to suit the pan-European scale and avoid generating misleading findings that could result from heterogeneous input data.

Despite this work's significant contribution towards a better understanding of the distribution and the potential threats of soil erosion by wind process, future research studies are encouraged to include further elements in the base model designed for this study. Future modelling approaches could optimise the spatial resolution of the climate data and topsoil moisture module. Considering the significant impact that the soil moisture content has in reducing the number of potentially erosive days, a snow cover factor needs to be incorporated in future modelling exercises. Moreover, further components should be included in the model to allow for the biophysical and land management differences within the heterogeneous environment of Europe. For instance, one could consider aspects such as: (i) vegetation growth modules based on phenological analysis; (ii) a more accurate identification of bare soil conditions (e.g. tillage and sowing preparation); (iii) downscaling of the climate data by integrating local topographic controls; (iv) a description of the agricultural field size and boundary characteristics; (v) post-harvest residue cover management; and (vi) agricultural field irrigation.

3.6 Effects of soil erosion by wind on other soil threats

Wind erosion can affect several other soil threats (Figure 3.8). For example, it can result in a reduction of organic matter content, water holding capacity, chemical soil fertility and biological activity (De Vries en Brouwer, 2006).

There is a clear link with loss of organic matter. Wind erosion removes the upper part of the soil, which in general is also the part that has the highest organic matter content. It can also influence soil structure (Riksen et al 2003), both directly and indirectly due to the positive effect that soil organic matter has on soil structure. Loss of organic matter due to wind erosion problems has been observed not only on mineral soils, but also on peat soils, such as in the Broddho area in Sweden.

There is also a link with contamination, as wind erosion can also transport fertilisers, herbicides, and pesticides (Van Kerckhoven *et al.*, 2009), 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. Based on data from Chardon and Van der Hoek (2002) they estimate that wind erosion provides 7-15% of all fine dust in the Netherlands on a yearly basis.

Another soil threat that is influenced by wind erosion is desertification. Semi-arid and arid areas are intrinsically susceptible to wind erosion, as plant cover tends to be low, while winds can be very strong. This can result in loss of productivity, or even complete loss of the topsoil. This problem can occur in the Mediterranean area, but also in Iceland. Wind erosion can also affect biological activity in the soil, as this activity is usually concentrated in the upper part of the soil.

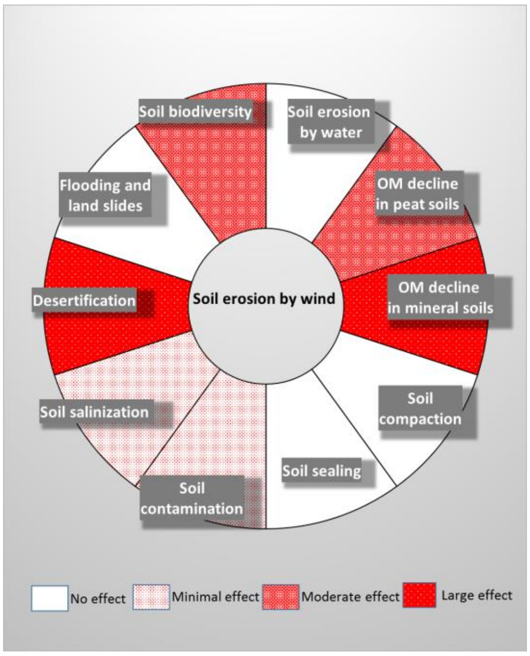

Figure 3.8: Effects of soil erosion by wind on other soil threats. Red is negative effect.

3.7 Effects of soil erosion by wind on soil functions

Wind erosion can affect soil functions both on-site and off-site. On site, the topsoil is lost, and therefore, the part of the soil that is most fertile resulting in a loss of productivity. Furthermore the erosion process itself can cause damage to crops due to sand blasting. Wind erosion may also affect other soil functions due to its effect on soil structure, for example, water holding capacity can be reduced. As a result of the selective transport of the finer particles, wind erosion may also result in a coarsening of the remaining soil. Off-site, sand transported by the wind can damage machines, buildings and crops, while sand deposits may bury fields and waterways. Burial of fields obviously has a major impact on the functions the soil on the site can perform, including its production function.

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4 DECLINE IN SOIL ORGANIC MATTER IN PEATSOILS

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4.1 Description of decline in SOM in peatsoils

Decline of organic matter in peat soils directly threatens one of the main ecosystem services of peat soils: the storage of carbon. Peat soils cover more than 420 million ha worldwide, equivalent to 3% of the Earth's land surface (Strack, 2008) and contain 20-30% of the world's soil organic carbon (Moore, 2002). Joosten (2009) estimated the total C stock in peat soils in the world to be 445 700 Mton C. This makes peat soils one of the major stocks of C in the world, even more than in the atmosphere. Byrne *et al.* (2004) reported a total area of peat soils of 34 million hectares in the EU Member States and Candidate Countries with an estimated total C store in peat of 17 Pg (17 000 Mton) or about 20-25% of the carbon in soils of the EU. Of the 34 million hectares about 5.80 million ha is drained of which 3.60 million ha is in agricultural use as cropland (0.95 million ha) or grassland (2.65 million ha) (Schils *et al.*, 2008).

The complete FAO definition of peat soil (histosol) or organic soil is rather complex (FAO, 1998, 2006/7). It not only refers to the thickness of soil layers and their organic content but also to their origin, underlying material, clay content and water saturation. Essentially, apart from shallow (≥10 cm) organic rich soils overlying ice or rock, organic soils (histosols) are identical to peat and peaty soils of at least 40 cm total thickness within the upper 100 cm, containing at least 12% organic carbon (~20 % organic material) by weight (Couwenberg, 2009). This definition differs from several European definitions of peat, which have definitions with layers of 30 cm or even 20 cm and slightly higher minimum organic matter contents (Joosten andClarke 2002). This means that in the national inventory of these countries the area of peat soils generally is overestimated compared with an inventory based on the FAO definition of peat soil.

Mineralization or oxidation of peat soils is a main cause of reduction of organic matter stocks and degradation in northern Europe. Several factors are responsible for a decline in soil organic matter and many of them relate to human activity: drainage, cultivation and conversion to arable land, liming, fertilizer use /nitrogen (Kechavarzi *et al.*, 2010) causing rapid mineralization of organic matter. Also, tillage, crop rotations with reduced proportion of grasses, soil erosion, wild fires and climate change causing warmer climatic conditions and periods with drought, increase mineralization and so the decline of organic matter stored in peat soils.

Peat soils in agricultural use are the most affected because, due to the related drainage, agricultural peat soils mineralize and the decline of organic matter (OM) is about 10-20 tonnes OM per hectare per year. This causes large emissions of CO_2 which amount to 20-40 tonnes of CO_2 per hectare per year (Oleszczuk *et al.*, 2008; Van den Akker *et al.*, 2008). It should be noted that considering peat soils means that we in fact exclude organic soils with a layer thickness < 40 cm. These organic soils cover large areas and are as vulnerable to degradation as peat soils, they can lose large quantities of OM and have also large CO_2 emissions. The same accounts for gyttja soils which have a high OM-content and can have CO_2 emissions of the same order as peat soils.

Over the last centuries, the pressure for land in Europe has resulted in the reclamation of large areas of peat land to make them suitable for agriculture or other land uses. As a result, natural or semi-natural peat lands have become rare in countries like the Netherlands, Germany and Poland, where 70 – 85 % of the peat land area is in agricultural use. The required drainage to reclaim peat lands results in subsidence and degradation of peat soils by shrinkage and biological degradation (oxidation). The rate of subsidence is very variable and depends on a number of factors, such as the type of peat, rate of decomposition, density and thickness of peat layers, drainage depth, climate, land use, wind erosion and history/duration of its development. Typical peat subsidence rates in Europe range from a few millimeters to as much as 3 centimeters per year depending on drainage and climatic (a.o. temperature) conditions (Kasimir-Klemedtsson *et al.*, 1997).

The subsidence can be subdivided into its three major component processes: (1) consolidation and compaction; (2) loss of organic matter due to biochemical decomposition (oxidation) and (3) shrinkage by drying. Oxidation is the main factor responsible for subsidence over the long term. Usually drainage levels will be adapted to the lowered surface from time to time, so in that way the oxidation and subsidence process can continue until the whole peat layer is oxidized and has disappeared. This loss of organic matter (OM) can also be expressed in loss of carbon C or loss of carbon dioxide CO_2. Multiplication factors to convert C into respectively OM or CO_2 are:

$$C = 3.67 \cdot CO_2 \approx 1.82 \cdot O \qquad [4.1]$$

The conversion of C to CO_2 is based on atomic weight, the conversion of C to OM is based on Schothorst (1977). Subsidence damages infrastructure and buildings and water management becomes more complex and expensive. Many wetlands are difficult to preserve as "wetland" because subsidence of adjacent agriculturally improved (i.e. drained) land, results in “islands of peat” surrounded by lower elevation agricultural lands. The net effect is a constant drainage of the peat lands in nature reserves towards the surrounding agricultural area with a much deeper drainage level. Due to the ongoing subsidence drained peat lands become lower than river water levels and sea levels increasing the flooding risk. This situation is worsened by climate change and sea level rise. Peat soils in arable agriculture are also vulnerable to wind erosion, which can cause losses of 3 – 30 t ha^{-1} y^{-1} peat also causing air pollution (fine organic particles) and deposition of peat on nearby fields and water courses (Parent *et al.*, 1982; Kohake *et al.*, 2010).

The degradation of peat soils also causes off-land problems. Degradation products, such as nutrients, peat particles and dissolved organic matter (DOM) can be a source of water contamination. The oxidation of peat is an important source of CO_2 and part of the mineralized nitrogen will be converted into N_2O, which is a strong Green House Gas. Schils *et al.* (2008) concluded that the largest emissions of CO_2 from soils are resulting from land use change and especially drainage of organic soils. They also concluded that the most effective option to manage soil carbon in order to mitigate climate change is to preserve existing stocks in soils, and especially the large stocks in peat and other soils with a high content of organic matter. This conclusion is one of the reasons to pay special attention to the decline of organic matter stored in peat soils in the RECARE project.

4.2 State of decline in SOM in peatsoils

According to Schils *et al.* (2008), the current area of peat in the EU Member States and Candidate Countries is over 318 000 km^2. More than 50% of this area is in just a few northern European countries (Norway, Finland, Sweden, United Kingdom); the remainder mainly in Ireland, The Netherlands, Germany, Poland and Baltic states. Of that area, approximately 50% has already been drained, while most of the undrained areas are in Finland and Sweden. Based on figures of Joosten (2009) we calculated that the EU(27) had in 2008 about 229 000 km^2 peat soils with a conservative estimated C stock of 18 700 Mton of C. The estimated CO_2 emissions are:

- agriculture – 100.5 Mton CO_2 per year;
- forestry – 67.6 Mton CO_2 per year;
- peat extraction – 5.6 Mton CO_2 per year (on-site).

In total this makes an emission of drained peat soils of the EU(27) of 173 Mton CO_2 per year, which means that the European Union is, after Indonesia and before the Russian Federation, the world's second largest peat land emission hotspot (Joosten, 2009 and 2012). It should be noted that the CO_2 emissions by ‘peat extraction’ are only the on-site emissions of the peat land and stockpiles and <u>not</u> the CO_2 emissions of the extracted peat used for horticulture or (the major part) combustion for energy. By dividing the figures for CO_2-emissions by 3.67 the decline in C in peat soils can be derived (see equation 2). By dividing the figures for CO_2-emissions by 1.82 the decline in OM (organic matter) in peat soils can be estimated. The estimated CO_2 emissions worldwide are (Joosten, 2009):

- agriculture – 1086 Mton CO_2 per year;
- forestry – 129 Mton CO_2 per year;
- peat extraction – 21 Mton CO_2 per year (on-site);
- peat fires – at least 400 Mton CO_2 per year.

The emission from cultivated and drained peat soils in EU(27) in agricultural use is approximately 91 Mton CO_2 per year and including the associated emission of N_2O approximately 100 Mt CO_2–eq per year (see Table 1). The amount of CO_2 emission in this table presented by Schils *et al.* (2008) is more or less in agreement with the earlier calculated 100.5 Mton based on figures of Joosten, (2009). As presented in Table 1 the loss of C by oxidation of agricultural peat soils is about 25 Mton per year. This loss of C is very frustrating considering the general expectation that sequestration of C in (mineral) soils can be an important sink of C. Schils *et al.* (2008) conclude that at this moment soils in Europe are most likely a sink and the best estimate is that they sequester up to 100 Mton C per year. This includes the loss of 25 Mton per year of oxidizing agricultural peat soils. Furthermore it should be noted that sequestration of C in mineral soils is limited and potential reversible, while the stock of C in (agricultural) peat soils is at least 3200 Mton. This probably means that in the EU(27) the potential loss of C stock in peat soils in agricultural use is larger than the potential sequestration in mineral soils. Schils *et al.* (2008) state that even though effective in reducing or slowing the build up of CO_2 in the atmosphere, soil carbon sequestration is surely no 'golden bullet' alone to fight climate change due to the limited magnitude of its effect and its potential reversibility.

A consequence of ongoing oxidation of peat soils is that in time peat layers can be completely converted into CO_2 and peat soils converted into (organic rich) mineral soils. Peat layers have been lost by oxidation during land use, but the estimate derivable from the published data, ca. 18 000 km^2, is probably underestimated (Schils *et al.*,2008).

Although peat soils in agricultural use have - per hectare and in total - the highest emissions of CO_2, the area of forested peat soils in the EU is much larger (Joosten, 2009). This area is almost completely concentrated in Finland (60 000 km^2) and Sweden (30 000 km^2) and causes annual CO_2 emissions of 41.6 Mton (Finland) and 7 Mton (Sweden).

Based on findings of Barthelmes *et al.*, (2009) and Couwenberg (2009) we did not use the figures from the IPCC (2006) national greenhouse gas inventories for emissions from LULUCF (Land Use, Land Use Change and Forestry). These yearly inventories include GHG emissions from organic soils in agricultural use and all EU countries take part in these inventories. However, analyses of Barthelmes *et al.* (2009) and Couwenberg (2009) show that these national inventories can be very confusing. For instance agricultural soil emissions are spread over various chapters; definitions for peat soils differ per country; methods to determine the CO_2 emissions differ per country and Emission Factors (EF) can differ by about a factor 4; climatic temperature regimes are not chosen according the climatic map of the IPCC (2006) guidelines; land use categories are mixed up or neglected; old IPCC guidelines are used; etcetera. Furthermore the FAO key for peat soils (FAO 1998, 2006/7)) is misrepresented by IPCC (2006) by failing to include the 40 cm criterion, and the default C-CO_2 emission factors (EF) of the IPCC (2006) guidelines prove to be too low in most cases (see Table 2). In fact, the IPCC defines organic soils as soils having an organic layer more than 30 cm thick (Couwenberg, 2011). These problems with the IPPC (2006) were the reason the inventory of Schils at al. (2008) was used, see Table 1. It should be noted that the IPCC default values for GHG emissions have been updated recently (Wirth and Zhang, 2013, IPCC (2013)). The old and new values differ considerably (see Table 2) and the values of 2013 are much more realistic.

Table 4.1: Emissions of GHG of peat soils in agricultural use. Calculation are based on: grassland emissions 20 tonne CO_2 ha^{-1} a^{-1}; cropland emissions 40 tonne CO_2 ha^{-1} a^{-1} (see Oleszczuk et al., 2008); C/N ratio = 20 (assuming that the major part of agricultural peat soils are fen peats); 1.25 % of mineralized N converted into N_2O (Mosier et al., 1998). Crop area and grassland area are based on Byrne et al. (2004). (Table 7 in Schils et al., 2008).

Country	Agricultural Area km^2	Crop area km^2	Grass area km^2	CO_2 - C Mt / a	CO_2 Mt / a	N_2O CO_2 eq Mt / a	Total CO_2 eq Mt / a
Member states of the EU							
Belgium	252	25	227	0.15	0.55	0.05	0.60
Denmark	184	0	184	0.10	0.37	0.03	0.40
Estonia	840	0	840	0.46	1.68	0.14	1.82
Finland	2930	0	2930	1.60	5.86	0.49	6.35
Germany	14133	4947	9186	10.41	38.16	3.18	41.33
Ireland	2136 [a]	896	1240	1.65	6.06	0.50	6.57
Italy	90	90	0	0.10	0.36	0.03	0.39
Latvia	1000 [a]	1000	0	1.09	4.00	0.33	4.33
Lithuania	1900 [b]	1357	543	1.78	6.51	0.54	7.06
Netherlands	2050 [c]	75	1975	1.16	4.25	0.35	4.60
Poland	7600	55	7545	4.18	15.31	1.27	16.58
Sweden	2500 [d]	630	1870	1.71	6.26	0.52	6.78
UK	392	392	0	0.43	1.57	0.13	1.70
Total EU	36007	9467	26540	24.80	90.95	7.57	98.51
Other European countries							
Iceland	1300 [a]	0	1300	0.71	2.60	0.22	2.82
Norway	6100 [a]	4200	1900	5.62	20.60	1.71	22.31
Russia (Europe)	26400 [a]	2640	23760	15.84	58.08	4.83	62.91
Belarus	9630 [a]	963	8667	5.78	21.19	1.76	22.95
Ukraine	5000 [a]	5000	0	5.45	20.00	1.66	21.66

[a] based on Byrne et al. (2004); [b] based on Oleszczuk et al. (2008); [c] based on Kuikman et al. (2005); [d] based on Berglund and Berglund (2009).

Table 4.2: Default C and CO_2 emission factors (EF) of organic soils according IPCC (2006) and IPCC (2013) guidelines for national greenhouse gas inventories for emissions from LULUCF (Land Use, Land Use Change and Forestry). Note: Virtually all European peat lands are situated in the climatic temperature regimes "Boreal/Cold Temperate". The IPCC (2006) default values prove to be much lower than general accepted values (Couwenberg, 2009) (grassland 20 ton CO_2 ha^{-1} yr^{-1}; cropland 40 ton CO_2 ha^{-1} yr^{-1}).The IPCC (2013) are in better agreement, however, are still lower for cropland.

Climatic temp. regime	**Grassland**		**Cropland**	
IPCC (2006)	(ton C ha^{-1} yr^{-1})	(ton CO_2 ha^{-1} yr^{-1})	(ton C ha^{-1} yr^{-1})	(ton CO_2 ha^{-1} yr^{-1})
Boreal/Cold Temperate	0.25	0.9	5	18.3
Warm Temperate	2.5	9.2	10	36.7
Tropical/Sub-Tropical	5	18.3	20	73.3
IPCC (2013)				
Boreal	5.7	20.9	7.9	29.0
Temperate	6.1	22.4	7.9	29.0
Temperate, grassland nutrient poor	5.3	19.4		

4.3 Drivers/pressures

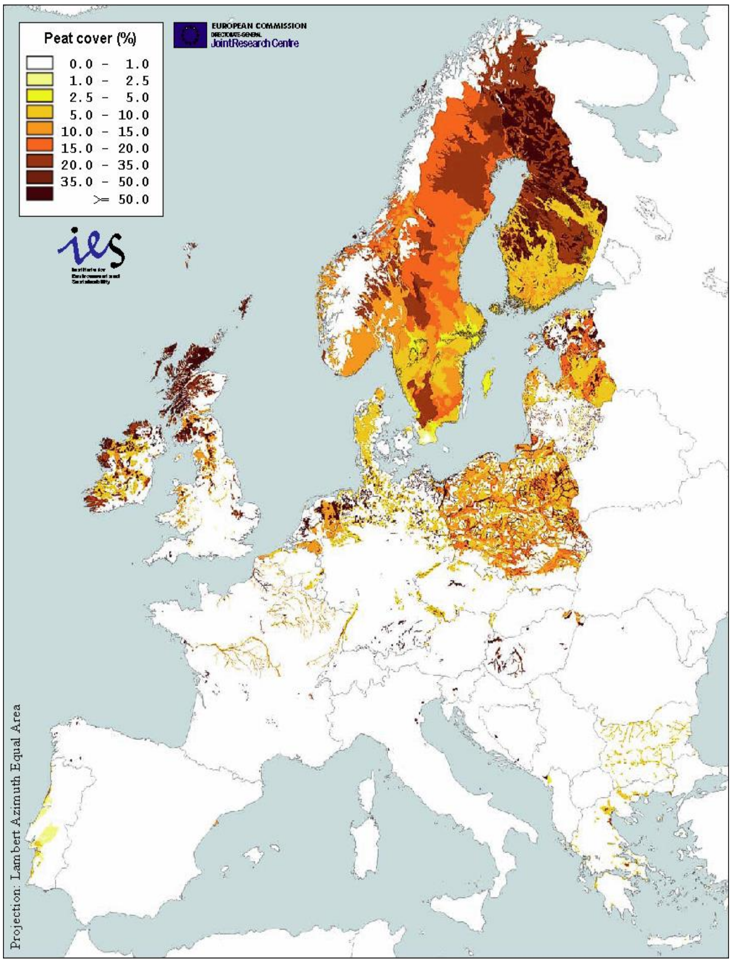

Figure 4.1: Relative cover (%) of peat and peat-topped (0 – 30 cm) soils in the SMUs of the European Soil Database (Montanarella et al., 2006). (SMUs: polygons, known as Soil Mapping Units comprising one or more Soil Typological Units (STUs) e.g. Histosol).

The major socio-economic driver for the reclamation and drainage of peat soils is the need for agricultural land and food production. Fen peat soils are eutrofic or mesotrofic and therefore very suited for agricultural use, however, they have by nature a high drainage basis, so are difficult to drain. Human activities with drainage and cultivation are important drivers for processes leading to decline in organic matter. The human activities are highly influenced/driven by the socio-economic conditions supporting cultivation.

Bog peat soils have in general a much deeper drainage level and by digging deep ditches from the rivers into the bog a very effective drainage system was created. In time the possibilities to lower the drainage basis by pumping improved, first by windmills and later on with steam engines and nowadays mainly electricity driven pumps. Also the improved possibilities to dig long canals for the water regulation and transport made it possible to drain large untouched peat lands. In countries with a limited wood or coal production, like Ireland, The Netherlands and north of Flandres large peat soil areas were also drained for the production of peat as fuel and it is said that the availability of cheap fuel was one of the sources for the Golden Age in The Netherlands. Also large areas of peat soils in the coastal areas were exploited and burned for the extraction of salt.

Nevertheless by far the largest areas of peat soils were drained to make it suitable for agricultural land. In the 20th century with World War I and II with severe food supply problems a great urge for food security was felt and this was a driver to drain very large areas of peat soils for agricultural use in Europe including the Nordic countries and the former USSR. Most drained peat soils were used as arable land or for horticulture for the production of grains, potatoes, carrots etc. However, due to the subsidence by oxidation the surface lowered ever more to the regional drainage basis and many of these drained peat soils became too wet for arable agriculture and horticulture use. Most of the peat soils then were shifted to grassland and nowadays most peat soils in agricultural use are permanent pasture (see Table 4.1). Depending on the hydrological situation and market opportunities for the agricultural products it could be profitable to lower the regional drainage basis below the natural drainage basis by pumping. Examples are the Fens in East Anglia, England, UK, with its very profitable horticulture and the Netherlands where the production of butter and cheese was profitable enough to cover the costs of pumping and construction and maintenance of dikes along the rivers and lakes. In these areas horticulture and dairy farming is still profitable due to the good infrastructure and marketing opportunities, which is for example reflected in The Netherlands in the market value of peat grassland, which is about € 50 000 per hectare. Large areas of agricultural peat soils in less favored areas with limited agricultural infrastructure and marketing possibilities are now abandoned, especially in the former Sovjet Union where large areas of peat soils in agricultural use were abandoned after 1990. It should be noted that these abandoned agricultural peat soils do not return to their natural state, because the drainage status is permanently changed and the upper peat layer is partly humified and enriched with nutrients and the new vegetation cover differs from the original natural vegetation. These abandoned drained peat areas are also prone to fire, and in 2010 large areas in Russia were burned causing a boost of CO_2 emission and air pollution. Abandoned peat lands require controlled rewetting to diminish further degradation and emissions of GHG and erosion and water pollution and risk of fire (Joosten *et al.*, 2012; Maljanen *et al.*, 2013).

Main drivers for improvement of peat land management and pressure for sustainable use are the costs and risks of subsidence in river delta areas (see 1. Description of threat) and the CO_2 emissions and rapid decline of C in one of the most important and vulnerable C stocks on earth (Joosten *et al.*, 2012). Gobin *et al.*, (2011) estimated that 13-36% of the current soil carbon stock in European peat lands might be lost by the end of this century. Moreover climate change will have a major impact on peat soil degradation and increase of CO_2 emissions, partly due to the increase of decomposition rate by the temperature rise, and mainly by the increased occurrence of long periods with extreme drought.

Concrete drivers that really put pressure on national and local policy and landowners and land- and water managers in the EU are a series of international treaties and framework directives of the EU and national laws and framework directives on water management, water quality and soil protection. In the EU especially EU wide signed international treaties and EU Framework Directives are potent drivers for an effective improvement of peat land management. A summary is presented below of some international treaties and EU Framework Directives:

EU Water Framework Directive and the Nitrate Directive: degrading peat soils in agricultural use are in many cases sources for nutrients, DOC and peat particles causing eutrophication and pollution of water bodies protected by the Water Framework and Nitrate Directive. To fulfill the standards of the Water Framework and Nitrate Directive a decline in degradation of peat soils is inevitable.

EU Habitats Directive and Birds Directive: Most peat lands in agricultural use are grassland and large areas of these peat lands are an essential habitat for some meadow bird species for breeding. Extensive use in at least spring is required. Degradation and abandonment results in growing of trees, bushes etc which makes the peat land unsuitable as habitat for these meadow birds.

The Kyoto Protocol: At COP17 in 2011 in Durban, Republic of South Africa, peat lands and organic soils were recognized by the international climate change regime as an accountable factor and a potential target for mitigation action. From 2013 onwards, coinciding with the second commitment period of the Kyoto Protocol, Annex I Parties (a.o. all EU countries) to the UNFCCC (United Nations Framework Convention on Climate Change) were given the opportunity to account for GHG emissions by sources and removals by sinks resulting from “Wetland Drainage and Rewetting” (WDR) under Article 3.4 of the Kyoto Protocol. <u>This means that Annex I countries can use peat land rewetting to meet their emissions reduction targets.</u> COP 17 in Durban furthermore decided that, in contrast to the first commitment period (2008–2012) “forest management” will be mandatory for accounting in the second commitment period (2013–2017). This means that drainage and rewetting of peat lands used for forestry in Annex I Parties must now be accounted for under the Kyoto Protocol. For the time being accounting of grazing land management and cropland management remains voluntary. However, regarding the scope of LULUCF (Land Use, Land-Use Change and Forestry) accounting, the EU has gone further than what was agreed in Durban without compromising the principles and rules laid down internationally. In addition it is also mandatory for Member States to account for Cropland Management and Grazing-land Management, however, accounting for drainage and rewetting of wetlands remains voluntary, as in the international context (http://europa.eu/rapid/press-release_MEMO-12-176_en.htm, the proposal entered into force in July 2013). Although LULUCF emissions and removals are now included in EU's international commitments they are for the time being excluded from the EU targets. The LULUCF emissions and removals will only be included once the accounting rules have been validated. See also the next paragraph on the EU ETS and EU ESD.

EU Emissions Trading System (EU ETS) and the Effort Sharing Decision (ESD): The EU has a clear framework to steer its energy and climate policies up to 2020. This framework integrates different policy objectives such as reducing greenhouse gas (GHG) emissions. The 20% GHG reduction target for 2020 compared to 1990 is implemented through the EU Emissions Trading System (EU ETS) and the Effort Sharing Decision which defines reduction targets for the non-ETS sectors (a.o. agriculture), and its achievement is supported through EU and national policies to reduce emissions. The aggregate target of the Effort Sharing Decision (ESD) is a 10% emission reduction at EU level in 2020 compared to 2005. National targets are distributed between Member States according to their economic capacity. Some need to reduce emissions compared to 2005 whilst others are permitted a limited growth in emissions. The ESD target of The Netherlands, Germany, UK, Ireland and the Nordic countries is therefore at least a 16% reduction. In the next framework for EU levels in 2030 GHG emissions are reduced by 40% in the EU to be on track to reach a GHG reduction of between 80-95% by 2050, consistent with the internationally agreed target to limit atmospheric warming to below 2°C (COM(2014) 15 final). The aggregate target of the Effort Sharing Decision (ESD) will be a 30% emission reduction in 2030 compared to 2005. To ensure that all sectors contribute in a cost-effective way to the mitigation efforts, agriculture, landuse, land-use change and forestry (LULUCF) should be included in the GHG reduction target for 2030.

EU Common Agricultural Policy (CAP): In the CAP post-2013 the issue of GHG emissions is addressed as: "Although GHG emissions from agriculture in the EU have decreased by 20% since 1990, further efforts are possible and will be required to meet the ambitious EU energy and climate agenda. It is important to further unlock the agricultural sector's potential to mitigate, adapt and make a positive contribution through GHG emission reduction, production efficiency measures including improvements in energy efficiency, biomass and renewable energy production, carbon sequestration and protection of carbon in soils based on innovation. CAP measures that can include protection of peat soils in agricultural use are:

- Cross Compliance
- The new green payment in Pillar 1
- Rural development measures in Pillar 2

Under the previous CAP rules, protection of wetland and carbon-rich soils was included as a cross-compliance standard as a "good agricultural and environmental condition", (GAEC 7). In the new CAP rules this GAEC standard has been moved into the basic text and it is now part of the permanent grassland eligibility condition for the green payment in Pillar 1. This has implications for the penalties that farmers face if they decide to ignore the restriction (cross-compliance penalties as a GAEC standard, the loss of the green payment as a green payment requirement). In addition, the wording has become more specific. Member states now have an obligation and an option. The obligation is to designate permanent grasslands which are environmentally sensitive in areas covered by the Habitats or Birds Directives, including peat and wetlands situated in these areas, and which need strict protection in order to meet the objectives of those Directives. The option is, in addition, to decide to designate further sensitive areas situated outside areas covered by these Directives, including permanent grasslands on carbon-rich soils. Farmers are not allowed to convert or plough permanent grassland situated in these areas designated by member states. If member states really take the opportunity to designate all potential carbon 'hot-spots', this could potentially be an effective measure in limiting future carbon emissions from soils. The new CAP includes the GAEC 6 of the previous 2009 CAP, which requires the maintenance of soil organic matter through appropriate practices. Member states have the flexibility to interpret how to implement these standards. This flexibility is, in principle, desirable to account for the heterogeneity of agricultural conditions across Europe, however, made in practice these standards up to now have not been very effective. Altogether it can be concluded that each EU Member State can decide for itself the extent to which it uses CAP 2013 to implement measures to increase or protect

OM in soils. Nevertheless, CAP 2013 offers in principal good possibilities to promote national measures required to fulfill national, EU and international commitments made (Van Zeijts *et al.*, 2011).

EU Soil Thematic Strategy and Soil Framework Directive: The EU Soil Thematic Strategy was adopted by the Commission (COM(2006) 231) on 22 September 2006 and the Commission put forward a proposal for a Soil Framework Directive in 2006 which would have required landowners to take responsibility for soil degradation. It would have obliged member states to ensure that any land user whose actions affect the soil in a way that can reasonably be expected to hamper significantly the soil functions set out in the Directive, including acting as a carbon pool, is obliged to take precautions to prevent or minimize such adverse effects. However, the proposal was prevented from advancing further by a blocking minority in the Council, including Britain, France, Germany, Austria and the Netherlands. After several new submission attempts the Commission in May 2014 decided to withdraw the proposal for a Soil Framework Directive. In this way, the efforts to introduce a very effective instrument to protect soils in general and thus also peat soils were frustrated.

EU Seventh Environment Action Programme: The 7th EAP entered into force on 17 January 2014, and might compensate in some way for the blocking of the Soil Framework Directive. The 7th EAP recognises that soil degradation is a serious challenge. It provides that by 2020 land is managed sustainably in the Union, soil is adequately protected and the remediation of contaminated sites is well underway and commits the EU and its Member States to increasing efforts to reduce soil erosion, increase soil organic matter and to remediate contaminated sites.

4.4 Key indicators on decline in SOM in peatsoils

In the ENVASSO project **Peat stocks** are one of the Key Indicators for the decline of organic matter (Huber *et al.*, 2007). Peat stocks is a crucial indicator, and considered to be one of the three Key Indicators for Decline of SOM, because peat soils are much richer in organic matter than mineral soils and, therefore, can be considered 'hot-spots' where decline in SOM content should be monitored. The other two are: (1). Topsoil organic carbon content (%) and (2) Soil organic carbon stocks (t/ha). These indicators are not suitable for peat soils, because 'decline of SOM' of peat soils results in subsidence of the soil surface, so it is not clear what should be considered as topsoil. Peat stocks as an indicator is easy to interpret by policy makers, but requires monitoring and repeated calculations. There is a consensus that peat soils should be protected, or even that formerly drained wetlands should be re-established. A baseline value could be the present status of peat stocks in Europe (Montanarella *et al.*, 2006). One approach could be to set threshold values so that no further decrease should occur in the mass of peat.

In the ENVASSO project Jones and Verheijen (2008) propose to calculate peat stocks from:

$$P_{Stock} = P_{Area} \cdot P_{Depth} \cdot 10^{-4} \cdot D_b \qquad [4.2]$$

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} (Mg m^{-3})

Huber *et al.* (2008) listed some disadvantages of peat stocks as a Key Indicator that are difficult to overcome:

- To calculate peat stocks accurately, measures of variations in soil depth and bulk density are needed.
- Determining bulk density in peat (organic) soils is notoriously difficult because it is not easy to take undisturbed samples.
- Many peat soils are very deep, and measuring their thickness is rarely practicable.

4.5 Methods to assess status of the decline in organic matter in peatsoils

4.5.1 Determination of OM decline and CO2 emissions from change in peat stock in time

Kluge *et al.*, (2008) used this method to determine peat stocks of a peat land area in 1963 and 2003 and calculated the mean annual decline of C as 0.69 kg/m^2 per year. The same was done by Dawson *et al.*, (2010) resulting in a mean annual C decline of 0.58 kg/m^2 per year. The method is very robust and the resulting figures for emissions are considered to be very reliable and represent a mean value of annual C emissions over a historic period of at least 10 years and in many cases over several decades. However, the method requires detailed historic measurements concerning dry bulk densities and organic matter contents of the soil profile. Furthermore, ideally the land use and hydrological situation (drainage base) should be the same all over the historic period considered. In general these requirements are not met and C emissions are directly measured or determined from the peat stock change over a certain period. Below we present a series of methods to determine peat stock changes:

4.5.2 Direct CO_2 measurements

One method to determine the decline of OM or C stock of peat soils is the direct measurement of CO_2. These measurements are in most experiments performed with closed chamber methods. The advantage is that it is a direct measurement: you measure what you want to know. Disadvantages are that (1) this measurement includes not only peat decomposition but also large CO_2-fluxes by plants and soil respiration based on the decomposition of fresh organic matter and (2) the measurements are not continuous, so not 24 hours a day and 365 days per year. There are several designs of closed chambers with advantages and disadvantages and the measurements require several corrections and calibrations (Duran and Kucharik, 2013; Venterea and Parkin, 2012; Koskinen *et al.*, 2014; Pedersen *et al.*, 2010). It is also a problem that most measurement series are limited in time to sometimes just a season and in best cases to one or a few years, and in place to a few spots in the field. Another method is the use of micro-meteorological measurements making use of eddy-covariance techniques (Aubinet *et al.*, 2000, 2003, Jacobs *et al.*, 2007). However, although the measurements cover large areas and usually continue over at least one year, the problem remains that not only peat oxidation but also the much larger respiration of soil biota and fauna and vegetation is measured.

4.5.3 Determination of OM decline and CO_2 emissions from subsidence

Subsidence of drained peat soils is caused by consolidation of the peat layer and by oxidation of the organic matter of unsaturated peat above groundwater level and by permanent shrinkage of the peat soil above the groundwater level (Schothorst, 1977; Kasimir-Klemedtsson *et al.*, 1997). In the first years after drainage of peat lands, a major part of the subsidence is caused by consolidation and permanent shrinkage. In this respect it should be noted that in Europe many peat soils have been drained some time ago and were subjected to an improved drainage in the 19th and 20th century when new techniques made this possible. After that first period with consolidation and shrinkage, the major driver of the ongoing subsidence is oxidation (Schothorst, 1977, Pronger *et al.*, 2014). When the ditch water levels are adapted to the ongoing subsidence, which is common practice in agriculture, this subsidence can go on for a very long time. The amount (mass) of peat that is oxidized and the related yearly CO_2–C emission due to the subsidence of peat soils can be calculated according to (Kasimir-Klemedtsson *et al.*, 1997):

$$CO_2 - C_{em} = F \cdot S_{mv} \cdot \rho_{so} \cdot fr_{os} \cdot fr_c \cdot 10^4 \qquad [4.3]$$

Where $CO_2 - C_{em}$ is CO_2 emission (kg. CO_2 – C $ha^{-1}.yr^{-1}$), F is fraction subsidence due to oxidation of organic matter compared to total subsidence, S_{mv} is total subsidence ($m.yr^{-1}$), ρ_{so} is bulk density of peat ($kg.m^{-3}$), fr_{OS} is organic matter fraction of peat (-) and fr_C is carbon fraction of organic matter (-). The factor 10^4 is needed to convert the carbon emission C from (kg $C.m^{-2}.y^{-1}$) into (kg $C.ha^{-1}.y^{-1}$). In equation (1) the parameters F, ρ_{so}, fr_{OS} and fr_C are according Kasimir-Klemedtsson *et al.* (1997) related to the peat in the topsoil (upper 20 to 30 cm of the soil). The subsidence is determined from elevation measurements or measurements of peat depths in time. Difference in time should be at least 10 years. Van den Akker *et al.* (2008) recommend measuring surface elevations or peat depths in early spring, because at that time the peat is in general completely wet and swollen, so no temporally subsidence due to drying shrinkage is measured.

A disadvantage of determining the CO_2-C emission according to Eq (2) is, however, that the bulk density and fraction subsidence due to oxidation of organic matter compared to total subsidence (F) is generally not known, and this sometimes even holds for the organic matter content. According to a literature study by Armentano and Menges (1986), F varies mostly between 1/3 to 2/3, which is a relatively large range. The bulk density of peat soils can also vary from approximately 100-300 $kg.m^{-3}$ (Kasimir-Klemedtsson *et al.*, 1997) with the higher value being indicative of the topsoil of peat soils in agricultural use, with well decomposed peat, classified as “sapric”, and the lower value being indicative for freshly reclaimed or semi natural peat lands that contain more fibres, classified as “fibric”. In between are peats that are somewhat decomposed, classified as “hemic” (Andriesse, 1988).

To overcome this disadvantage of varying peat density, Van den Akker *et al.* (2008), used a fraction F = 1 combined with values of ρ_{so}, fr_{OS} and fr_C for the unripened (fibric) peat layer in the subsoil (at a depth of e.g. 120 cm). Van den Akker *et al.* (2008) used a mass balance approach, in which the peat soil layer above the phreatic groundwater level in summer is considered to be stable after decades of ripening (permanent shrinkage, oxidation and humification). This means that the total amount of organic matter in this layer is more or less constant. This can only be true if the subsidence caused by the oxidation is continuously followed by a coincident lowering of the ditch water level (and so the groundwater level). In that way the loss of organic matter by oxidation, resulting in an outflow of C as CO_2-emission, is compensated by an inflow of C in organic matter (unripened, fibric peat) from below the stable layer. In a way the upper layer is "eating" its way downward. Advantages of the method are that it is robust (Couwenberg and Hooijer, 2013) and that a subsidence over 20 to 30 years represents the cumulated oxidation and CO_2-C emissions during these years.

4.5.4 Determination of OM decrease and CO_2 emissions from N mineralization.

This is an indirect method, in which CO_2 emissions are calculated from the N mineralization caused by the oxidation of the peat soil. The C mineralization is derived by multiplying the N flux with the C/N ratio of peat soils and this is the basis for the calculation of the CO_2 emission. Peat soils supply more nitrogen to the crop than mineral soils. The additional N-supply can be calculated from the N-content of the dry matter yield of e.g. grass. The loss of organic matter (peat) can be calculated from the additional N-supply, taking the N-supply of mineral soils as a reference level (Schothorst, 1977). A disadvantage of this method is that it is an indirect method and that it requires an accurate figure for the percentage of the mineralized nitrogen that is used by the grass and yielded as crop. An advantage is that it can be considered as a more or less continuous measurement, because per year the cumulated mineralized N is considered.

4.5.5 Determination of OM decrease and CO_2 emissions from the relative increase of mineral parts

This method is based on the progressive increase in peat mineral content after drainage and cultivation, as measured over a long time period. Organic matter loss can be computed, assuming that it is the essential driver for the concomitant increase in soil mineral content. This method was presented by Grønlund *et al.* (2008) who compared it with two other approaches: long-term subsidence rates, and soil CO_2 flux measurements. All three methods yielded comparable OM decreases and CO_2 emissions. It is essential in this method that the ash contents (mineral parts) are measured on two dates, which means that historic measurements are needed. About 10% of the increase of mineral parts could be accounted to liming (mainly) and fertilization. Leinfeld *et al.* (2011) used the same method, however, they used only recent measurements from single soil profiles measuring the difference in ash content between the acrotelm, which is the aerated upper part of a mire, and the underlying catotelm, which is permanently water saturated. This means that no historic data is needed, because the ash content in the catotelm represents the historic undrained situation. Leinfeld *et al.* (2011) investigated four ombrotrophic peat lands (bogs); results were satisfactory for two sites and it was assumed that disturbance of these bogs was minimal without former drainage periods or import of minerals. It was assumed that in general fen peat soils are too much disturbed in history for this method.

4.5.6 Determination of subsidence, OM decline and CO_2 emissions based on water levels

There is a strong relation between subsidence and water levels (Andriesse, 1988; Dawson *et al.*, 2010; Kluge *et al.*, 2008; Renger *et al.*, 2002; Schothorst, 1977; Van den Akker *et al.*, 2008; Wessolek *et al.*, 2002) and therefore also between water levels and CO_2 emissions. However, it should be noted that the relation between water level and CO_2 emission is not undisputed: Berglund and Berglund (2011) measured in a lysimeter experiment significantly lower CO_2 emissions at water table depths of 80 than at 40 cm. These findings could be confirmed with results in the literature. Nevertheless a meta-analysis of published flux measurements by Couwenberg *et al.* (2008, 2011, 2013) shows that indeed mean annual water level is a good proxy for CO_2 fluxes. In Schils *et al.* (2008) and Verhagen *et al.* (2009) a relation between the CO_2 emission and mean annual water level is presented (Figure 4.2). The figure shows that this relation is much stronger for shallow than for deeper ditchwater levels. Van den Akker *et al.* (2008) showed that the relation between the deepest groundwater and subsidence is more than twice as strong as the relation between ditchwater level and subsidence: a 10 cm deeper deepest groundwater level results in an extra annual subsidence of 3.7 mm, while a 10 cm deeper ditchwater level results in an extra annual subsidence of 1.5 mm. Including temperature in the relations between water level and subsidence and CO_2 emissions makes it possible to account also for climate change (Renger *et al.*, 2002; Wessolek *et al.*, 2002). The use of relations between water tables and subsidence, OM decline and CO_2 emissions makes it possible to use hydrological models to compute the impact of water management strategies on the reduction of subsidence, OM decline and CO_2 emissions

(Querner *et al.*, 2012; Renger *et al.*, 2002; Wessolek *et al.*, 2002). According to Querner *et al.* (2012) a climate scenario W+ with an increase of temperature with 2^0 C in 2050 and a change in air circulation with more

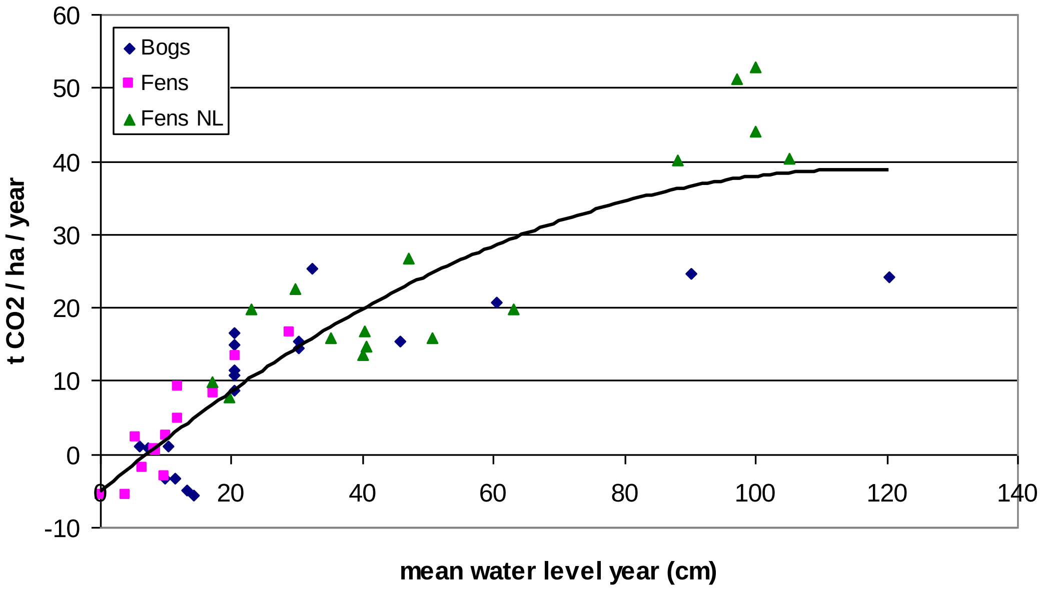

Figure 4.2: CO_2 emission of peat (Y) in relation to mean annual water level (X) below soil surface: $Y = -0.0033 X^2 + 0.7488 X - 5.21$ ($R^2 = 0.81$)
Agricultural peat soils have at least a mean ditchwater level of 20 cm minus soil surface. Data collected by Couwenberg et al., (2008) based on direct measurements of CO_2 emissions and Van den Akker (Fens NL, unpublished data) based on CO_2 emissions calculated from measured mean annual subsidence.

easterly wind and drier summers will increase the subsidence rate and so the CO_2 emission by almost 70 %. This can be subdivided by 25% due to the temperature rise of 2^0 C and 45% due to the deeper groundwater levels due to the drier summers.

4.5.7 Determining greenhouse gas emissions from peat lands using vegetation as a proxy

Couwenberg *et al.* (2011) outline a methodology to assess emissions and emission reductions from peat land rewetting projects using vegetation as a proxy. Vegetation seems well suited to indicate GHG fluxes from peat soils as it reflects long-term water level, affects GHG emissions via assimilate supply and aerenchyma and allows fine-scaled mapping. The methodology includes mapping of vegetation types characterised by the presence and absence of species groups indicative for specific water level classes. GHG flux values are assigned to the vegetation types following a standardized protocol and using published emission values from plots with similar vegetation and water level in regions with similar climate and flora. The use of vegetation as a proxy for GHG fluxes allows for a rapid and relatively cheap estimate of baseline and project scenario emissions and thus of emission reductions from rewetting.

4.5.8 Determination of subsidence, OM decline and GHG emissions with process-based models

Hendriks *et al.*, (2008) used the process-based model combination SWAP-ANIMO to simulate peat land loss of OM and GHG emissions of CO_2, CH_4 and N_2O, soil subsidence and nutrient loading of surface waters. The model combination comprises of two dynamic models: SWAP (Kroes and Van Dam, 2003) for simulating the hydrology of saturated and unsaturated zone and soil temperature, and ANIMO (Groenendijk *et al.*, 2005; Renaud *et al.*, 2006) for simulating the carbon(C)-, N- and P-cycles, C-, N- and P-leaching to ground- and surface water and evolution of CO_2. Hendriks *et al.* (2008) extended the model with a description of soil surface subsidence and evolution, sulphate (SO_4) cycle, transport and emission of N_2O and CH_4. N_2O evolution is related to nitrification and denitrification in the model. The model was calibrated and validated with field measurements. Overall performance of the model proved to be satisfactory. A disadvantage of such a model is the required detailed input of soil properties such as soil hydraulic properties, decomposition rates, shrinkage characteristics, chemical properties etc and data to run the model. On the other hand a complete picture of the impact of peat soil degradation on the environment is derived. The detailed hydrological basis of the model allows for scenario studies of effects of water management measures. Scenario runs were performed to evaluate the effects of climate change. The worst case climate change scenario results in almost doubling of subsidence and GHG emission at the end of this century (Hendriks *et al.*, 2008).

4.6 Effects of decline in SOM in peatsoil on soil functions

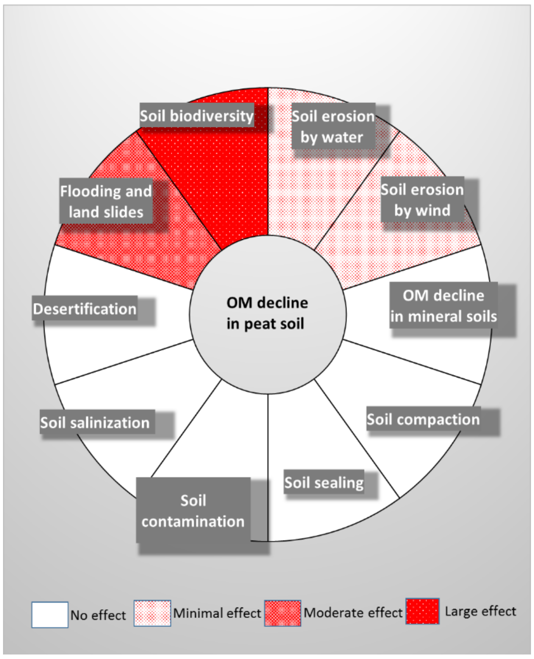

Figure 4.3: Effects of decline in SOM in peatsoil on other soil threats. Red is negative effect.

Degraded peat soils in arable agriculture or in overgrazed grassland 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. However, in this paragraph we will concentrate on the effect of peat degradation on other soil threats and not the other way around.

Peat soils in arable agriculture are vulnerable to severe drying of the topsoil resulting in severe hydrophobia making the soil less suitable for agriculture and very prone to water erosion and especially wind erosion.

Natural peat lands and even not completely degraded peat lands store water and act as a sponge. They absorb and retain water during periods with a surplus of precipitation and slowly release water in times of water deficit. In this way, peat lands slow down peak discharge and prevent water erosion. An example for the importance of peat lands for water regulation are the Ruoergai peat lands on the eastern Tibetan Plateau (Joosten *et al.*, 2012), which reduce downstream flooding and guarantee a steady supply of water to the Huanghe (Yellow) River. The loss of this regulation function results in an increased risk of desertification of downstream regions. These peat soils are severely endangered by overgrazing.

Natural peat soils are hotspots of biodiversity. After drainage this biodiversity changes and decreases, however, if used as grassland these peat soils are still rich in biodiversity and can also be a very important habitat for, among other things, meadow birds. Further degradation can result in degradation of habitats of certain species. On the other hand rewetting can also be very harmful to certain species e.g. meadow birds.

4.7 Effects of decline in SOM in peat soils on soil functions

The soil functions that will be considered are the ones identified in the Soil Thematic Strategy:

4.7.1 Food and other biomass production

Oxidation of peat soils results not only in emissions of CO_2, but also in mineralization of N, which means that the degradation of peat soils provides an important supply of nutrients and therefore increases the food and biomass production considerably! The bearing capacity of wet peat soils is low resulting in reduced trafficability and possibilities for soil management and soil tillage. In the case of grassland trampling and loss of grass production peat soil degradation can be severe. Therefore the drainage basis must be adapted from time to time to the subsidence otherwise peat soils become too wet for agriculture. Ongoing oxidation and loss of peat results in time in the total loss of the peat layer. Food production and suitability for agriculture than depends on the fertility and soil physical properties of the soil underneath the original peat layer. Many of these soils are acid (Wösten *et al.*, 1997) and require a lot of time and effort to improve. Leaching of acid water from these acid soils can cause severe problems in adjacent open waters.

Biomass production on peat soils other than food or grass is very questionable. It must be economically viable and there must be a profitable balance between the GHG emissions and the C- sequestration in the biomass. This might be possible in very wet systems such as paladiculture. Paludicultures (Latin ‘palus’ = swamp) are land management techniques that cultivate biomass from wet and rewetted peat lands under conditions that maintain the peat body, facilitate peat accumulation and sustain the ecosystem services associated with natural peat lands (Joosten *et al.*, 2012).

4.7.2 Environmental interaction: storage, filtering, buffering and transformation

Peat soils have a high storage, filtering, buffering and transformation capacity. Loss of peat results in loss of these capacities, especially the storage of C. Moreover the oxidation of peat results in GHG emissions and mineralization and release of N, P and S and other minerals including eventually chemical contaminants, dissolved OM (DOM) and peat particles towards surface waters.

4.7.3 Biological habitat and gene pool

Drainage of natural peat soils results in a significant change in biodiversity. In this chapter we concentrate on agricultural soils, which can be subdivided in grasslands and croplands. Grassland has a high capacity for biodiversity and this is much higher on peat soils than on for example. sandy soils. This depends also on how intensively the grassland is used. Peat land meadows are also important habitats for meadow birds and other animals and insects. Peat soils in arable agriculture have a much lower biodiversity capacity than grasslands. If the peat layer has completely disappeared by oxidation, then this means in general also a strong decrease in biodiversity, especially if the soil underneath the original peat layer is suitable for arable agriculture, while the original peat soil was mainly suitable and in use as grassland.

4.7.4 Physical and cultural heritage

Peat soils are by nature historical archives and can store artefacts of ancient cultures and human bodies. Drainage and oxidation of peat results in a total loss of this historical archive. Several peat land landscapes are considered to be of high cultural and historical value. An example is the central part of Holland and Utrecht in the Netherlands, the so called Green Heart. Loss of peat and subsidence and eventually complete loss of the peat layer can change this landscape considerably. Wet nature reserves degrade due to drainage towards surrounding lowered agricultural land with a much deeper drainage basis. In some countries peat land areas are highly appreciated as open landscape in a further “closed” forest landscape. Abandonment of agricultural peat land and spontaneous growth of trees will destroy this open landscape.

4.7.5 Platform for man-made structures: buildings, highways

Peat soils are in fact not suited for building and infrastructure. Nevertheless many buildings are erected in peat land areas. Because the bearing capacity of peat soils is very low these buildings have in general a foundation on piles reaching into the firm deep subsoil. Historically these pile foundations are made of wood. Lowering water table levels by drainage to compensate for the subsidence from time to time results in exposure of the wooden foundation to oxygen and in that way to rot. On the other hand the subsidence of gardens and agricultural land and infrastructure without a pile foundation is going on. The driver of the subsidence of the gardens and agricultural land is oxidation. The driver of the subsidence of the infrastructure without piles is the consolidation and compaction of sometimes 16 m thick peat layers by the weight of the roads and other infrastructure which have to be raised periodically to compensate the subsidence. The difference in elevation of the buildings on piles and the subsiding gardens, pavements and roads are causing problems with the connection of public infrastructure such as sewage, electricity, drinking water etc with the houses. We see also problems with exposure of foundations, such as the need to add stairs to get an entrance into buildings and increasing problems with drainage and periodically flooding of roads and garden during wet periods.

4.7.6 Source of raw materials

Peat is drained and dried to the air and collected (harvested) for energy and horticulture. Only a relatively small part of the total European peat land area is affected, however, because almost the whole peatlayer is harvested it represents a notable impact on the peat C balance (Schils *et al.*, 2008). Extraction for energy has declined since the mid 19th century, but remains significant in Ireland, Finland, Sweden, the Baltic States and Russia (Byrne *et al.*, 2004). Extraction for horticulture has led to the loss of a large part of the lowland bog area in the UK (Moore, 2002). Alm et al, 2007 estimated a GHG emission of about 7.3 ton CO_2-C-eq. ha^{-1} yr^{-1} of the drained peat land and stockpiles, these are exclusively the emissions due to oxidizing of the peat for horticulture and peat for heating and energy.

It can be argued that it is better to exploit and harvest drained peat lands in agricultural use or abandoned peat lands than let the peat oxidize and be “burnt” in that way. After removal of most of the peatlayer the land can be restored for agricultural use or even better as (potential) peat growing wetland. It should be noted that just a part of the peat or peat land is suited for harvesting and combustion or horticulture. Arguments against this are that harvested peat will be converted to CO_2 in a very short time, so the emission expressed in ton CO_2-C yr^{-1} is very high. Furthermore, restoration of agricultural peat land to wetland without first harvesting the peatlayer will be a better option to reduce CO_2 emissions.

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5 DECLINE IN ORGANIC MATTER IN MINERAL SOILS

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5.1 Description of decline in SOM in mineral soils

While almost all soils consist of varying amounts of organic components and mineral components, most soils are either predominantly organic or predominantly mineral. In general, soil horizons that contain ≥20-35% of organic matter, by weight, are classified as organic (USDA-NRCS, 1998). Soils themselves are then characterized as organic if the cumulative thickness of organic horizons amounts to at least half of their uppermost 80 cm, while the other soils – the topic of this chapter - are regarded as mineral.

In spite of four centuries of research on soil organic matter (SOM), a generally accepted, standardized definition of SOM is still lacking (Kleber and Johnson, 2010). This problem exists because of the heterogeneity of SOM in terms of source, composition, functions and dynamics, on the one hand, and, on the other hand, the advancements in understanding favoured by modern spectroscopy techniques. Huber *et al.* (2008) reviewed various definitions of SOM and identified three main causes for discrepancies: (i) inclusion/exclusion of living biomass; (ii) inclusion/exclusion of litter, fragmentation layers; and (iii) threshold level of decomposition. The Soil Science Society of America (SSSA, 1987) defines SOM as the total organic fraction of a soil exclusive of non-decayed plant and animal residues. A similar definition was adopted by the European Union's Soil Thematic Strategy (EC, 2006). SOM contents/stocks, however, are typically quantified as soil organic carbon (SOC) contents/stocks (Baldock and Nekson, 2000). Carbon is the prime element present in SOM, comprising 48%-58% of the total weight. Other elements include hydrogen (H), oxygen (O), nitrogen (N), phosphorus (P) and sulphur (S) with an average C:N:P:S ratio of 100:10:1:1. Chemical components of SOM encompass those derived from plant, animal and microbial residues, and their transformation products, termed as humic substances (Table 5.1; Dungait *et al.*, 2012).

Other widely used approaches to describe SOM composition are based on functional criteria related to microbial activity and chemical components (Table 5.1). The former, so-called functional approach has been widely applied in SOM simulation modelling (e.g. Century, RothC), in which different SOM pools are defined kinetically by means of their mean resident time (MRTs) reflecting the integrated effect of biochemical and physical factors on SOC turnover and C fluxes. Usually three pools (Table 5.1) are identified (Stockmann *et al.*, 2013) labile (MRT <4 yrs), stable (MRT = 15-100 yrs) and passive (MRT = 500-5000 yrs).

Soil, after the oceans, is the largest pool of carbon on earth. The SOC pool is about twice the size of the atmospheric carbon pool and about three times the size of the biotic carbon pool. The global SOC pool to a depth of 1m is estimated at 1,500 billion tonnes (Batjes, 1996), ranging from 30 t ha^{-1} in arid climates to 800 t ha^{-1} in permafrost-affected regions (Lal, 2004). Other estimates put these figures even higher. The topsoils of the EU-28 store around 73 Gt of OC (Jones *et al.*, 2005), of which 21.5 Gt are associated with farmland (13 Gt on arable and 8.5 Gt on pasture) (OCTOP database). However, there are great uncertainties in assessing the stock of SOM across Europe (Panagos *et al.*, 2013a) and, hence, of changes therein. The C pool in world soils has been estimated to have decreased 78 ±12 Gt between 1850 and 1998, mainly through accelerated mineralization (two-thirds) and soil degradation (one-third) enhanced by soil cultivation and disturbance (Lal, 2004).

SOM decline has been widely recognised as a major threat for sustainable soil management because of the pivotal role played by the organic material on many soil functions, like food and biomass production, storage and filtering, biological habitat and gene pool, etc. The problem of SOM decline thus has become a topic of active scientific research in the last decade (2004-2013), having been mentioned in more than one thousand publications (SCOPUS database) at a yearly publication rate that has progressively increased from 61 to 154.

Table 5.1: *Pools of SOM defined according to their mean residence time and corresponding compound classes (After Dungait et al., 2012).*

Residue type	Pool category	Residence time (years)	C/N	Compounds
Litter	Fast (or labile)	0.1-0.5	10-25	Simple sugars, Amino acids Starch
	Fast (or labile)	2-4	100-200	Polysaccharides
SOM	Fast (or labile)	1-2	15-30	Living biomass, Particulate organic matter, Polysaccharides
	Slow (or stable)	15-100	10-25	Lignified tissues, Waxes, Polyphenols
	Passive	500-5000	7-10	Humic substances, Clay: OM complexes, Biochar

5.2 State of decline in SOM in mineral soils in Europe

There is great uncertainty about SOM/SOC stocks in Europe and trends therein. To detect SOC changes at a regional level there are very few long-term soil monitoring networks with sufficient number of sampling sites (Saby *et al.*, 2008) and contrasting SOC trends among countries are often reported. For instance, Reynolds *et al.*, (2013) reported no SOC change in top soil of Great Britain for the period 1978–2007, as confirmed by Chapman *et al.*, (2013) analyzing data collected in Scotland for the same period. No unequivocal trend was detected in Denmark (Heidmann *et al.*, 2002) while small SOC increases were found in the Netherlands on a regional basis (Reijneveld *et al.*, 2009). Decreases in SOC content in Finnish cropland in 1975-2009 were reported by Heikkinen, et al, (2012) as well as in Belgium (Goidts and van Wesemael, 2007) and Bavaria cropland (Capriel 2013).

All the other existing estimates of SOC stocks are based on modelling exercises and contain a significant level of uncertainty, either because of the model used or due to uncertainties in the input datasets. EU Member States are currently not obliged to report on SOC in a harmonised manner and the reported SOC values involve several assumptions on the relationships of SOC values with soil type, land cover/use and climate (also affected by elevation). Uncertainties in the input datasets result amongst others from non-standard definitions of land cover categories, outdated soil information, and small-scale climatological information that do not reflect locally important microclimates.

The SOC values that are presented in most EU policy documents were derived from the OCTOP (Organic Carbon in TOP soil) dataset of the early 2000s. However, recent studies suggest that values derived from OCTOP may, in some circumstances, over-estimate SOC levels. These studies include the EIONET SOC data collection performed by the European Soil Data Centre in 2010, the LUCAS 2009 Topsoil Survey performed by the JRC in 2009, and the application of the novel modelling framework CAPRESE in the agricultural soils of Europe in 2013. The results of these four studies are summarized underneath.

5.2.1 OCTOP (Organic carbon in topsoil)

European topsoils (0-30 cm) store around 79 Gt of organic carbon, of which 73 Gt are stored in the EU-28. These estimates are derived from the OCTOP topsoil SOC content database (Jones *et al.*, 2005). They were based on a combination of revised pedo-transfer rules (PTR) and soil properties data contained in the European Soil Database.

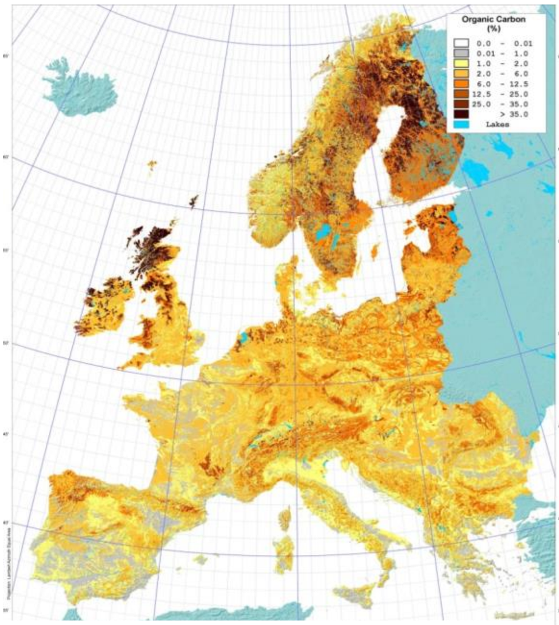

Figure 5.1: Topsoil (0-30 cm) organic carbon content (%) in Europe (Jones et al., 2005).

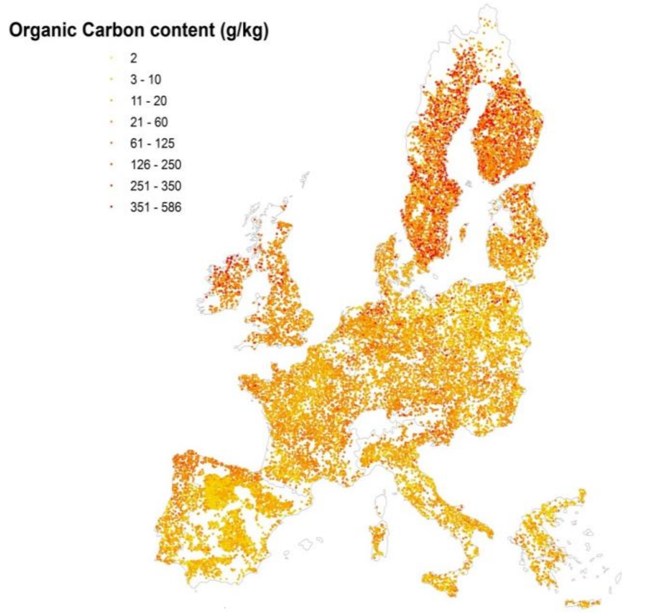

Figure 5.2: LUCAS Topsoil organic carbon content (g/kg) (Source: Toth et al., 2013a).

The data (Figure 5.1) clearly showed the broad distribution of SOC across Europe with a decreasing gradient from north (high levels correspond to peatlands) to south. Spatial analysis of the OCTOP database suggests that 30% of the EU topsoil SOC stock (around 22 Gt C) is in agricultural soils, of which 15-17% (around 13 Gt C) relates to cropland and 12% (around 8 Gt C) to pasture (as defined in the CORINE Land Cover database). By comparison, nearly 50% of the total EU stock is located in woodland soils.

5.2.2 EIONET 2010 SOC data

In 2010, the Joint Research Centre of the European Commission (JRC) conducted a project to collect data on soil organic carbon (and soil erosion) in Europe, using the European Environment Information and Observation Network for soil (EIONET-SOIL). Data were received from 12 EIONET members (32% of the total EIONET network) but only 5 members provided information for more than 50% of their geographical coverage. Furthermore, some of these five members provided data for agricultural soils only (e.g. Austria, Slovakia). The general pattern of the geographical distribution of EIONET data was followed by the OCTOP, but with higher values. The main reasons for the higher modelled data are due to peat lands that were drained, outdated input information, conditions of the pedo-transfer rule (PTR) in OCTOP, and the large Soil Mapping Units (SMUs) in the European Soil Database (Panagos *et al.*, 2013a).

5.2.3 LUCAS 2009 Topsoil Survey

The 2009 LUCAS Topsoil Survey included a component of soil sampling, with around 20 000 sampling sites in 25 EU Member States (except Romania and Bulgaria). LUCAS soil samples were taken from all land use/ cover types but with a focus on agricultural areas. The objective of the soil sampling was to improve the availability of harmonized data on soil parameters in Europe. The analysis results formed the LUCAS soil database (Toth *et al.*, 2013a), including SOC content in top soil (0–30 cm), expressed as g/kg (Figure 5.2). A data quality assessment was performed on the dataset, taking into consideration the main climatic zones, regions, land cover classes and management practices (Toth *et al.*, 2013b). Results highlighted important links among these factors and helped to understand and quantify the potential of European croplands concerning C content and other indicators of soil health. Woodland and shrubland showed the highest levels of SOC in all climatic regions; this pattern was in line with the common understanding of high values of SOC in forest compared to other land cover classes. The lowest levels of SOC were observed in the Mediterranean climatic region; this general pattern confirmed that SOC content is higher in northern than in southern parts of the continent. Levels of SOC in arable land in the boreal climatic region were at least three times higher than in the other climatic regions. LUCAS topsoil data were compared with the modelled European topsoil organic carbon content data (OCTOP). The best agreement existed at the NUTS2 level but showed underestimation by the OCTOP values in southern Europe and overestimation in the new central eastern Member States (Panagos *et al.*, 2013b). The agreements were especially good for certain regions in countries such as the United Kingdom, Slovenia, Italy, Ireland and France.

5.2.4 CAPRESE Modelling of SOC stocks in European agricultural soils

The evolution of EU policies on C accounting and sequestration may be constrained by the lack of accurate SOC estimations and the lack of tools to conduct scenario analyses, especially for agricultural soils. Therefore, a comprehensive model platform was established at a pan-European scale (EU + Serbia, Bosnia and Herzegovina, Croatia, Montenegro, Albania, Macedonia and Norway), using the agro-ecosystem SOC model CENTURY. Almost 164 000 combinations of soil-climate-land use were analysed, including the main arable crops, orchards and pasture. The model was implemented with the main management practices (e.g. irrigation, mineral and organic fertilization, tillage) derived from official statistics.

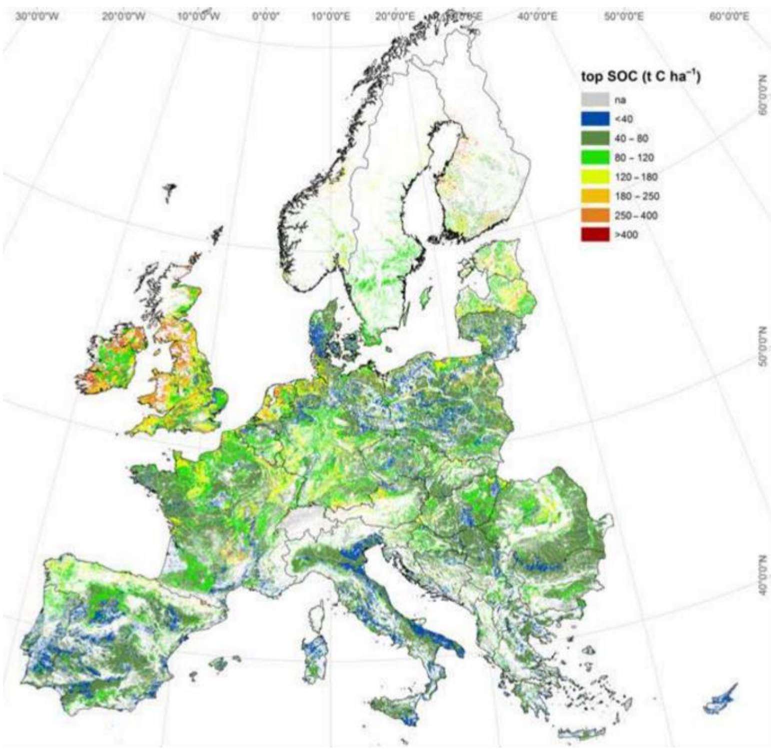

Figure 5.3: Soil organic carbon (SOC) stock in the top-soil layer (0–30 cm) of European agricultural soils (Source: Lugato et al., 2014.)

The CENTURY-based estimate of the current SOC stock in the top 0-30 cm of agricultural soils was 17.63 Gt (Figure 5.3), with an uncertainty below 36% in half of the NUTS2 regions. This supported the view that the OCTOP data have a degree of over-estimation.

The results showed that topsoil SOC values varied significantly across the EU. Using 2010 as baseline, the lowest values occurred in the Mediterranean region (often below 40 t C ha^{-1}) while the highest values occurred in north-eastern Europe (on average between 80 and 250 t C ha^{-1}). In the central and eastern European countries, the interaction between pedo-climatic and agronomic conditions resulted in complex SOC patterns. At latitudes above 50°N, the model simulated values < 40 t C ha^{-1} for parts of Denmark, northern Germany, Poland and Lithuania, which are characterized by coarse parent material deposited during the last glacial period. By contrast, the model simulated SOC stocks ranging between 80 and 120 t C ha^{-1} across Hungary and Romania, where the soils are very rich in clay. Some hot-spot situations were predicted in Ireland, UK, Netherlands and Finland with values > 250 t C ha^{-1}, corresponding to peatland areas. The model predicted an overall increase of this pool according to different climate-emission scenarios up to 2100. C losses in southern and eastern Europe (corresponding to 30% of all the simulated agricultural land) were compensated by gains in central and northern Europe (Lugato *et al.*, 2014). Generally, higher soil respiration was offset by higher C input as a consequence of increased CO_2 atmospheric concentration as well as more favourable crop growing conditions, especially in northern Europe.

5.3 Drivers/pressures

SOM amounts in the soil depend on biotic processes – essentially net primary production (NPP), the distribution of photosynthates in roots and shoots, and soil heterotrophic respiration – as well as physical processes such as leaching, runoff and erosion. Biotic processes are the most relevant but soil erosion may have a significant influence on SOM content (Lal, 2004).

All these processes strongly depend on physical, chemical and biological drivers of both natural and human origin, which are listed in Table 5.2. Since most of these drivers are the same as the ones that influence the composition of terrestrial ecosystems, SOM and ecosystem types show strong correspondences to one another (Post, 2006).

In natural ecosystems, climate is the main driver through the effects of temperature, moisture and solar radiation. Sensitivity of NPP to moisture availability is higher than that of decomposition rates, while the opposite is observed in the case of temperature (Post, 2006). As a result, SOM is positively correlated with precipitation and negatively with temperature, explaining for example the general pattern of decline from northern to southern Europe (cf. 5.2).

There are anyway uncertainties about the effects of climate change on SOC content. Wu *et al.* (2011) after analysing data from 85 manipulation experiments across the world 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.

At local scale the effect of vegetation type on SOM increases in importance. The initial decomposition rates of plant residues are negatively correlated with the substrate C: N ratio (Table 5.1) or the fraction of plant tissue that cannot be solubilized by strong acid treatments (operationally defined as "lignin"). This lignin fraction encompasses the plant derived molecules (e.g. lignin, polyphenols) and confers what is defined as "primary recalcitrance" (Kleber, 2010). The "secondary recalcitrance" is associated with microbial products, humic polymers and charred materials (Amelung *et al.*, 2008; Schmidt *et al.*, 2011; von Lützow *et al.*, 2006).

Soil type is a major factor involved in the stabilization mechanisms of SOM by means of physical preservation (von Lützow *et al.*, 2006; Dungait *et al.*, 2012), such as occlusion of SOM within aggregates, adsorption onto minerals and regulation of microbial activity as substrate supply. The saturation of physical stabilization mechanisms (Six *et al.*, 2002) limits the potential SOM storage. According to this emerging view, the persistence of SOM is primarily an ecosystem property, affected by physicochemical and biological characteristics rather than by the molecular properties of organic matter (Schmidt *et al.*, 2011)

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.

The impact of agricultural management on SOM and soil quality has been studied for a long time, as SOM plays a central role in the nutrient and water cycles One of first scientific experiments dates back to 1850 when the effect of organic fertilisation on plant macro elements uptake was investigated at Rothamsted (Johnston *et al.*, 2009). Most recently, interest in the topic has been boosted (e.g. Schils *et al.*, 2008, Gobin *et al.*, 2011; Aguilera *et al.*, 2013) because of the potential contribution of "judicious" land use and recommended management practices (RMPs) to soil carbon sequestration. Management systems affect SOM mainly through: a) the input rates of organic matter (e.g. NPP and external C input) and its decomposability; b) the distribution of photosynthates in roots and shoots; c) the physical protection of SOM. According to Lal (2004) high SOM accumulation is favoured by management systems, which add high amounts of biomass to soil, cause minimal soil disturbance, improve soil structure, enhance activities and species diversity and strengthen mechanisms of element cycling.

In general, low disturbance, high C input (e.g. litter and roots) and SOM content typify grasslands and forests. This implies that their conversion to arable crops causes a dramatic depletion of SOC stock, estimated in Europe at around 1-1.7 t C ha^{-1} y^{-1} from grasslands to arable and 0.6 t C ha^{-1} y^{-1} from forest to arable (Freibauer *et al.*, 2004). The figures reported by the same authors are more variable in the case of the reverse conversion - from arable to grassland - ranging from 0.3-0.6 to 1.2- 1.7 t C ha^{-1} y^{-1}. Agricultural practices that increase C input encompass the use of crop residues and straw, manure or hexogen (e.g. compost) C input, and, in general, actions that lead to a higher complexity of the cropping systems (e.g. introduction of cover crops, ley crops, higher number of plant species). Differences in root architectures between crops also influence the repartition between above-ground and belowground biomass. The impacts of deep root systems have been estimated to amount to an additional 0.6 t C ha^{-1} y^{-1}, The effects of residues incorporation are more uncertain. For example, Powlson *et al.* (2011) reported little to no change in 25 long-term experiments, whereas intermediate results were observed by Morari *et al.* (2006) who observed small increases (0.1 t C ha^{-1} y^{-1}) in a long-term experiment (50 years). The relevance of root C input has been emphasised by Rasse *et al.*

(2005), who estimated that the mean residence time of root-derived C was 2.4 times that of shoot-derived C. Similar results were obtained by Kätterer *et al.* (2011). The higher stabilisation was attributed to physical protection mechanisms and only a small proportion to chemical recalcitrance.

The chemical nature of manure (i.e. effect of animal species) appears not to be relevant. According to the meta-analysis conducted by Maillard and Angers (2014) of 49 experiments of manure applications (average study duration of 18 years), only the manure C-input significantly affected SOC stocks. Conflicting evidence on the effects of the nature of the manure (e.g. animal species, manure management systems) on SOM has been reported by other authors. In any event, indications drawn from Rothamsted experiments highlight that achieving a significant increase in the equilibrium level of SOM in a farming system under temperate climate requires continuous application of large inputs of organic matter (Johnston *et al.*, 2009). Conversely, variations in SOM level driven by normal C input are usually small and in most cases, the new SOC equilibrium is only reached after many years (Johnston *et al.*, 2009).

Conservation tillage (e.g. no-tillage-NT) has been widely endorsed as reducing soil disturbance, preserving the soil structure and enhancing SOM content. Consequently, a transition from conventional to NT has generally been considered as an efficient strategy to improve C sequestration (with an additional 0.1-1 t C ha^{-1} y^{-1}) (Luo *et al.*, 2010). Estimates of SOC accumulation are usually derived from data collected in the topsoil (< 30 cm) and are probably biased by the sampling depth. On the contrary, studies that involved deeper sampling generally showed no C sequestration advantage for conservation tillage (Baker *et al.*, 2007). Vertical SOC distribution would be primarily affected by the root growth and distribution with depth, since a well-developed root system may transfer the biomass to deeper layers and enhance its residence time in the soil. NT would improve a more superficial root lateral development and limit penetration due to excessive soil compaction, increased water accumulation in the soil surface and suboptimal soil temperature. SOC stratification can also be affected by ploughing by incorporating crop residues into the deeper soil profile.

Table 5.2: Drivers affecting of SOM content (Elaborated from STS, CLIMSOIL and CAPRESE projects).

a) Natural
• Climate (precipitation, temperature, solar radiation, etc.) • Topography • Soil type and properties (e.g. soil texture, soil temperature, moisture, pore structure) • Land cover/vegetation type
b) Anthropogenic/human activities

• Land management
    o Grazing intensity and grass coverage
    o Tillage and soil disturbance
    o Residues management/Bare fallow
    o Crop variety and species management
    o Intensive farming (e.g. Fertilisers/manuring/pesticides, simple crop rotation and high mechanisation)
    o Deforestation
    o Biomass burning
    o Drainage of wetlands
• Land use change/conversion (e.g. grasslands and woodlands to agriculture or urban areas – “soil sealing”)
• Contamination/Pollution

c) Socio- economic-politics
• Technological change/development • Policies (Agricultural – Environment – Energy sectors) • Economic growth and cost/price squeeze

The SOM cycle is also affected by other external drivers and pressures such as government policies (e.g. agri-environment, energy), technological developments, climate change and demographic trends, etc. (Table 5.2), mainly through changes in land use and agricultural management. For example, Good Agricultural and Environmental Conditions (GAECs) form part of the requirements under Cross Compliance and apply to farmers who receive payments under the Single Payment Scheme (SPS) and certain Rural Development schemes. Maintaining land in GAEC includes requirements to maintain SOM as well as soil structure, and to reduce soil erosion. Maintenance of grassland areas is an additional compulsory component of GAECs. According to Gobin *et al.* (2011), abolishing the EU permanent grassland restrictions would cause a 30% higher loss in SOC stocks than maintaining these restrictions would. The same authors simulated the implications of a bioenergy-production oriented policy in Europe and found a detrimental effect on SOM content as well as GHG emissions.

5.4 Key indicators for decline in SOM in mineral soils

In the context of the ENVASSO project (Huber *et al.*, 2008) a working group identified several key indicators for SOM decline (Table 5.3). The indicators address three main issues: SOM status, SOM quality and human-induced causes of SOM change. The selection of these indicators was done based on a ranking of expert judgement according to the indicators' sound science, measurement, policy relevance and geographical coverage. The experts further identified topsoil organic carbon (SOC) content (%) and SOC stocks (t ha^{-1}) as the most appropriate indicators for evaluating SOM status. SOC content has been measured by soil surveys (e.g. LUCAS), so that there are sufficient data to use this indicator at national as well as European level. SOC stock is determined by combining SOC content with either bulk density measurements or modelling applications (Century, DNCD, etc.). The selected SOM quality indicator was the C: N ratio of the topsoil (%). Numerous data are available on topsoil C: N ratios in the soils of Europe. Indicators of human-induced causes of SOM change were selected as: land cover change (Km^2), wild fires (Km^2), crop residues burning (Km^2), exogenous organic matter additions (t ha^{-1}) and organic farming (%). All these indicators can be assessed using remote sensing products or available census data at NUTS3 level.

In the selection process, the ENVASSO working group rejected a number of indicators, including 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. Those indicators were not selected due to their poor geographical coverage, a lack of existing data, a lack of scientific consensus on methodological issues and/or the lack of sufficiently robust methods.

Nonetheless, it is increasingly accepted that carbon at greater soil depths should be accounted for in future assessment, because it contributes to more than half of the global soil carbon stock and because; its response to land use change can equal that of the top layer (0-30 cm) (Schmidt *et al.*, 2011).

Dexter *et al.* (2008) proposed the clay/SOC ratio as a potential indicator of the relationship between soil physical conditions and SOM. The authors based their suggestion on the theoretical assumption that a soil's physical behaviour is not regulated by the total SOC but by its complexed fraction. This concept was verified by the authors for the matrix porosity of French and Polish soils, and by Schjønning *et al.* (2011) for the clay dispersibility of Danish soils.

5.5 Methods to assess status of SOM in mineral soils

The most appropriate methods to assess SOM status depend on several factors such as the temporal and spatial resolution of the survey, the availability of soil, land use and management data at local/regional scale, and the existence of harmonized monitoring networks. In their overview, Kuikman *et al.* (2012) reviewed risk assessment methodologies on soil organic matter decline and highlighted that an official methodology is still lacking in the EU.

IPCC (2006) has built a decision tree for the identification of appropriate tier levels to estimate changes in carbon stocks in mineral soils. The methods suggested for tier 1 and tier 2 were based on the application of default (tier 1) or country specific (tier 2) stock change factors that included a land-use factor, a management factor and a input factor representing different levels of C input to soil. However, these methods provide a simple representation of the SOC dynamics, and do not capture the complex annual variability in C fluxes or the long-term effects of land use and management. Citing IPCC (2006), "tiers 1 and 2 represent land-use and management impacts on soil C stocks as a linear shift from one equilibrium state to another". In order to assess the non-linear behaviour of SOC in soils, the implementation of a measurement-based inventory or advanced mathematical models (e.g. CENTURY, Roth-C) have been suggested (tier 3).

Table 5.3: Key indicators for soil organic matter (after Huber et al., 2008; Dexter et al., 2008).

Indicator	Item	Advantages	Disadvantages
Topsoil organic carbon content (g/kg)	SOM status	• indicator measured directly • indicator related to other potential soil threats (e.g. erosion, decline in soil biodiversity) • indicator available in most European Member States	• indicator not always easily accessible and harmonized • differences in terms of analytical methods and sampling protocol (e.g. sampling depth) • indicator changes difficult to assess in the short term period (< 5 yrs)
Topsoil organic carbon stocks (t/ha)	SOM status	• indicator appropriate for monitoring SOM changes • indicator measured directly (SOC + bulk density) • indicator related to other potential soil threats (e.g. soil compaction, decline in soil biodiversity)	• bulk density requires more analytical efforts and adds uncertainties in the results (high spatial and temporal variability)
C:N ratio	SOM quality	• simple indicator • numerous data are already available at EU level	• indicator not always easily accessible and harmonized • differences in terms of analytical methods and sampling protocol
Deep (1-m depth) soil organic carbon stocks (t/ha)	SOM status	• indicator appropriate for assessing global C cycle and GHGs emissions	• geographical coverage of SOC and bulk density measurements to this depth in existing soil monitoring networks is very poor
Clay/SOC	SOM quality	• indicator able to describe interaction between SOM and mineral particles	• indicator not yet thoroughly tested

Assessment of SOM status in soils according to measurement-based inventory requires two different methodological issues to be considered: a) measurement of SOM content and its conversion to SOM stocks; b) optimization of soil sampling, in order to maximize sensitivity to field-level changes in soil C following changes in land use or management (Conant and Paustian, 2002). Spatial variability in SOM is often many times greater than temporal variability in SOM, introducing unavoidable uncertainties in detecting changes in SOM stocks.

SOM content is difficult to measure directly (Islam, 2006). Therefore, most methods measure SOC content and then multiply it with a conversion coefficient to obtain SOM content. This conversion coefficient ranges from 1.72 to 2.0 but is usually set to 1.72, based on the assumption that the C content of SOM amounts to 58% (Baldock and Nelson, 2000). Methods to measure SOC have been reviewed by Islam (2006) and are listed in Table 5.4, including their main merits and limitations.

Assessment of SOM stocks requires, besides measurement of SOM content, the determination of soil bulk density (BD) as well as stone content, both of which vary in space and are associated to different measurement errors (Schrumpf *et al.*, 2011). Typically, SOC stock is quantified to a fixed depth as the product of BD, depth and SOC content (IPCC, 2003). However, as reported by Wendt and Hauser (2013), the fixed depth method may introduce errors in quantifying SOC stocks and differences/changes therein, especially when BD differs between treatments (e.g. minimum tillage increased BD) or changes considerably over a monitoring period. Error propagation analyses performed by Schrumpf *et al.* (2011) showed that BD contributed most to SOC stock variability in upper soil layers of croplands, while SOC concentration was more important in other conditions (e.g. deeper layers). In order to reduce monitoring errors, SOC estimations should be standardized according to equivalent soil masses as explained by Wendt and Hauser (2013). Sampling designs that maximize sensitivity to field-level changes in SOC contents/stocks have been described by different authors (e.g. Conant and Paustian, 2002; Stolbovoy *et al.*, 2007; Schils *et al.*, 2008) but are not reported here for brevity.

While measurement-based inventories yield “transparent estimates” of changes in carbon stocks, they are expensive and require long sampling intervals in order to reduce the uncertainties associated with C data.

SOC models could help reduce the costs of the soil survey methods and, at the same time, allow current and future scenarios to be analyzed. More specifically, models are suitable to simulate the mechanistic effect of anthropogenic (land management) drivers as well as natural (climate and soil) drivers (Lugato *et al.*, 2014), offering the possibility of identifying SOC baselines/thresholds that are not strictly related to the present land use (Kuikman *et al.*, 2012), and of harmonizing SOC data.

Extensively used models include CENTURY (Parton *et al.*, 1987), DNDC (Giltrap *et al.*, 2010), Roth-C (Coleman *et al.*, 1997). Models operates at daily (DNDC) or monthly step (CENTURY and Roth-C), allowing upscaling processes from the plot to field and regional levels. In addition to C and N pools, models (e.g. CENTURY) allow to simulate phosphorous (P) and sulphur (S) cycles. CENTURY was recently selected as the most suitable model for a pan-European SOC assessment for its capability of simulating with a reduced computational time a large number of combinations of cropping systems and management practices (Lugato *et al.*, 2014).

Despite their mechanistic nature, SOC models still need to be thoroughly tested with measured data to compensate for the uncertainties associated with the models' theoretical background. For this purpose, IPCC (2006) suggested establishing a set of benchmark monitoring sites, with a statistically replicated design that captures both anthropogenic and natural drivers.

Table 5.4: Methodologies used for determination of SOC (Islam, 2006; mod.).

Measurement	Basic method	Principle	Merits (+) and limitations (-)
From difference between soil total C and inorganic C	Dry or wet combustions	Total C and inorganic C are determined on separated samples SOC = Total C-Inorganic C	• Reliable results on acid soils (+) but not on soil with high pH (-) • Effective to oxidize resistant organic carbon forms (+) but not carbonates (-) • Fast and precise automated methods(+) • Two separate analyses are required (-)
From determination of soil total C after removal of inorganic C	Dry or wet combustions	Total C is determined after removal of inorganic C with acid pre-treatments	• Total C measured after removing inorganic C is equal to SOC contents, so only one analysis is required (+) • Not very effective for complete removal of inorganic C (-)
From rapid dichromate redox reactions	Wet oxidation of Cr^{+6} with or without (simple heat of dilution) external heating	Amount of oxidized Cr^{+6} of reduced Cr^{+3} species determined in solutions is proportional to SOC; Cr^{+6} is determined by redox titration or by colorimetric measurements (for the latter also Cr^{+3})	• Simple and rapid methods, easy to use for a wide range of soils (+) • Recoveries of SOC by simple heat of dilution are incomplete; a correction factor is required (-) • Errors due to thermal decomposition of Cr^{+6} in case of external heating (-)
From near-infrared reflectance spectrometry	Spectroscopic	SOC is determined from distinct spectral features of H-C bonds in SOM	• Rapid and non-destructive methods (+); spectral responses can be affected by water, inorganic C, total N (-); allows *in-situ* measurements (+)

5.6 Effects decline in SOM in mineral soils on other soil threats

SOM decline has strong implications for other soil threats and, in particular, soil erosion by water and wind, compaction, biodiversity and desertification (Fig. 5.4).

SOM plays a pivotal role on the aggregate stability and cohesion (cf. section 5.7), which in turn affects water erosion. Simple applications of the Universal Soil Loss Equation (USLE) show that peaks in soil erodibility can be observed in soils with clay and SOM contents of around 10-20% and 1-2%, respectively, while an increase in SOM from 2% to 4% halves the predicted soil losses.

SOM also exerts an important control on soil wind erodibility, by influencing the detachment and transport of soil particles. The effect of SOM on wind erosion can be quantified by the soil erodible fraction (EF), which represents the portion of the top 25 mm of the soil that can be transported by wind (Fryrear *et al.*, 1998). An increase in SOM from 1% to 5% decreases the EF from 0.65 to 0.55, and reduces the maximum transport capacity of wind by 15%.

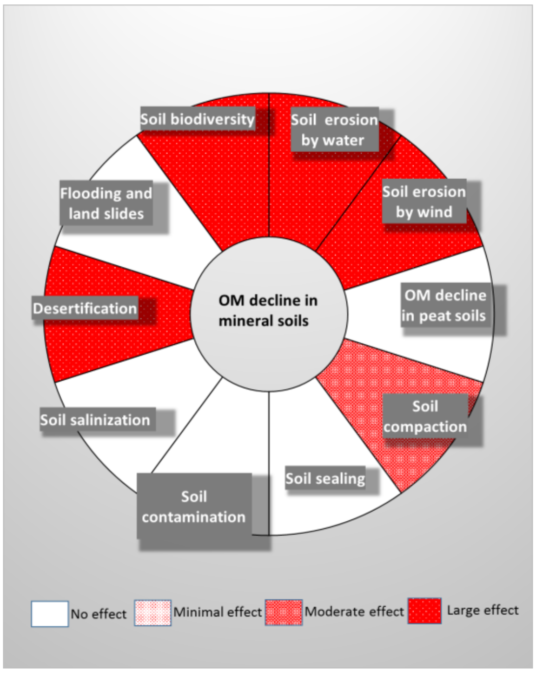

Figure 5.4: Effects of decline in OM for mineral soil on other soil threats. Red is negative.

SOM reduces soil compaction, as it improves the soil structure in terms of total porosity as well as pore size distribution. Dal Ferro *et al.* (2013) proved that SOM has a dual significant effect on the soil structure as it favours the formation of both micro- (<5 µm) and macro-pores (>560 µm). Schjønning *et al.* (2002) observed a significant influence of SOM on soil aeration and drainage capacity, which was associated to the sponge-like pore systems of high-SOM soil. However, the role of SOM in soil compaction should not be overestimated, as compaction mainly concerns the deeper layers where SOM content is usually insufficient to impact positively on soil structure.

There are profound links between SOM and soil biodiversity. Indeed SOM is the main source of energy for the decomposer organisms and an important pool of macronutrients (cf. section 5.7). The ENVASSO project included SOM as an indicator of soil biological functions for its significance, standardised methodology, measurability and costs. However, a unique SOM threshold value that is applicable to all soils is difficult to define, because the activity of soil organisms also depends strongly on other drivers such as pedo-climatic conditions and management (Huber *et al.*, 2008).

SOM exerts a significant control on desertification since, *inter alia*, it increases the water retention capacity and improves the soil structure, as mentioned earlier. For these reasons, the ENVASSO project identified SOC content as an indicator of desertification, even if it was not possible to identify baseline/threshold values.

5.7 Effects of SOM on soil properties and functions

In general, SOM depletion in mineral soils negatively affects the soil function of food and other biomass production. Direct effects are a reduction in the pool of nutrients (i.e. N, P S and micronutrients) and their plant-available forms (e.g. P solubility), a reduction in ion exchange capacity (e.g. CEC), and a reduction in water and nutrient use efficiency due to the higher losses by drainage, evaporation and volatilisation. SOM depletion also negatively influences the biological activity and its complex biogeochemical mechanisms related to biomass production. Additional effects include the reduction of the available water capacity particularly in sandy soils while the impact in fine-textured soils is more ambiguous.

Potential negative effects on crop growth are also related to poor structural conditions (e.g. compaction) observed in SOM-depleted soils. On the one hand, poor soil structure inhibits root growth causing a reduction of rooting volumes and in turn of the amounts of available nutrient and water, and on the other it can negatively affect soil aeration and drainage increasing the risk of excessive soil water. Potential indirect effects can be caused by the interactions between SOM and erosion that may trigger a vicious cycle leading to soil degradation and eventually biomass production decline.

A value of 2% SOC for agricultural soils is often considered the limit below which the soil becomes unstable, more prone to structural deterioration and erosion, and crop yield reduction. In their review, Loveland and Webb (2003) concluded that in the case of soil physical properties there is no evidence for a SOM threshold below which a “catastrophic failure” occurs while for crop nutrition, SOM may be critical where mineralization is the only way to sustain crop yield.
SOM depletion has negative consequences for the soil’s capacity for storage, filtering, buffering and transformation. Reduced storage capacity of energy and nutrients is directly related to depletion of SOM stocks, which represent the most fundamental reserve of metabolic energy and the largest pool of

macronutrients in non-cultivated soils (>95%). SOM depletion can also have an indirect effect on the storage and filtering capacities by affecting the soil hydraulic properties and ultimately the water cycle (infiltration, runoff, etc.). Relationships of hydraulic properties with organic carbon content are influenced by proportions of textural components, but generally, in SOM-depleted soils runoff is favoured at the expense of infiltration and water storage. In addition, the concurrent occurrence of runoff and unstable aggregates increases the risk of sediment loads and nutrients (e.g. P) transport in the rivers (Panagos *et al.*, 2014).

SOM depletion can also have negative consequences for the filtering and buffering capacities by reducing the soil capability of adsorbing and/or biodegrading pollutants. In particular, the surface properties of SOM (high specific surface area, high charge density, etc.) exert a profound influence on the soil surface adsorption properties. Positive effects of SOM are observed in terms of reducing the potential toxicity and transport of heavy metals (e.g. Al^{3+}, Pb^{2+}) and xenobiotics (e.g. PCBs) (Collins *et al.*, 2010). SOM can also act as a reactor for biodegradation of contaminants. Soil transformation capacity is also mediated by SOM interaction with biological activity. SOM depletion could have a negative impact favouring the formation and leaching of nitrate during the mineralization process.

SOM depletion by soil respiration negatively affects the soil capacity to buffer GHGs emissions by increasing CO_2 emissions directly and N_2O indirectly. On the contrary, SOM depletion by accelerated erosion does not necessarily increase emissions of GHGs in the atmosphere (Lal, 2009). Indeed only, a part of the SOC redistributed over the landscape by erosion is mineralized and contributes to CO_2 emissions while the other can be buried and then sequestered. The reverse soil C sink capacity (c.f. sequestration) has also a fundamental role on climate regulation.

SOM plays a crucial role on soil function of biological habitat and gene pool. SOM depletion is usually associated with a lower biological activity and diversity even if it is not still clear if there exists a threshold level of SOM that is required to maintain all the functions of the microbial population. A maximum level of biodiversity is reached when sufficient levels of water and energy and low-to- medium level of nutrients are encountered (Primavesi, 2006). Biodiversity is also related to the nature of organic matter since each type of soil organism favours different substrate and nutrient sources.

SOM depletion has negligible effects on soil functions of “physical heritage”, “platform for man-made structures” and “provision of raw materials”.

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6 SOIL COMPACTION

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6.1 *Description of soil compaction*

Soil compaction is defined as: "The densification and distortion of soil by which total and air-filled porosity are reduced, causing deterioration or loss of one or more soil functions" (van den Akker, 2008). The term compactness is sometimes used for the resulting density state of the soil following compaction. Compaction takes place when soils are subjected to stresses that exceed its strength.

Within agriculture, compaction of soil is induced by trampling of animals and by agricultural machinery and this affects nearly all soil functions. In an isotropic stress field, soil compaction primarily reduces the size and volume of pores, while their continuity and tortuosity are unaffected. However, any point in the soil beneath a loaded surface is subject to a complex stress field. This may induce distortion of the soil pores, which can have significant impacts on soil functions (e.g. Berisso *et al.*, 2013). Often, however, only the vertical stress component and vertical strains are considered in models developed for practical applications. It is beyond the scope of this chapter to go into detail with the more sophisticated aspects of the stress-strain dynamics in a complicated stress field.

The driving force behind the compaction problem is the efforts of farming to remain economically viable in a society where salaries are generally high. Reduction of the workforce involved in farming operation requires larger and more efficient machinery. The pressures on the soil system are increased by frequent traffic and/or heavy machinery.

Compaction of the subsoil is especially a stealthy evil because it is invisible, cumulative and persistent (Håkansson and Reeder, 1994; Horn *et al.*, 1995). Sustainable Land Management (SLM) options that avoid subsoil compaction are crucial to secure the continued delivery of soil ecosystem services for generations to come. The non-transparent nature of the compaction damage to the subsoil implies that farmers are not always fully aware of the threat. This calls for a combination of scientific knowledge and stakeholder experience to identify SLMs capable of meeting the challenge.

Compaction of the subsoil has proven to be particulary persistent (see section "Effects of soil compaction on soil properties and functions"), while drying-wetting, freeze-thaw, the action of soil biota, and tillage greatly assist in the alleviation of compaction damages of topsoil layers. Compaction of the topsoil though, has a significant impact on crop yield, and residual effects may last for some years, especially for clay soils. The persistent nature of subsoil compaction calls for society concern and potential regulation, which is the reason for focusing on subsoils in this text. For soils in agricultural use, subsoil is defined here as the layers below the tillage depth. For ploughed soils, this is often approximately 0.25 m. For non-tilled soil including forest soils, subsoil may be defined as soil not significantly affected by the above-mentioned natural amelioration processes, in effect also a couple of decimeters.

6.2 *State of soil compaction*

Quantification of soil physical properties is laborious, especially for the subsoil. Hence, there are only few thorough inventories based on measured indicators, and these only cover regional areas. In a global context, Oldeman *et al.* (1991) estimated a total of ~68 million hectares to be affected by compaction but this figure has probably increased considerably today. Although more than ten times this area was estimated to be affected by erosion (Oldeman *et al.*, 1991), the persistency of subsoil compaction classifies it as one of the most important threats to soil quality (Håkansson and Reeder, 1994). Batjes (2001) evaluated soil degradation in central and Eastern Europe. He estimated that around 11% of the study area was affected by compaction, ~68% of this moderately or strongly. According to Van den Akker and Hoogland (2011) about 50% of the most productive and fertile soils of the Netherlands have compacted subsoils. Recently, Widmer (2013) produced an inventory showing that around one third of the agricultural area in Central Switzerland has critically high densities.

The SPADE8 soil database (Koue *et al.*, 2008) is a further development of the SPADE1 database initiated in 1992 (Breuning-Madsen and Jones, 1995). The SPADE database was constructed to support the EU-soil map at scale 1:1,000,000 with soil analytical data for modelling purposes. The SPADE8 database includes a range of soil properties for a total of approximately 900 soil profiles (~3500 soil horizons) across 28 countries in

Europe. Soil texture, organic matter and bulk density are estimated by soil experts in the different countries. We calculated the Relative Normalized Density *RND* (see section *Indicators*) for an average of subsoil horizons in each of the SPADE8 database soil profiles / mapping units (Figure 6.1). White areas on the map indicate that not all mapping units in the data base have been given values of both bulk density and clay content, which is a prerequisite for calculating *RND*. Excluding organic soils (organic matter >10%) and addressing only subsoil horizons covering the depth interval 0.25–0.7 m, it was found that ~29% of a total of 692 SPADE8 profiles displayed *RND* values above 1, indicating critically high densities. This corresponds to about 23% of the total area characterized.

A similar analysis as the above has been done for the Danish Soil Data Base (Breuning-Madsen and Jensen, 1985), which includes measured values of bulk density and clay for >4800 soil horizons deriving from 1292 soil profiles. If excluding organic soils (organic matter >10%) and topsoil horizons (i.e., considering only horizons including depths 0.25 – 0.7 m), it was found that ~39% of the profiles in the data base had critically

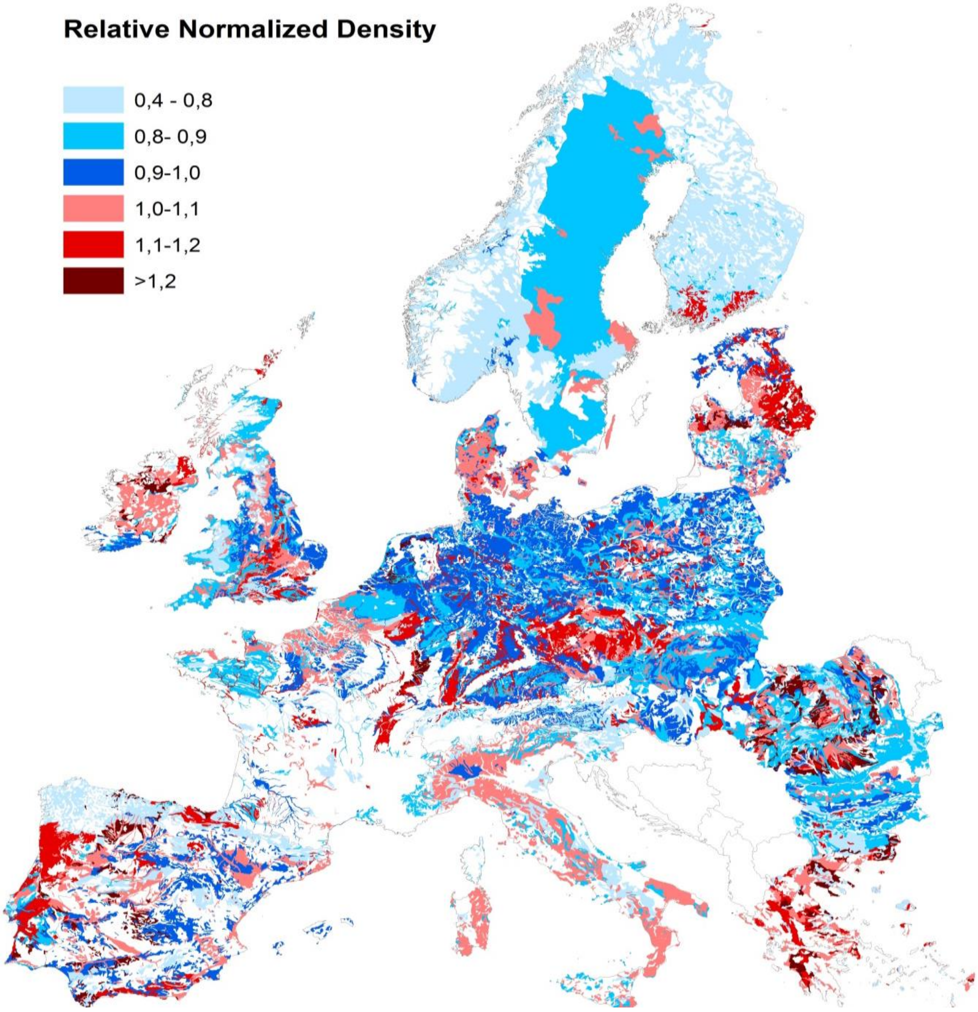

Figure 6.1: Relative normalized density (RND) for European subsoil horizons covering the depth 0.25 – 0.7 m as calculated by Eqs. 3ab based on the SPADE8 database (Koue et al., 2008). RND>1 may be considered a dense soil.

high densities (*RND*>1). The profiles were approximately equally distributed across geo-regions over the entire country (not shown). The data thus indicate that approximately 39% of the Danish agricultural soils have critically high densities in the upper subsoil.

The SPADE8 data set is based on expert judgement and is thus sensitive to subjectivity. Nevertheless, the map in Figure 6.1 is currently, the best possible illustration of European regions with high-density subsoils. According to the analysis, below-critical densities are found in large parts of Central Europe, while above-critical areas are found in parts of the Baltic area, in Denmark, in and around the former Czechoslovakia, in northern Portugal, in Italy, and in parts of the United Kingdom. It is interesting to note that the boundary between glacial tills (typically loamy soils) and alluvial deposits (often sandy material) normally very distinct on pedological maps is not detectable to the same degree with respect to the *RND*-parameter (e.g., Denmark). This is a reflection of the normalization included in the *RND* term, where less dense tills are equally critical to soil functions as higher-density sands.

The above exercise should be regarded a first approach for assessing the extent of machine-induced compaction of subsoil layers across Europe. We note a discrepancy between *RND* values estimated from the SPADE8 data set (Figure 6.1) and measured density data in the Netherlands (van den Akker and Hoogland, 2011). This calls for a 'calibration' of the expert-guesstimated SPADE8 data to measured data for as many regions as possible. We also see an urgent need to extend the SPADE8 data base to cover all Europe. We further encourage additional studies for the support or modification of the most relevant level of threshold density in the *RND* parameter.

6.3 Drivers/pressures

The driver for mechanization in agriculture is the need to replace expensive man-power with efficient and hence cost-effective machinery. In the developed and industrialized countries, modern agriculture is characterized by an intense mechanization that allows for production of food at affordable prices. A massive movement of the labour force away from agriculture has occurred in recent decades. To maintain an income that is comparable to the rest of society, farmers are forced to make every part of the production as efficient and cost-effective as possible. A key driver for the mechanization is the obligation to pay salaries comparable to those in other societal sectors. Hence, the advent of larger and more technologically advanced machinery for field operations, where a single person can manage a large area in a short time.

The pressure with respect to soil compaction is caused by (frequent) traffic with heavy machinery. The success in agricultural engineering has enabled extremely high-capacity field operations. As an example, early combine harvesters designed around 1960 processed about 4 Mg small-grain cereals per hour, while the same figure for modern combines 50 years later is ~40 Mg. Thus, the capacity has increased by a factor 10 in a 50-year period (personal communication, S. Trampedach, April 2014).

An important side-effect of the development described above is a significant increase in the weight of the machines travelling on our fields. That is, the pressures on soils have increased also literally speaking. For example, Vermeulen *et al.* (2013) estimated that on average wheel loads in slurry application in the Netherlands have increased from about 3.5 Mg in 1980 to 5.6 Mg in 2010. Historical data on combine harvesters (http://www.dronningborg.de/) similarly confirm that the weight of the fully-loaded machines has increased by a factor of ~6, from ~4.3 Mg in 1958 to ~25 Mg in 2009 on average. The largest combine harvesters on the market today (mid 2014) are even heavier than this. The weight of machinery for harvesting sugar beets and potatoes, and for application of slurry, can exceed 50 Mg.

Vermeulen *et al.* (2013) predicted the vertical stress reaching different depths of the soil profile when trafficked with the machinery typically used in the Netherlands in 1980 and 2010 (Figure 6.2). The SOCOMO model (SOil COmpaction MOdel; Van den Akker, 2004) was used to calculate the stresses at depths of 0.25 and 0.5 m. Calculations included the use of rubber tracks when harvesting sugar beet and potatoes as well as driving in the open furrow during ploughing. For the simulation of stresses below tracks, the stresses underneath rollers and wheels were assumed to be uniform, while a zero stress level was assumed for the areas in between rollers and wheels. The results in Figure 6.2 indicate a general trend in the 30 years of considerably increased vertical soil stresses at both 0.25 and 0.5 m depth. The improvement of tyres during this 30-year period has not been able to counterbalance the increase in wheel loads, where the estimated stresses even at 0.25 m depth are often higher in 2010 than in 1980 (Figure 6.2). Driving directly on the subsoil in the open furrow during ploughing proves to be very destructive, also at a depth of 0.5 m. The use of tracks seems to be an improvement, at least in these calculations. However, more field measurements are needed to confirm these findings. Uneven stress distribution in the contact area for tracks implies that the potential benefit from the larger contact area for tracks as compared to tyres is not (always) achieved: the stresses reaching the upper layers of the subsoil are not significantly reduced as compared to tyres (e.g., Arvidsson *et al.*, 2011).

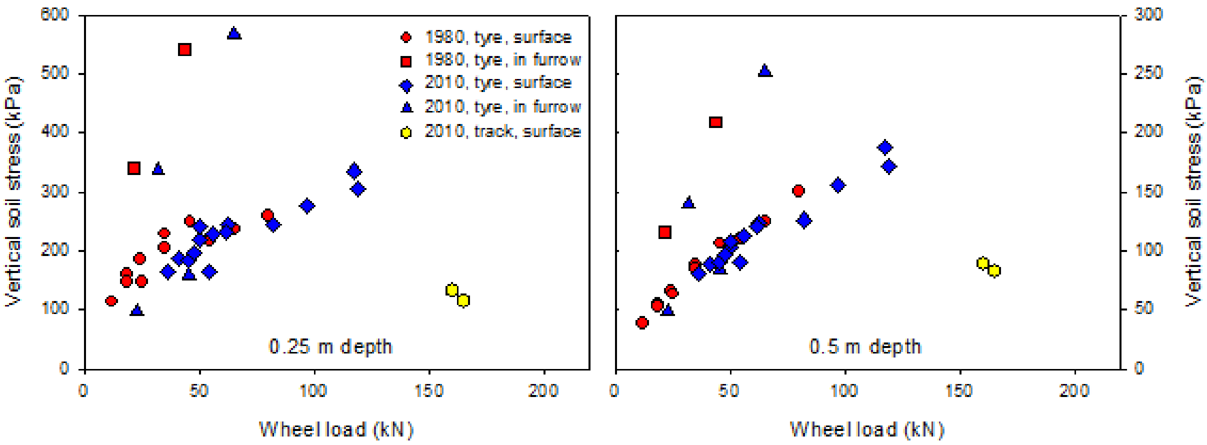

Figure 6.2: Increase of wheel loads in the period 1980 – 2010 and corresponding resulting vertical soil stresses calculated by the SOCOMO model at a depth of 0.25 and 0.5 m. Note different stress scales for the two depths. Redrawn from Vermeulen et al. (2013).

Schjønning *et al.* (2015) did a similar simulation of the stresses exerted on the soil from machinery. They compared nine combine harvesters produced in the period 1958-2009. The stress propagation in the soil profile was predicted with a model in principle following the previously mentioned SOCOMO model. The specific simulations were run in the integrated model complex Terranimo® (Lassen *et al.*, 2013; www.terranimo.dk). The wheel load of tested machines increased by a factor of ~6 during the 50-year period considered, the volume of tyres increased by even more (a factor of ~12), while the tyre-soil contact area increased by a factor of only ~3.5 (Schjønning *et al.*, 2015), effectively indicating an increased pressure exerted on the (sub)soil. They also found that the vertical stress increased significantly for the 0.25, 0.5, 0.75 and 1 m depths, by factors of 1.9, 3.0, 3.9 and 4.6, respectively (analyses not shown).

The above analyses document that the vertical soil stresses induced by commonly used agricultural machinery have increased for all depths of the soil profile during the period of mechanization in agriculture,- in effect primarily since World War II. The use of wider and more voluminous tyres has not been able to counteract the increase in loads of the machinery. The simulations accord with measured data as reviewed by Hallett *et al.* (2012), who noted the significant implications this has for mitigation measures for soil compaction.

There is clearly some difference among European countries with respect to the size of machinery used for some field operations. In Germany slurry application with a tractor-trailer combination is estimated to lead to wheel loads of typically ~42 kN when using the largest machinery available (VDI, 2007). In Denmark, wheel loads for similar field work often exceed 60 kN. The difference among European countries seems to be less significant for a range of other field operations, for example the harvesting of sugar beet and potatoes. Here, very high loads are generally imposed on the soil. Self-propelled sugar beet harvesters may reach wheel loads of more than 12 Mg when the hopper is full (e.g., VDI, 2007). In potato harvesting, wheel loads may become as high as 15 Mg (VDI, 2007). The trend is now for these machines to be fitted with tracks rather than tyres.

For some eastern European countries and also 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.

The above discussion of drivers and pressures explaining the development in the soil compaction threat only relates to the 'disturbing agent' (the machinery exerting mechanical stresses to the soil, with no focus on the soil) and the 'system' threathened (OECD, 2003; Schjønning *et al.*, 2015). However, climate changes may also be regarded a driver for soil compaction. This is because soils' ability to withstand the mechanical stresses is decreases with increase in soil water content (e.g. Arvidsson *et al.*, 2003). Scenarios indicate significant change in the amount and pattern in 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. However, 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.*, 2000).

6.4 Key indicators of soil compaction

The EU-funded ENVASSO project arrived at two prioritized indicators for soil compaction: 1) soil density, and 2) the air-filled pore space when drained to a matric potential in the range 30-60 hPa (van den Akker, 2008). Strictly speaking, the indicators would rather reflect compactness than compaction, which by definition is a process and not a state. Ignoring this semantics, the use of such simple indicators is appealing because they are rather easily measured and easily understood. The density, however, ought to be modified by the soils' content of mineral fines to give a relative expression of density in order to lend itself to comparisons among different soil types. The modification may be accomplished by the following procedure.

The packing density (*PD*) of soil was introduced by Renger (1970) as a texture-modified expression of density:

$$PD = Db + 0.0009 \cdot Clay \qquad [6.1]$$

where *PD* is the packing density (g cm^{-3}), *Db* is the dry bulk density (g cm^{-3}) and *Clay* is the soil content of mineral fines <2 µm (kg 100 kg^{-1}). Three classes of *PD* were suggested for rating the soils: low: *PD*<1.4, intermediate: 1.4<*PD*<1.75, and high: *PD*>1.75. We note that *PD* is not a density in physical terms but rather an index that allows a comparison of the state of compactness across differently textured soils. Also Heinonen (1960) found that the 'natural' density of a soil decreases with increase in clay content. A study on density effects on soil rootability has later confirmed that a density modified in an analogue way (using fines < 60 µm) explained plant response better than the 'raw' bulk density (Pabin *et al.*, 1998). This suggests that *PD* is not only normalizing density to something 'natural' for a given textural composition but also that *PD* reflects some threshold for biotic activity across soil types.

The Danish Soil Data Base (Breuning-Madsen and Jensen, 1985) was consulted, and it was found that the bulk density of subsoil layers decreased with increasing clay contents for clay contents higher than ~15% (analysis not shown). This generally agrees with results by Keller and Håkansson (2010), who found the so-called reference bulk density (Håkansson, 1990) to decrease for high clay contents. Van den Akker and Hoogland (2011) presented measured data for bulk density that displayed a trend with clay contents very close to the one mentioned above for the Danish Soil Data Base.

The *PD* term has been suggested as a soil-type independent measure of density in a soil compaction context (Jones *et al.*, 2003). More research is necessary regarding the relationship between PD and soil functions. The threshold bulk density equivalent to *PD*=1.75 g cm^{-3} is given by:

$$\sigma_{critical} = 1.75 - 0.0009 \cdot Clay \qquad [6.2]$$

For sandy soils, this yields $\sigma_{critical}$ close to 1.75 g cm^{-3}. This is quite high judged from general experience. Van den Akker and Hoogland (2011) adapted Renger's (1970) suggestion of bulk densities corresponding to *PD* values higher than 1.75 g cm^{-3} as critical for soils with clay contents above 0.167 kg kg^{-1}. For sandy and loamy soils they suggested a clay-content-independent constant critical bulk density of 1.6 g cm^{-3} as based on experience in the Netherlands. The idea of van den Akker and Hoogland (2011) is adapted and the term, Relative Normalized Density (*RND*), introduced:

$$Clay\ content < 16.7\,\%\frac{w}{w}: \quad RND = \frac{\sigma}{\sigma_{critical}} = \sigma/1.6 \qquad [6.3a]$$

$$Clay\ content < 16.7\,\%\frac{w}{w}: \quad RND = \frac{\sigma}{\sigma_{critical}} = \sigma/(1.75 - 0.0009 \cdot Clay) \qquad [6.3b]$$

where σ is actual bulk density (g cm^{-3}) and *Clay* is clay content in % w/w. The parameter *RND* is the physically-based ratio between the actual and the threshold value of bulk density. Van den Akker and Hoogland (2011) labeled the ratio as "degree of overcompaction".

6.5 Methods to assess status of soil compaction

Quantification of the soil density indicator can be done by sampling of undisturbed soil samples or by indirect methodologies (e.g., gamma-ray (density) and penetration resistance measurements, Schjønning and Rasmussen, 1994). The volume fraction of air-filled pores in addition requires measurements of soil water content at matric potentials in the range -30 to -60 hPa.

The state of soil compaction may be assessed by surveys of any specific region in question, where the indicators then are normalized to natural (uncompacted) threshold values as explained in the former section for the density parameter. The normalization may alternatively use some threshold indicator values identified as being critical to selected soil functions (e.g. Lebert *et al.*, 2007). We emphasize the risk of overlooking important ecosystem services embedded in this latter approach.

Studies of virgin areas (parks, gardens etc) not subject to wheel traffic as compared to nearby arable soil exposed to the effect of agricultural machinery have been used as an alternative assessment method (e.g., Håkansson *et al.*, 1996). An extrapolation of the state of soil compaction can then be performed to other areas with similar traffic history.

6.6 Effects of soil compaction on other soil threats

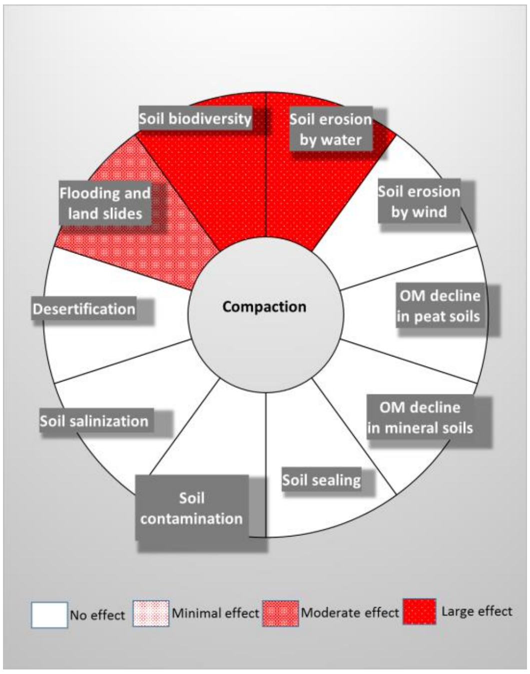

Figure 6.3: Effects of compaction on other soil threats. Red is negative effect.

The decrease in water conductivity from soil compaction as discussed in the following section may induce surface runoff of water. This, in turn, may carry pollutants and nutrients directly to surface waters. Also, surface runoff may trigger sheet as well as gully erosion. In extreme precipitation events, this may even trigger landslides, and surface runoff may contribute to flooding. Finally, a dense soil is non-optimal to a variety of soil organisms. Compaction may thus also reduce soil biodiversity.

6.7 Effects of soil compaction on soil functions

Most soil functions (and derived ecosystem services) relate to the characteristics of the soil pore system. Soil pores, in turn, are strongly affected by soil compaction. Soil pore properties are thus the key indicators for the compaction effect on ecosystem services. Berisso *et al.* (2012) showed that compaction affected the pore system and its functions to a depth of 0.9 m of a loamy till soil in southern Sweden (Figure 6.4). Experimental plots that had received no experimental traffic were used as controls, while plots subjected to four repeated wheelings (track-by-track to cover 100% of the area in the plots) with a 35 Mg sugar beet harvester 14 years prior to the investigation comprised the compacted plots. The compaction effect of this one-event traffic had persisted for 14 years with respect to the pore volume (Figure 6.4) as well as to the conditions for gas exchange by diffusion and convection (Figure 6.4). Studies on a clay soil in southern Finland demonstrated persistent compaction effects on the soil pore system to 0.5 m depth for three decades (Schjønning *et al.*, 2013). The compaction impact on the soil pores may seriously affect important soil functions like soil aeration and soil water transport.

Given the above basic impacts of soil compaction on soil properties, the effects on key soil functions can be described as follows:

(i) Food and other biomass production

Compaction affects root growth and the conditions for transport of water and gases in the soil. This, in turn, will affect a range of important functions that are crucial to crop production. Reduced root proliferation hinders crop exploitation of water and nutrients in the soil profile. Constraints in soil aeration during wet periods are similarly non-optimal to plants. As a result, a yield penalty is often observed for compacted soil. Hallett *et al.* (2012) made an inventory of observed compaction-induced yield decreases across geographical regions, soil types and crop species. The yield penalty ranged from nearly half the production of uncompacted land to positive effects from compaction. This reflects the need to focus the effects – and hence the degree of compaction – in the topsoil and the subsoil.

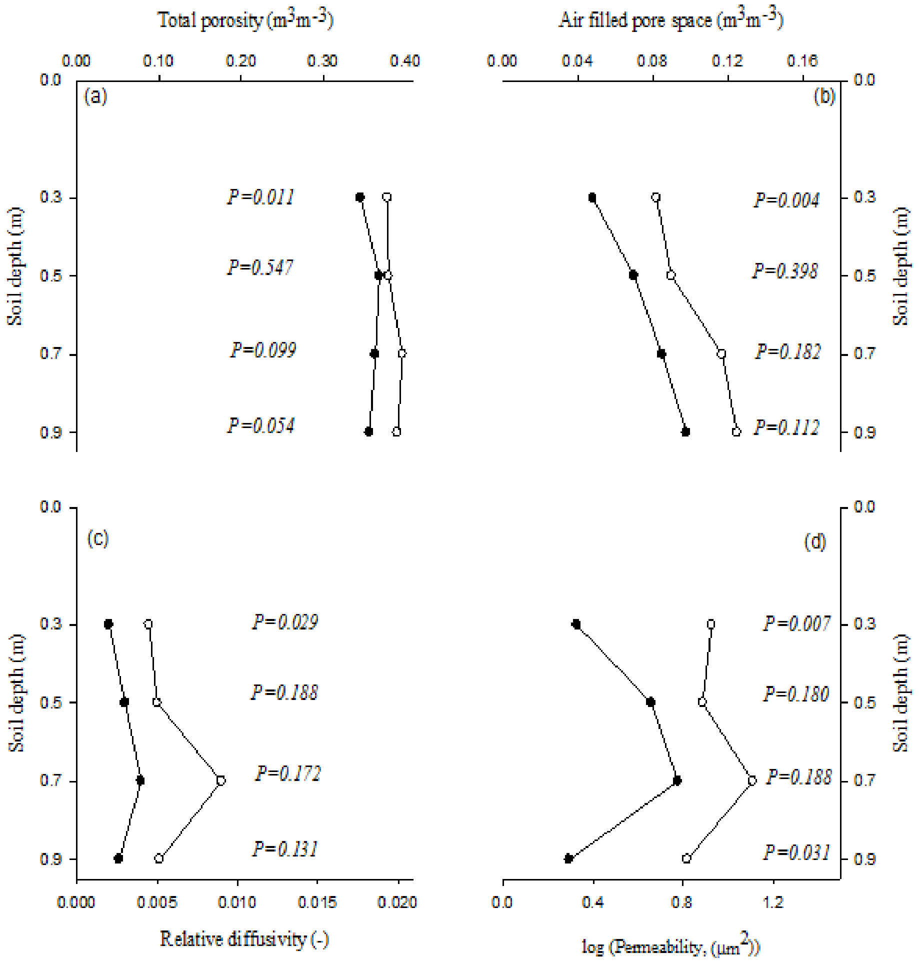

Figure 6.4: Compaction effects from four-times replicated traffic with a 35 Mg sugar beet harvester on soil properties persisting for 14 years in a loamy soil at southern Sweden. (a) Total porosity, (b) air-filled pore space, (c) relative gas diffusivity and (d) air permeability measured at -100 hPa water potential for compacted (shaded circles) and control treatments (open circles). P-values give probability of differences between control and compacted treatments being random. Reproduced from Berisso et al. (2012).

A series of long-term field experiments in northern Europe and North America initiated in the 1980s focused the penalty in soil productivity by applying a one-event compaction with loads of 10 Mg on single axles or 16 Mg on tandem axle units (Håkansson and Reeder, 1994). After the initial track-by-track treatments, all plots in all experiments were treated identically and with no heavy traffic. Annual ploughing to a depth of 0.20-0.25 m was performed in order to alleviate the compaction effects in the plough layer as quickly as possible. Generally, yields were significantly reduced the first year(s) following the experimental treatment. Then gradually, the yield reduction declined leaving on average a significant, residual (persistent) 2.5% yield reduction for a treatment with four passes as compared to control plots (Figure 6.5, left). Håkansson and Reeder (1994) ascribed the yield reductions to a plough layer effect (a), an effect from compaction of the 0.25-0.4 m layer (b) and, finally, an effect attributed to compaction of the soil at >0.4 m depth (Figure 6.5, right). Given the persistent effects of compaction also at 0.3 m depth documented 14 years following a one-event wheeling (Berisso *et al.*, 2012), this interpretation can be disputed. Rather, the permanent yield penalties probably relate to persistent structural changes of all the subsoil rather than just the horizons deeper than 0.4 m.

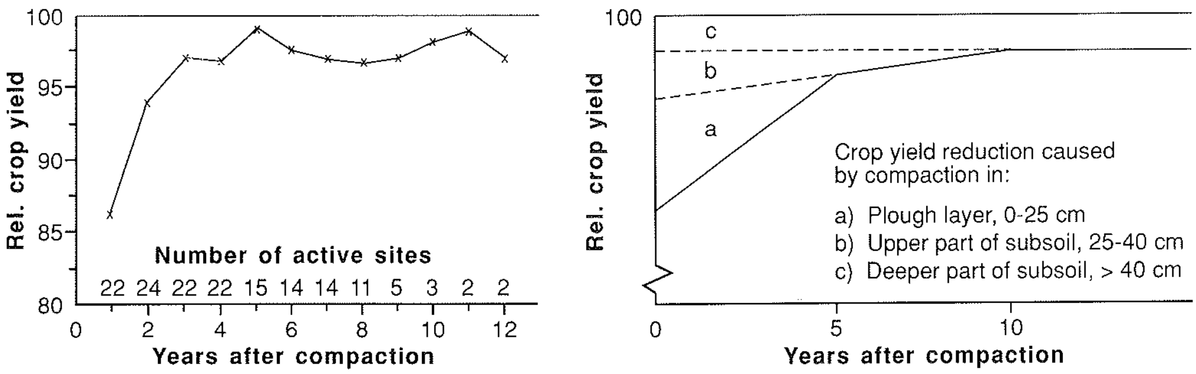

Figure 6.5: The results from a comprehensive international series of field trials with one initial soil compaction event (~5 Mg wheel load, four passes wheel by wheel). The Figures show the development in time of the relative crop yield in exact figures (left) and interpreted in relation to the compaction effect of different soil layers (right). Reproduced from Håkansson and Reeder (1994).

Only a few studies have quantified the effects of higher wheel loads (often used today) on crop yields. Contrasting results have been reported but persistent annual reductions up to 12% have been found, apparently higher for clay soils than lighter soils. The potential impact of subsoil compaction on crop yield may be much more severe than can be deduced from average results of even long-term field trials. Alakukku (2000) found that with subsoil compaction for a clayey soil, wet growing seasons led to higher yield reductions than dry seasons. Alblas *et al.* (1994) found yield reductions of silage maize up to 38% in a dry year for a sandy soil, indicating that subsoil compaction effects can also be serious for lighter soils. The observed variation in compaction effects may relate to the weather conditions. Compaction-induced poor drainage may reduce the number of workable days in the field, which, in turn, may affect soil trafficability, friability and growing conditions. It may also cause problems for harvests if the period is quite wet. The potential complete loss of a year's crop adds significantly to the average effect of compaction. Precipitation is expected to increase in northern Europe due to climate change, which may significantly exacerbate the compaction problem on food and other biomass production.

(ii) Storage, filtering, buffering and transformation

Emission of greenhouse gases: Compaction reduces aeration of the soil matrix between the vertical macropores and increases the risk of anaerobic conditions. Denitrification is a potential undesired side-effect of compaction where plant-available nitrogen is removed from the soil and potentially adds to the atmospheric concentration of the potent greenhouse gas (GHG) nitrous oxide (N_2O). Several studies have clearly shown significant increases in N_2O emission following compaction of the topsoil (e.g., Simojoki *et al.*, 1991; Ball *et al.*, 2008). Teepe *et al.* (2004) found that N_2O emission rates increased by as much as 40-fold when trafficking a forest soil with 2 Mg wheel loads at 250 kPa inflation. These experimental conditions indicate non-critical mechanical stresses reaching subsoils (applying the "8-8 rule", Schjønning *et al.*, 2012). The increased emission was thus probably due to denitrification in the topsoil layers. In addition, compaction of topsoil layers may change a soil from being a net sink to becoming a source of the greenhouse gas namely CH_4 (Ruser *et al.*, 1998; Teepe *et al.*, 2004). Based on the above and other sources (e.g., Hallett *et al.*, 2012), there is no doubt that compaction of the topsoil layers may significantly increase the emission of greenhouse gases from soil. However, it is less clear whether the (persistent) compaction effects on subsoil layers contribute to the GHG emission. Schjønning *et al.* (2015) reviewed the literature but only found indications that subsoil compaction contributes to greenhouse gas emission.

Filtering for contaminants: Compaction is known to decrease the saturated hydraulic conductivity, K_s, which governs the drainage of saturated soils (e.g., Etana *et al.*, 2013). However, K_s is not a determinant for one of the most important hydraulic functions of soils: the filtering of solutes and pollutants present in surface waters. When water flows in unsaturated soil, solutes in surface water will be distributed by diffusive processes to the whole soil matrix. In contrast, when the drainage demand exceeds the unsaturated hydraulic conductivity, K_{unsat}, water may bypass the soil matrix and quickly move to deeper soil layers through vertical (bio) pores. This may be a governing process for transport of P, pesticides, pathogens and even soil colloids. Iversen *et al.* (2011) analysed a comprehensive data set for conservative soil characteristics determining the near-saturated hydraulic conductivity (K_{unsat}), and K_s. Their results indicate that compaction is likely to decrease K_{unsat} and hence increase the risk of preferential flow. This is supported by leaching experiments focusing the degree of preferential flow for soil cores with varying soil pore systems (see review in Schjønning *et al.*, 2015).

A few field studies have demonstrated compaction effects on preferential flow (Etana *et al.*, 2013; Kulli *et al.*, 2003). The latter study showed that sprinkler irrigation on soil compacted by multiple passes of a sugar beet harvester resulted in surface ponding and strong non-equilibrium solute transport into the subsoil, primarily through earthworm burrows. Worm channels were also observed in the control plot, but the more widely distributed finer macropore system, which had been degraded in the trafficked plot, soaked up most of the applied water without ponding, and preferential flow was much less pronounced.

To sum up, the results discussed above indicate that compaction affect the rate and flow paths of water movement in the soil profile and hence the soil filter function. However, more studies are needed to strengthen the empirical basis for this interpretation for a wider range of conditions.

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7. SOIL SEALING

Grzegorz Siebielec, Prokop Gundula, Hedwig van Delden, Simone Verzandvoort, Tomasz Miturski, Artur Lopatka

7.1 Description of soil sealing

Soil sealing can be defined as the destruction or covering of soils by buildings, constructions and layers of completely or partly impermeable artificial material (asphalt, concrete, etc.). It is the most intense form of land take and is essentially an irreversible process (Huber *et al.*, 2008; Prokop *et al.*, 2011). There are other terms commonly used to describe relationships between urbanisation processes and soil or land use changes. Land take is also known as "urbanisation" or "increase in artificial surfaces" and represents an increase of settlement areas (or artificial surfaces) over time, usually at the expense of rural areas. This process can result in an increase of scattered settlements in rural regions or in an expansion of urban areas around an urban nucleus (urban sprawl). A clear distinction is usually difficult (Prokop *et al.*, 2011). Settlement areas are also known as “urban land” and “built-up land” and include areas for housing, industrial and commercial activities, areas for health care, educational infrastructure, traffic areas (streets and railways), cemeteries, recreational areas (parks and sports grounds), and dump sites. In local land use plans this category usually corresponds to all land uses beyond agriculture, nature, forests, and water courses (Prokop *et al.*, 2011). The term “artificial surfaces” is used in the CORINE Land Cover nomenclature and refers to “continuous and discontinuous urban fabric (housing areas), industrial, commercial and transport units, road and rail networks, dump sites and extraction sites, but also green urban areas (Prokop *et al.*, 2011).

Figure 7.1 left shows a typical suburban pattern, with houses, gardens, drive ways and yards. This pattern corresponds to the term “settlement area” or “artificial surface”. On the right side the sealed soil of the same settlement area is shown as hatched pattern. In this case about 60% of the settlement area is actually sealed by buildings and streets.

Figure 7.1: Visualisation of the terms “settlement area”/"artificial surface" and “sealed soil”. Source: Prokop et al. 2011

Urban sprawl is commonly used to describe physically expanding urban areas. The EEA has described sprawl as the physical pattern of low-density expansion of large urban areas, under market conditions, mainly into the surrounding agricultural areas. Sprawl is the leading edge of urban growth and implies little planning control of land subdivision. Development is patchy, scattered and strung out, with a tendency for discontinuity (EAA, 2006).

Urbanisation of rural areas is not necessarily linked to an urban nucleus and is understood as an increase of scattered settlement patterns with low population density (dispersed urban development) (Prokop *et al.*, 2011).

Brownfield redevelopment can be defined as land that has previously been developed, but which is not in current active use or is available for re-development. Recycling of brownfields instead of developing on greenfield reduces land take and further soil sealing (Huber *et al.*, 2008).

7.2 State of soil sealing

According to the EEA soil sealing map (EAA, 2013) 2.3% of the European Union's territory were actually sealed in 2006, and 4.4% of the territory were subject to artificial surface formation (Prokop *et al.*, 2011). Figure 7.2 shows the percentage of sealed area aggregated to NUTS 3 region based on the EAA soil sealing map. In the European Union, on average 51% of artificial surfaces are sealed, but this share varies considerably among Member States, depending on dominant settlement structures and the intensity of the interpretation of artificial surfaces (Prokop *et al.*, 2011).

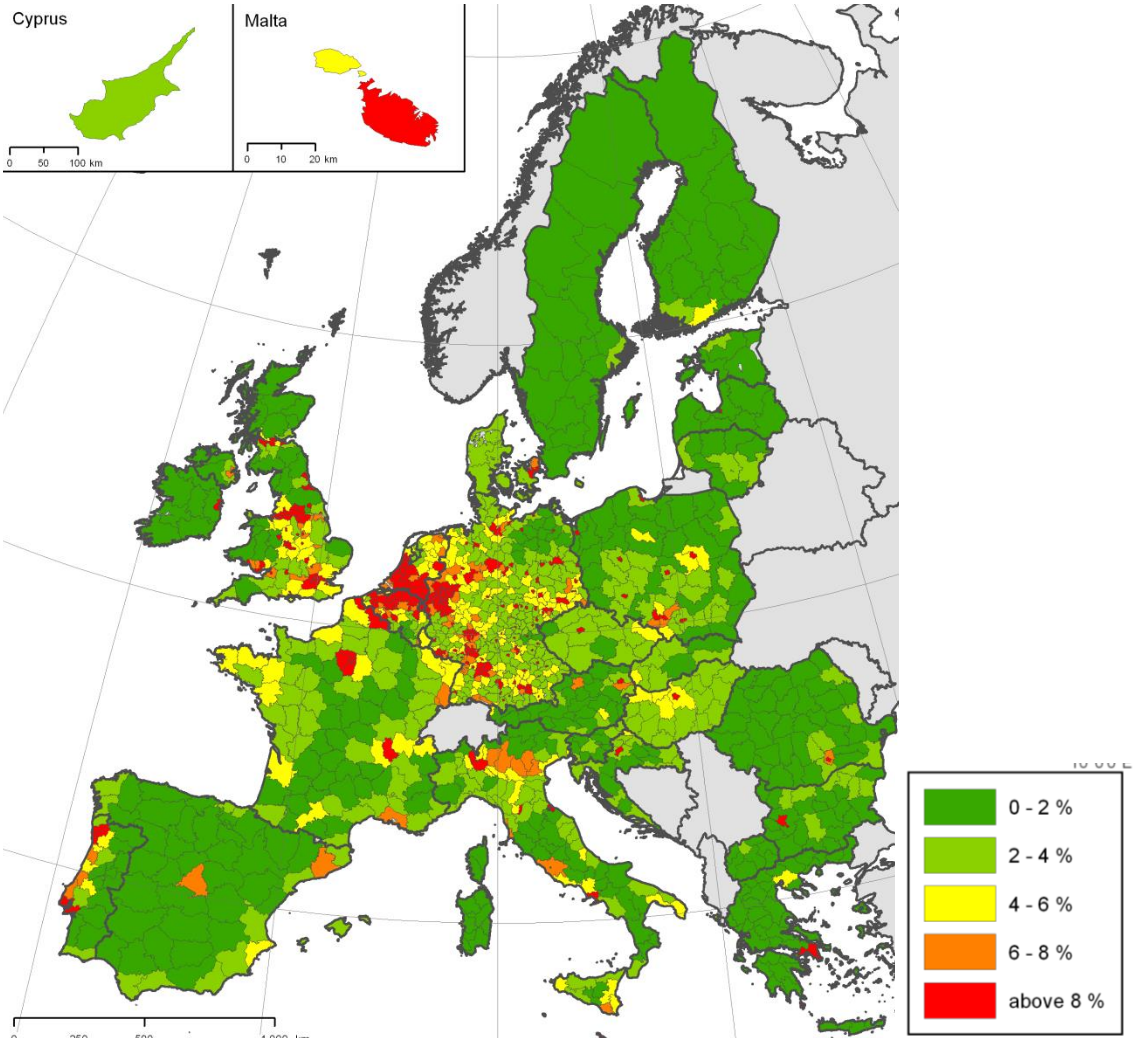

Figure 7.2: Percentage of soil sealing according to EAA soil sealing layer, year 2006 (Prokop et al., 2011).

According to CORINE land cover spatial databases artificial areas covered 4.1%, 4.3% and 4.4% of the EU territory in 1990, 2000 and 2006, respectively. This corresponds to an 8.8% increase of artificial surface in the EU between 1990 and 2006. In the same period, population increased by only 5%. In 2006 each EU citizen disposed of 389 m² of artificial surfaces, which is 3.8% or 15 m² more compared to 1990 (Prokop *et al.*, 2011).

Land take rate data at European level can be gathered from CORINE land cover spatial databases. It must be noted that some inconsistencies exist in these data layers related to non-homogenous methodologies applied in the countries. However, between 1990 and 2006 a slight decrease of annual land take can be observed from 100.640 hectares in the period 1990 – 2000 to 92.016 hectares in the period 2000 to 2006. In Belgium, the Czech Republic, Germany, Luxembourg, Poland, and Slovakia significant decreases of annual land take with more than 25% decrease can be observed and more moderate decreases with less than 25% are visible in Ireland, Italy, Latvia, the Netherlands, and Portugal. In all other Member States average annual land take clearly increased after the turn of the century (Prokop *et al.*, 2011).

A more precise analysis of increase in highly sealed artificial surfaces (continuous residential area, commercial/industrial area and transport facilities) was performed for selected cities of Central Europe (Siebielec *et al.*, 2010). The work aimed to assess the quality of soils lost due to urban sprawl. The study developed land use change maps based on consistent satellite image data, analysed land use change trends in a 15-year period, and subsequently assessed soil quality affected by urbanization. The analysis was performed for Bratislava, Prague, Vienna, Stuttgart, Milan, Salzburg and Wroclaw. Land use maps of 10-meter resolution were produced for the periods 1990-1992 and 2006-2007. The area sealed within the 15-year period in the test areas ranged from 160 to 780 ha. The data provided here may be somewhat different from the official statistics that use different methodologies. However, the advantage of the applied approach was that it enabled an analysis of spatial trends of land use change utilising the same method for all cities, and their linkage to soil quality information. Soils in each city were classified according to available information, usually based on the national classifications of soil/land productivity for agricultural use or urban ecosystem services. It was evident that the best soils were efficiently protected in Bratislava. The proportion of best soils in newly urbanized areas in Bratislava was 5 times smaller than their proportion in total area. It was assumed that the regulations present in Slovakia helped to protect the most valuable soils in terms of productivity potential for agriculture. The soils classified as most valuable in the assessment are protected by a fee payment system (1-4 classes from total 9). The transformation of these agricultural lands into other land use types is charged with obligatory payments ranging from 6 to 15 EUR per square meter. A similar system existed in Poland until 2008. However, this practice did not ensure the efficient protection of the most valuable soils in Wroclaw. The assessments performed for Wroclaw, Prague, Vienna and Salzburg revealed negative trends of preferential use of the most valuable soils for agricultural use, whereas in Stuttgart and Milan the conversion of high quality soils for agriculture and ecosystem service supply into other uses was rather proportional to their share in the total soil pool. The analysis of soil protection efficiency referred to the period between early nineties to 2006/2007, thus it does not relate to any soil management systems introduced recently.

7.3 Drivers/pressures

In order to explain the broader context of soil sealing, the DPSIR framework of the EEA is used, which is a common tool to explain environmental effects.

Driving forces of soil sealing refer to the need for new housing, business locations and road infrastructure related to economic development of cities (Figure 3). Most social and economic activities depend on the construction, maintenance and existence of sealed areas and developed land. Soil consumption has, however, considerable consequences for society and economy.

The new housing or infrastructure developments usually take place at the border of existing settlements creating **pressures** on previous agricultural lands and increasing areas of artificial surfaces and sealed soils.

Intensity of urban development and patterns of spatial planning lead to a certain **state** of soil sealing and land use change. This can be measured by the degree of soil sealing or land transformation but urban sprawl also creates certain states of increased traffic and noise and decreased production potential of land and performance of environmental soil functions.

Soil sealing **impacts** through interrupting the exchange between the soil system and other ecological compartments, including the biosphere, hydrosphere and atmosphere, which affects processes in the water cycle, biogeochemical cycles and energy transfers. This leads to a number of negative effects:

- Less availability of fertile soils for future generations.
- Reduction of soil functions, such as sink and diluter for pollutants and transformation of organic wastes and a reduction of the water storage capacity.
- Loss of water retention areas and at the same time increase in surface water runoff, which leads to additional flood risk and in some cases to catastrophic floods.
- Less soil carbon sequestration and carbon storage.
- Landscape fragmentation and loss of biodiversity through reduction of habitats with remaining systems too small or isolated to support species
- Unsustainable living patterns such as the increase of scattered buildings leading to an increase in traffic and air emissions, infrastructure costs for municipalities concerned and urban development on high-quality agricultural land that leads to a loss of productive soils for food and other biomass production.

- Sealed surfaces have higher surface temperatures than green surfaces and alter the micro climate particularly in highly sealed urban areas (EEA, 2010). Recent surface temperature surveys from the cities of Budapest (Hungary) and Zaragoza (Spain) revealed that temperatures in highly sealed areas can be up to 20 °C higher compared to green shaded surfaces (Prokop *et al.*, 2011).

Responses: These processes as described above can be interrupted by either reducing future land take or by implementing desealing measures. The second option is only rarely applied and very cost intensive. Reducing future land take can above all be realised by influencing planning policies and building rules, promoting reuse of already developed land and brownfields, strengthening inner urban development instead of urban sprawl, and implementing building techniques which consume less soil or maintain some soil functions (in particular permeability). These measures can be of binding or of a voluntary nature.

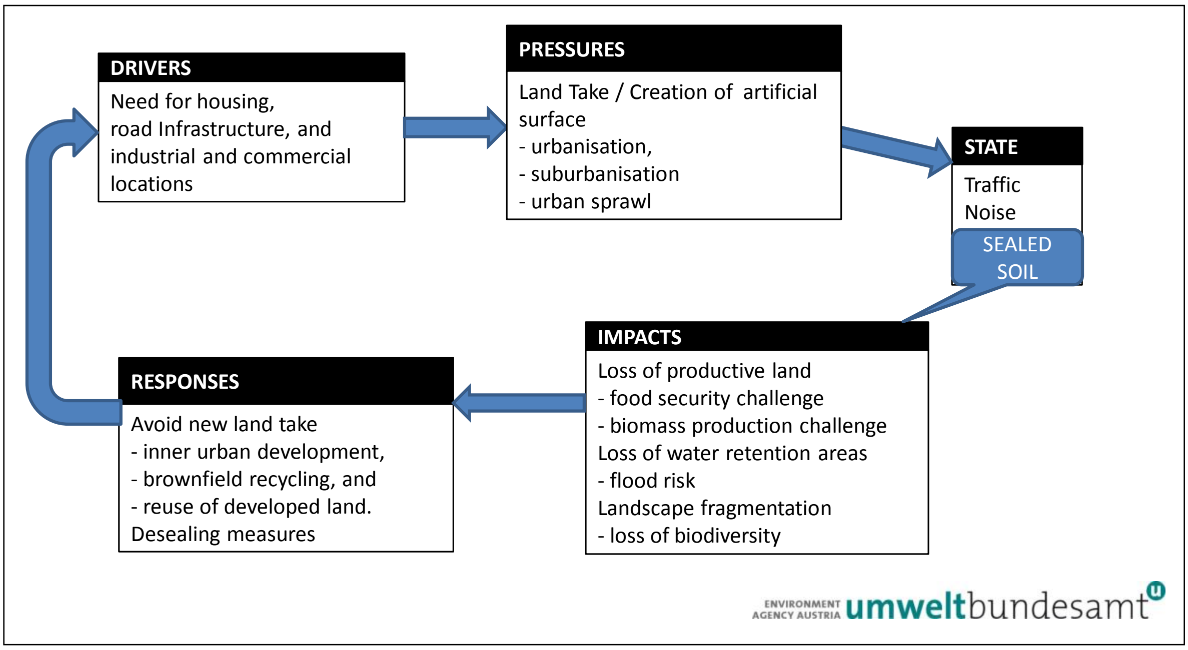

Figure 7.3: Soil Sealing in the context of driving forces, negative effects and possible responses. Source: Prokop et al., 2011.

Soil sealing and land take are not regulated at European level. However, in 2012 the European Commission published *Guidelines on best practice to limit, mitigate or compensate soil sealing* (SWD (2012) 101 final/2) which primarily encourage the Member States to decrease their annual soil sealing rates (EC, 2012). The guidelines collect examples of policies, legislation, funding schemes, local planning tools, information campaigns and many other best practices implemented throughout the EU. They are mainly addressed to relevant authorities in Member States (at national, regional and local levels), professionals dealing with land planning and soil management, and stakeholders in general, but may also be of interest to individual citizens. The best practice examples collected in the guidelines show that smarter spatial planning can limit urban sprawl.

7.4 Key indicators on soil sealing

Indicators on soil sealing or land take can be used for the impact assessment of soil management policies, historical and simulated urbanization trends or for assessing spatial development plans. Sealing Rate indicators (SRI) are meant to reflect how past and future urban development did/will impact soil resources. These are usually simple indicators characterising the proportion of a sealed area in a total area or the increase of a sealed area in a specific period.

Sealing rate indicators can be used for certain assumptions regarding soil protection targets – e.g. it is not sustainable to use soils performing important production or environmental functions for urban development. A simple transition index (TI) has been proposed to characterize the intensity of conversion of agricultural land observed on different soils, classified according to the ranges of land suitability or quality. The ratio between the proportion of a given soil type (class) within the changed area and the proportion of this class in the total soil cover is calculated. Such indicators were first used by Stuczynski (2007) and then applied in the Urban Soil Management Strategy (Urban SMS) project (Siebielec *et al.*, 2010). Similar indicators can be developed for various aspects of soil quality or functions and calculated for, e.g. classes of organic matter content, erosion risk, contamination level.

$$transition\ index(TI) = \frac{percent\ of\ soil\ class\ "n"\ in\ new\ built\ area}{percent\ of soil\ class\ "n"\ in\ whole\ urban\ area}$$

The guidelines for including soil in the Strategic Environmental Assessment (SEA) and Environmental Impact Assessment (EIA) were developed within the URBAN SMS project (Leitner and Tulipan, 2011). The guidelines deal with a list of potential impacts of urbanization on soil cover. Some impacts have a form of measurable indicators whereas for others the indicators in a spatial or point data format need to be matched. Impacts on soil quantity are e.g. characterised by soil sealing areas (hectare), soil conversion rates or ratios between sealed and green areas, both in total area and in an area of land use change. Impacts on soil functions might be represented by loss in soil function performance, loss of soil quality (e.g. due to topsoil stripping and storage), site contamination due to construction activities, compaction caused by e.g. construction damage, loss of natural soil layers and characteristic horizons, reduction in biodiversity, changes to soil water regime and changes in ground water level, fragmentation of green areas.

Another group of indicators refers to the characterisation of urbanisation consequences. In a participatory impact assessment a representative group of stakeholders is involved in assessing impacts of soil policy scenarios through evaluation of social, economic and environmental functions. For example in the procedure developed in the 6FP SENSOR project by Morris *et al.* (2011) and followed after slight modification in the Urban-SMS project, impacts of land take on social, economic and environmental functions of soil or land are semiquantified through relevant indicators (Siebielec *et al.*, 2011).

7.5 Methods to assess status of soil sealing

Methods to assess soil sealing, and its related concepts of artificial land, land take, and urban sprawl, can be divided into methods to assess historic, present and future soil sealing. Historic and present soil sealing, artificial land, land take and urban sprawl are often assessed by Remotely Sensed (RS) data. At smaller scales, use is also made of aerial photography. In addition, use can be made of municipal (planning) maps, cadastre maps or other maps indicating the location of the built environment. An example of using RS data to assess urban development at large scales is through light emitted from the Earth's surface at night. In other RS studies it is mostly the 'hard surfaces' or 'sealed soil' that is being looked at.

Understanding future soil sealing can be done by analysing and extrapolating past trends, through modelling, through stakeholder participatory exercises or a combination of these methods.

Looking at EU-wide datasets, relevant for RECARE as they provide comparable information all over Europe, there are four worth mentioning:
Before 2009 the CLC data (http://www.eea.europa.eu/publications/COR0-landcover) and LUCAS data (http://eusoils.jrc.ec.europa.eu/projects/Lucas/) were the only two EU-wide spatial data sets, with the CLC data being most widely used for assessing land cover and land uses changes as it covers the entire European territory, while the LUCAS data are a collection of points, specifically aimed for top soil analysis, on a grid throughout Europe, with information provided at each point that is much richer than the land use classification provided by CLC (Toth *et al.*, 2013).

In 2009, the European Environment Agency published a specific Soil Sealing Layer (SSL) of Europe, which covers the whole European territory and has a higher resolution compared to the CORINE Land Cover data sets (EAA, 2013). The minimum mapping unit is 20 m · 20 m sealed surface within a pixel size of 100 m · 100 m. This layer aims to analyse the proportion of sealed surface per region and per capita (http://www.eea.europa.eu/data-and-maps/data/eea-fast-track-service-precursor-on-land-monitoring-degree-of-soil-sealing)

Recently, the Joint Research Centre of the European Commission has developed a global human settlement layer (GHSL), which integrates several available sources reporting about the global human settlement phenomena, with new information extracted from available remotely sensed (RS) imagery (JRC, 2014). The

GHSL automatic image information extraction workflow integrates multi-resolution (0.5m-10m) multi-platform, multi-sensor (pan, multispectral), and multi-temporal image data. The GHSL is an evolutionary system, with the aim of stepwise improving completeness and accuracy of the global human settlement description by offering free services of image information retrieval in the frame of collaborative and derived-contents sharing agreements (http://ghslsys.jrc.ec.europa.eu/).

As these data sources have different purposes and are developed using different methods, they all have pros and cons in dealing with soil sealing, land take and urban sprawl issues.

A major benefit of CLC is that it is available for three periods in time 1990, 2000 and 2006 and a fourth period (2012) is upcoming. Moreover, often data derived from RS images has been assessed on its accuracy to match the actual data for a specific year, while changes over time in the derived products are not well accounted for. CLC does, however, take changes between years and their plausibility into account, which makes it a valuable dataset for assessing dynamics.

As to the accuracy of the CLC data, it has to be underlined that land use changes involving small settlements, as well as most linear structures, e.g. the road system or other transport infrastructure, are not sufficiently captured (EU, Guidelines on best practice to limit, mitigate or compensate soil sealing, p.43). Also, the dataset only includes a few urban classes and e.g. does not distinguish between industrial and commercial development, while this type of information is essential for understanding the underlying processes behind soil sealing. Also the dataset does not indicate what part of the urban development is actually sealed, or how (im)permeable the seal is.

The Soil Sealing Layer of Europe (EEA, 2013) has been developed specifically for the purpose of assessing soil sealing and hence does focus on sealed soil. The soil sealing layer is more accurate than CLC in providing information about sealed soil, however, it does not provide information on the type of sealing (house, industry, commercial activity, parking lot, road) or on the permeability of the sealed surface. It provides information on degree of soil sealing in a cell in a range 0-100%. At present there is also only one layer available (year 2006), which limits the option for temporal analysis.

The GHSL layer (JRC, 2014) provides detailed information on urban settlement locations at a very high resolution. As the GHSL focuses especially on the buildings, and not on roads, parking lots, and other infrastructure, it does not have the intention to map the sealed soil. Furthermore, the layer does not yet map the different types of human settlements (residential, industry, commercial, etc), although due to its high level of detail it does offer the possibility of deriving this information. This layer also does not provide information on (im)permeability of the sealed surface.

Finally, the LUCAS dataset (Tóth *et al.*, 2013) does provide much richer data than the datasets above. It also includes a time series of information, enabling a temporal assessment. However, the dataset is a point dataset that does not cover European territory and soil sealing and permeability of the sealed surface might be very dependent on local characteristics.

At present, in some Member States soil sealing, land consumption and some response measures (brownfield redevelopment, de-sealing) are monitored in a quantitative way by applying mostly statistical methods or aerial photograph interpretation. Much of these data is, however, not comparable since different methodologies are used. The MOLAND (Monitoring Land Use Dynamics) database allows an assessment of change rates in built-up areas at regional and local level for a limited number of urban areas (EEA, 2006). The Urban SMS project provided an assessment of the quality of soil lost in the urbanisation process within a 15 y period in Central Europe cities (Siebielec *et al.*, 2010). All monitoring approaches mentioned use the extent of built-up area as a proxy indicator to estimate the sealing degree of the land consumed.

Which methods and data sources would be most appropriate to measure and assess soil sealing, strongly depends on what the information is used for and hence what the relevant indicators are. If thinking, for example, about the infiltration capacity of the soil, simply estimating the percentage of artificial land in a certain location is not sufficient, as one would also like to know how permeable the artificial surfaces are. Another example is the impact of sealed soil on flood risk of storm water in urban environments. Here, just knowing what parts of the soil are sealed is not sufficient, one would also need more information on the entire drainage system. Finally, the location of the sealed surfaces and their structure in the wider landscape are often important as this has an impact on the hydrology of the system, but also on habitat fragmentation and the local climate.

Regarding the forecasting of soil sealing, two well-known models at EU level are capable of exploring future land take at detailed resolution (100-1000 m cells): the LUMP/CLUE modelling framework and the LUMOCAP/Metronamica modelling framework. Both frameworks model the land use changes that occur in the European Union using a classification derived from CORINE Land Cover. This means that the models simulate changes in artificial areas or land take, and hence an interpretation of artificial land, or land take, to soil sealing still needs to be made. These models allow the incorporation of additional information, such as the quality of the (sealed) locations and the permeability of different types of seals. They are also capable of assessing the impacts of urban development on land use functions. However, at present these models are not advanced enough that the procedures are automatic, although early attempts have been made to do so (see e.g. Van Delden et al, 2011).

7.6 Effects of soil sealing on other soil threats

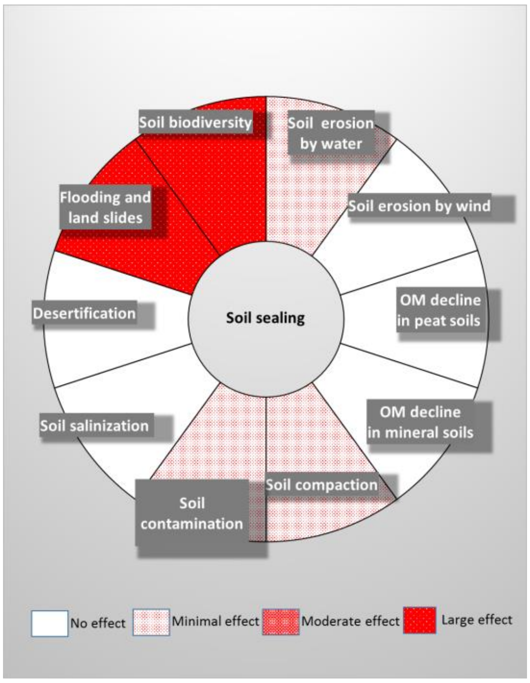

Figure 7.4: Effects of soil sealing on other soil threats. Red is negative effect.

The impact of urbanization on soil organic matter is twofold. The overall pool of soil organic matter is reduced in an area of urban sprawl. The organic top layer of the soil profile is often stripped prior to construction work. On the other hand collected humus might be used in another location to enhance soil functions and soil organic matter content in the treated soil. If construction work or soil sealing takes place on organic soils increased GHG emissions is observed. Profile disturbance of organic soil causes mineralization of peat and thus a release of GHG to the atmosphere.

Construction work causes soil compaction due to heavy machinery. This effect is increased when soil is excessively saturated with water. Destruction of the soil profile increases the susceptibility of soil to compaction.

Soil sealing drastically increases the risk of flooding. A high proportion of impermeable surfaces leads to enhanced runoff during rainstorms, which significantly contributes to riverine flooding. Proper spatial planning and use of partly permeable materials (e.g. for paths) reduces the risk of flooding. Landslides risk might be enhanced by human settlements due to loss of natural vegetation and disturbance of slope stability or increased runoff and related soil erosion.

Urbanization usually increases the background contents of pollutants in the soil (e.g. trace elements or polycyclic aromatic hydrocarbons) although not necessarily exceeding risk levels in soil. Soil pollution might appear locally as a direct effect 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 the inactivation of contaminants in land. On the other hand, a high soil sealing density accelerates air contamination, especially with particulate matter (PM). Mosaics of green spaces and sealed areas with trees reduce the concentration of contaminants in the air and inactivates them.

Urbanization causes a loss of overall biodiversity (diversity of animals, natural habitats, plant species) in areas of urban sprawl due to the fragmentation of land through transport infrastructure and commercial/residential buildings. Sealing of soil causes practically a complete loss of soil biodiversity since this soil function is no longer provided in the sealed spot.

Soil sealing might accelerate soil water erosion in certain cases since the high density of impermeable surfaces increases the surface flow of water. Furthermore, disturbance of the soil profile during construction work, loss of soil organic matter and often bare surfaces make the soil more susceptible to erosion.

There is no direct link of soil sealing or land take with soil salinization, except local accidental impacts of industrial origin.

7.7 Effects of soil sealing on soil functions

7.7.1 Food and other biomass production

Very often, the most productive soils can be found in sub-urban areas at the borders of urban agglomerations, which are predominantly used for agriculture. The existence of these areas has its rationale. It is mostly due to the ease of accessibility to crop markets and the unlimited sales opportunities offered by large metropolitan areas. Therefore, it is necessary to conserve and maintain the soil productivity and soil availability for production in sub-urban areas (Nizeyiamana *et al.*, 2001; Siebielec et al, 2012).

There are several studies on the state and trends of soil sealing in Europe (e.g. Jones *et al.*, 2012; EEA, 2014), but only a few studies have assessed the effects of soil sealing on soil functions and ecosystem services in Europe. Gardi *et al.* (2014) assessed the impact of land take and soil sealing on the EU's ability to produce food. They found that between 1990 and 2006, 19 EU countries lost around 1% of their potential agricultural production capability due to land take processes. Verzandvoort (2010) assessed impacts of soil sealing on food production, water retention, biodiversity and soil organic carbon stocks in the EU over the periods 1990-2000 and 2000-2030 (under the SRES B1 reference scenario 'Global Cooperation'). The loss of suitable land for arable cropping and permanent grassland amounted to 5% of the land area at the level of NUTS3 units, and to 1% of the land area in EU countries in the period from 1990 to 2000. For the period 2000-2030 and under the Global Cooperation scenario, a similar loss was projected. In addition, direct losses of suitable land for agriculture through building activities are currently compensated for by food production in other regions of the world, with potentially less favourable conditions for food and biomass production than the soils suitable for agriculture subjected to urban expansion in European countries (Tobias, 2013). Though the loss of suitable land for agriculture seems small compared to the total stock of agricultural land in the EU27 (46% of the land area in 2000, 40% in 2030 according to the Global Cooperation scenario), the loss may be significant in terms of net primary productivity. If the EU becomes more dependent on food and biomass production from its own land area, then the loss of suitable land for agriculture due to soil sealing may become an issue.

7.7.2 Environmental interaction: storage, filtering, buffering and transformation

The supply of soil water and groundwater in cities has a generally downward trend due to the "drying effect of cities". The hydrological cycle in urban areas has different features than in rural areas and it is affected by often disturbed soil profiles, limited infiltration, accelerated runoff, compaction of soils, presence of impermeable layers, physical barriers. Sealed areas produce intensive runoff whereas bare soils, as a result of compaction, absorb water to a limited extent. On the other hand the covered soils dry more slowly after flooding. An impermeable layer in the profile may cause water accumulation in the profile and water logging (Siebielec et al, 2012).

Soil components and organisms are responsible for filtering, degrading, immobilizing, and detoxifying organic and inorganic pollutants that enter the soil via industrial and municipal wastes or through atmospheric deposition. Some of these compounds (organic pollutants) are degraded by microorganisms in the soil and transformed into less harmful forms. Other pollutants (e.g. trace elements) are held in the soil which prevents secondary contamination of air and water (Siebielec *et al.*, 2012).

In urban areas, the storage and buffering functions are often reduced, being sometimes irreversibly lost. In fact, the typical mixing of soils with extraneous materials (bricks, debris from construction, etc.) strongly modifies its original physical-chemical properties, often leading to an increase in the coarse fraction, a reduction of organic matter and microbial activity and an increase in leaching of contaminants. Also, the fact that unsealed soils in urban areas are not covered by vegetation enhances the dispersion of contaminants through wind erosion of soil particles (Siebielec *et al.*, 2012).

Soil properties and functionalities of unsealed soils within built-up areas are changed through mechanical impacts on soil structure (Sauerwin, 2011). The affected properties of sealed and unsealed soil include physical properties (new substrates, water, radiation and heat budgets, aeration), chemical properties (pH-value, substance mobility), biological properties (the living conditions for flora and fauna) and site ecology (soils as the basis for biotopes) (Sauerwein, 2011; Niemela, 2011)).

Summarizing, soil sealing and urban sprawl increases the risk of floods and pollution, having negative consequences for human health, and consequently higher societal costs (Imhoff, 2004).

Soil sealing also has a negative impact on local climate. Sealed surfaces absorb heat and increase surface and ambient air temperatures ('urban heat island effect'). Increased runoff over impervious surfaces (in particular of car parks or paved gardens) causes reduced available water for evaporation, which would otherwise have a cooling effect in urban areas.

The loss of environmental functions can be somehow mitigated through the use of partly permeable layers and presence of green (plants) or blue (water) spaces. Urban land uses, such as street trees, lawns and parks, urban forests, streams and cultivated land, generate a range of ecosystem services, like micro-climate regulation, air filtering, water retention (e.g. Bolund and Hunhammar, 1999; Tzoulas, Korpela *et al.* 2007).

7.7.3 Biological habitat

The role of soil for the provision of biodiversity refers to two different issues: soil biodiversity referring to the biological diversity of soil organisms, and the function of soil as a habitat for plant and animal species.

Urbanization introduces remarkable changes in biodiversity (Williams *et al.*, 2009). Soil sealing results in habitat loss for soil organisms, plant species and animals. Such pressures lead to local extinction processes, elimination of native species and their displacement by non-native species. This, in consequence, threatens the biological uniqueness of ecosystems. Decrease in soil biodiversity leads to the inhibition or slowdown of organic matter and nutrient cycles (Siebielec *et al.*, 2012).

Soil sealing and urban sprawl cause a loss of habitat and ecosystem area due to the conversion of the original land cover into an artificial surface. Furthermore urban infrastructures result in the interruption or disturbance of ecological networks: habitat fragmentation and physical, thermal, visual or chemical barrier effects impacting the flux of material and species within and between ecosystems.

Impacts of soil sealing on biodiversity were assessed for the EU for the period 2000-2030 in the SRES B1 projection using the Mean Species Abundance index as an indicator (Verzandvoort *et al.*, 2010). A decrease in biodiversity in sealed areas was projected in all member states of the EU27, up to -35% points.

7.7.4 Physical and cultural heritage

The cultural heritage function of soil, measured as an archaeological value, is generally negatively affected by urbanization and soil sealing, despite the fact that some construction work might help to discover buried records of natural or human history.

Construction work can destroy archive information in soil. Deep digging of the soil profile might open previously buried fragile materials or forms of high archaeological value and make them susceptible to erosion or decomposition. Peatlands that contain lots of information on the natural history of the site are susceptible to degradation due to changes of water conditions. Soil sealing usually prevents the further investigation of cultural archaeological value of a site.

7.7.5 Platform for man-made structures: buildings, highways

Soil sealing obviously enhances some economic soil/land functions and related services, like the carrier service to provide ground for human activities and infrastructure (De Groot, 2006; Huber *et al.*, 2008; Van der Wel, 2011). Major functions provided by sealing and urbanisation as a whole are housing and workplace provision (industry, services, commerce) and transport infrastructure.

7.7.6 Source of raw materials

The intensity of soil sealing is correlated to the extraction of some raw materials from soil and parent rock material level. Some of them are used as feedstock materials in construction work (sand, clay, limestone). Therefore, soil sealing is a driving force for extraction of raw materials. Furthermore, development of the mining sector causes soil sealing because of the infrastructure (roads, buildings, landfills) that is needed for mining and related activities.

7.7.7 Participatory impact assessment of sealing impacts

Participatory impact assessment of urbanization consequences applied within Urban SMS project, involved participation of local stakeholders and collecting their opinions on possible urbanization impacts. The stakeholders were led through steps of an impact assessment in order to ascertain their opinions in a semi-quantitative form (Morris J. *et al.* 2011). The workshops were organized in the following cities of Central Europe: Celje, Vienna, Milan, Prague, Wroclaw and Bratislava. Three scenarios representing different soil protection approaches were assessed regarding their long-term impacts on the soil functions (Siebielec *et al.*, 2011). The baseline scenario assumed that nothing would change with regulations concerning soil protection. The no-change scenario was assessed by stakeholders as favorable to economic functions 'Housing and workplace provision' and 'Transport infrastructure' whereas all environmental functions were deemed as highly threatened (Figure 7.5).

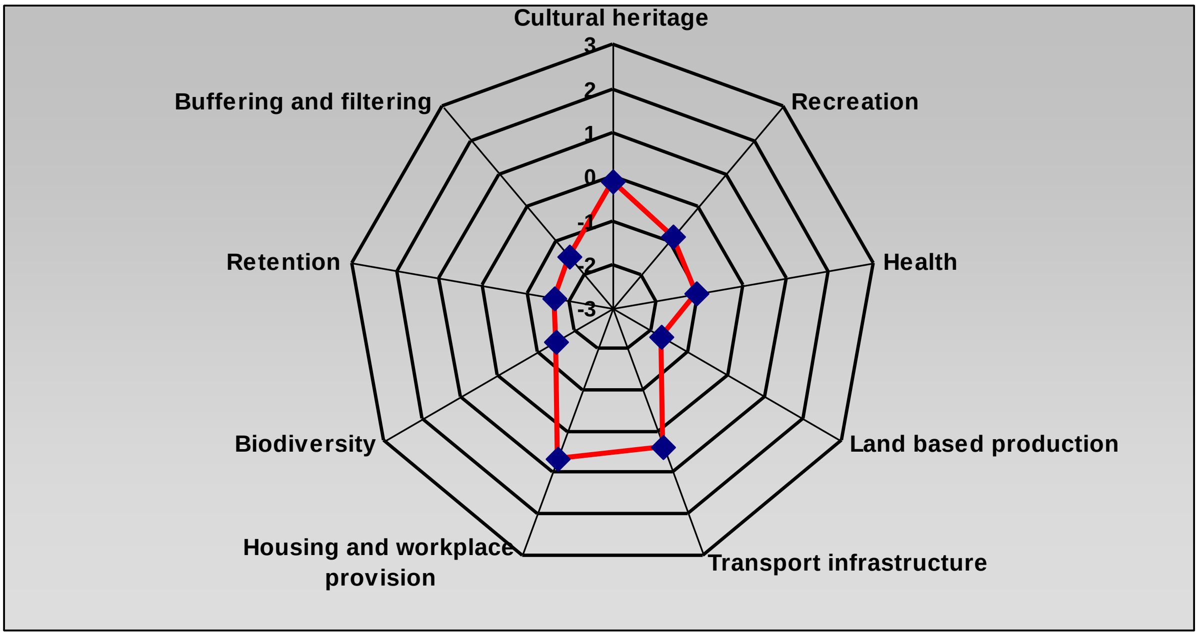

Figure 7.5: Impact of current soil protection scenario on soil functions across cities of Central Europe (positive values mean improvement, negative values mean loss of functions), (Siebielec et al., 2011).

As we reported, the effects of soil sealing on the soil functions vary from the loss of net primary productivity of the landscape and natural habitats to increased floods, pollution and health risks and consequently higher societal costs (Imhoff 2004; Scalenghe and Marsan 2009; Lorenz 2009). The performance of most ecosystem services declines with increasing sealed area (also including more densely urbanised areas) (Tratalos *et al.* 2007; Tobias 2013), but also the spatial configuration of ecosystems in the pattern of land use converted for urban development influence the supply of ecosystem services (e.g. Xiao *et al.* 2013; Haase 2007). Niemela *et al.* (2011) describe soil sealing as the key factor for changes in the water cycle, climate and vegetation cover, noting that this has not been adequately considered in urban ecosystem management, particularly with regard to the mitigation of effects of climatic extremes. This calls for an approach that integrates ecological (land management) and social solutions to foster ecosystem services in urban areas that rely on soil functions, like food and biomass production and biodiversity in cities, water regulation and the provision of recreational spaces.

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8 SOIL CONTAMINATION

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8.1 Description of soil contamination

A contaminant is any substance with the potential to cause damage, irreversible or not, in the environment. Environmental contamination concerns the presence in the environment of any (physical, chemical or biological) agent or combination of agents in a site in forms and concentrations that are or may be harmful to health, safety or welfare of the contaminant or prevent its normal use. Soil contamination is the occurrence of contaminants in soil above a certain level causing deterioration or loss of one or more soil functions (JRC, 2014). It is therefore a chemical degradation that causes partial or total loss of soil functions. Soil productivity is affected as well as soil organisms. A contaminated soil has exceeded its capacity for natural attenuation for one or more substances, and consequently passes from acting as a protector to cause adverse effects to the water system, the atmosphere, and organisms. The soil's biogeochemical equilibria are modified and abnormal amounts of certain components appear that cause significant changes in the physical, chemical and biological soil properties (Adriano, 2001).

The terms contamination and pollution are often used interchangeably. When a distinction is made, two main aspects are often considered:

- pollution as an activity that causes contamination
- contamination being the presence of a foreign substance, not necessarily harmful, while pollution indicates that harm is being done.

Some definitions combine both aspects. In this report we use the term 'contamination' for the soil threat, as this is also done in the European Committee documents, and in the RECARE project documents. The human activity that causes contamination is called 'pollution'.

In general, the typology of contaminants is varied and complex, following different criteria:
a) Sources
Contaminants can be released from point pollution sources, e.g. waste water treatment plants from urban or industrial areas, or from diffuse sources through atmospheric deposition or from crop and animal production.

On the European scale, point pollution is well documented and studied. However, diffuse contamination is the most widespread contamination and difficult to assess. This type of contamination is characterized by long distance transport of low concentrations of contaminants that are deposited in soils as a sink. They can be released in the environment with a specific purpose such as pesticides or as an unwished byproduct of production processes. Generally, they are distributed in small doses over large surfaces. Diffuse contamination is more difficult to control than point pollution because it is linked to a multitude of sources spread all over the land (Adriano 2001).

b) *Types of contaminants*
Nowadays, more than 700 emerging pollutants, their metabolites and transformation products, are present in the European environment (NORMAN 2014). Emerging pollutants (EPs) are defined as synthetic or naturally occurring chemicals or microorganisms that are not commonly monitored in the environment but which have the potential to enter the environment and cause known or suspected adverse ecological and (or) human health effects. EPs are categorised into more than 20 classes related to their origin (NORMAN, 2014, Figure 8.1). The prominent classes are: pharmaceuticals (urban, stock farming), pesticides (agriculture), disinfection by-products (urban, industry), wood preservation and industrial chemicals (industry).

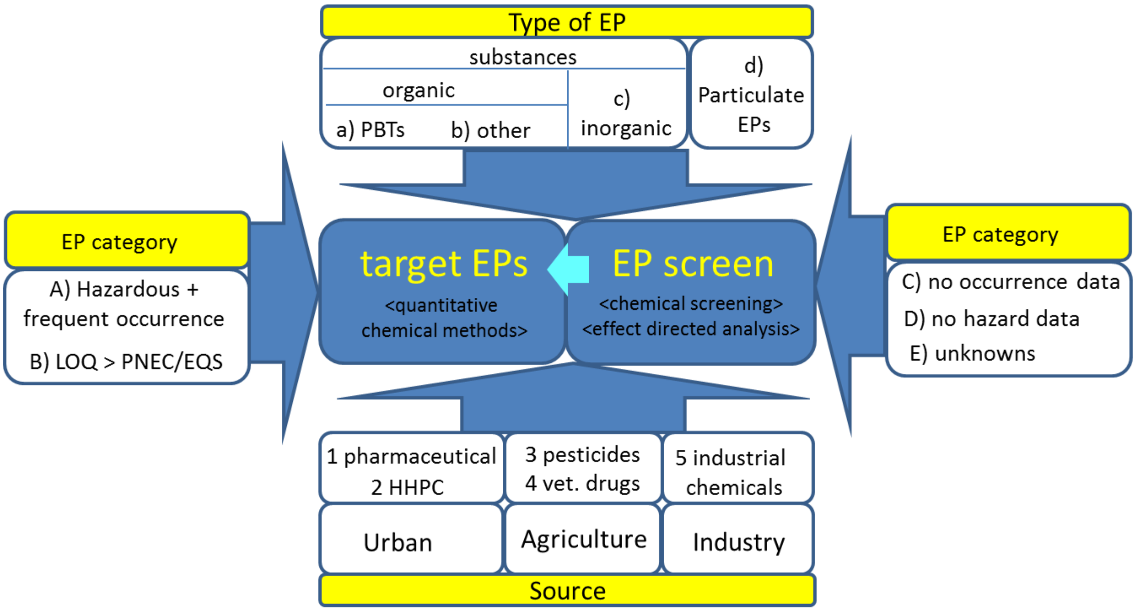

Figure 8.1: Groups of emerging pollutants (EP).

Soil contaminants can consist of various forms such as organic and inorganic or particulate contaminants (Mirsal, 2008). Organic contaminants are substances whose molecules contain one or more (often many more) carbon atoms covalent bonded with another element or radical (including hydrogen, nitrogen, oxygen, the halogens as well as phosphorus, silicon and sulfur) whereas an inorganic contaminant is any compound not containing carbon atoms such as heavy metals (Fig. 8.1). Inorganic pollutants do not underlay decay and therefore once released into the soils stay, whereas organic pollutants underlay a process of decay.

Soil enrichment in inorganic contaminants may be caused by both natural and anthropogenic factors. Examples of natural contamination are the serpentine soils, in which elevated concentrations of nickel and chromium are the best recognized case of natural enrichment related to the parent rock. Volcanic emissions and fires are also natural sources of soil pollution (Alloway, 2013). A natural process of bioaccumulation usually brings about significant differentiation between metal content in humus horizons and deeper soil layers.
However, the most important sources of contamination in soils are those connected with anthropogenic activities (Alloway, 2013), such as point pollution e.g. metal mining and smelting, industrial production, waste disposal and diffuse pollution by industrial activities, car emissions, application of agrochemicals, manure containing veterinary drugs, etc.

Soils in the vicinity of smelters and other industrial plants that formerly emitted large amounts of air-borne metal-rich particles will remain contaminated with metals for a long time (Davies, 1983), despite the fact that the emissions have recently been dramatically cut. Special attention is still being given to hazardous sites with large amounts of heavy metals, such as abandoned mines, mine spoils, tailings and other metal-bearing wastes (Adriano, 2001).

In light of the potential impact of these substances on aquatic life and human health, the lack of knowledge regarding their behaviour in the environment and the deficiency in analytical and sampling techniques, action is urgently required.

8.2 State of soil contamination in Europe

Soil contamination in Europe can be divided into different topics according to the source of pollution (point or diffuse, from industry, urban or agriculture) and the types of the (emerging) pollutant (organics, inorganics, particulate pollutants).

8.2.1 Sources of point contamination

In 2011-12, the European Soil Data Centre of the European Commission conducted a project to collect data on contaminated sites from national institutions in Europe using the European Environment Information and Observation Network for Soil (EIONET-SOIL). According to the received data, the total number of identified contaminated sites caused by point pollution is 2.5 million, the estimated number of potentially contaminated sites is 11.7 million (Panagos *et al.*, 2013). Municipal and industrial wastes contribute most to soil contamination (37%), followed by the industrial/commercial sector (33%). Mineral oil and heavy metals are the main contaminants contributing around 60% to soil contamination (Fig. 8.2). In terms of budget, the management of contaminated sites is estimated to cost around 6 billion Euros (€) annually (Panagos *et al.*, 2013).

8.2.2 Diffuse pollution with respect to heavy metals

Lado *et al.* (2008) present the results of modelling the distribution of eight critical heavy metals (arsenic, cadmium, chromium, copper, mercury, nickel, lead and zinc) in topsoils using 1588 georeferenced samples from the Forum of European Geological Surveys Geochemical database (26 European countries) (Fig. 8.3).

High values of Cr and/or Ni are mainly found in central Greece, northern Italy, the central Pyrenees, northern Scandinavia, Slovakia and Croatia and show a strong correlation between the contents of Ni and Cr and the magnitude of earthquakes. The seismic activity is indirectly correlated with heavy metal concentrations — such materials provide high quantities of Ni and Cr to the soils by weathering processes. Cadmium, Cu, Hg, Pb, Zn present a high concentration in Central Europe and are mainly related with agriculture and with quaternary limestone. The use of fertilizers, manure and agrochemicals are important sources of these elements. They are also inversely correlated with distance to roads (Lado *et al.,* 2008).

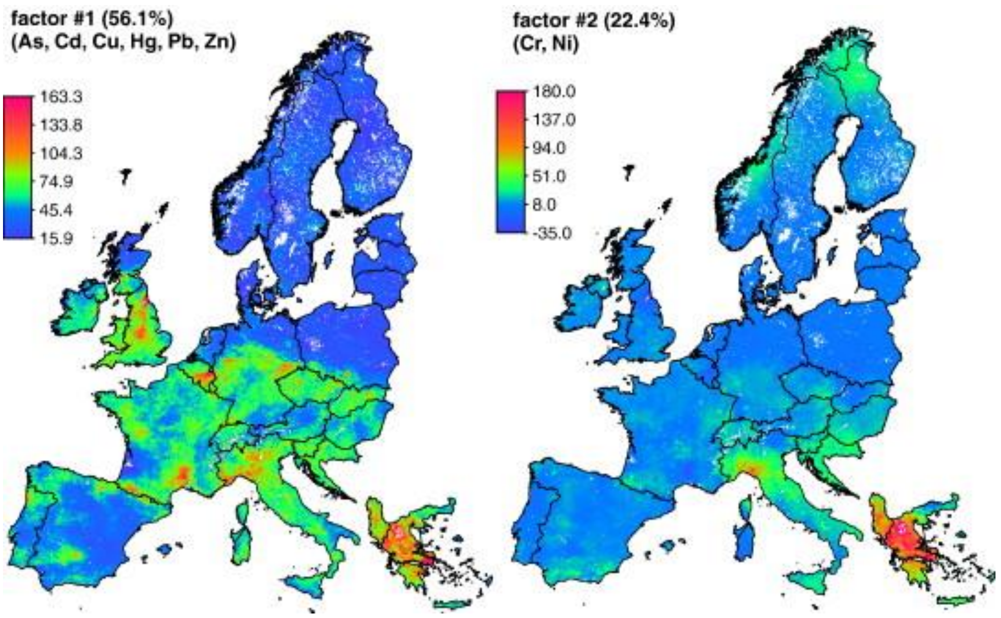

Figure 8.3: Heavy metal content in European soils (Lado et al. 2008).

8.2.3 Diffuse pollution with respect to emerging pollutants from industrial/urban sources

Although there are 700 emerging pollutants described in the European environment (NORMAN, 2014), until now, they are only taken under consideration in the aquatic environment. Their presence and concentration in the terrestrial ecosystem is unknown as is the potential risk for the environment. Aerial transport of pollutants from industrial and urban sources is even more difficult to monitor because their distribution and the fall out is not easily known.

8.2.4 Diffuse pollution with respect to agrochemicals

More than 3000 different types of pesticides have been used in the European agricultural environment in the past 50 years. It has been estimated that less than 0.1% of the pesticide applied to crops actually reaches the target pest; the rest enters the environment, contaminating soil, water and air, where it can poison or otherwise adversely affect non-target organisms (Pimentel and Levitan, 1986). Furthermore, many pesticides can persist for long periods in an ecosystem—organochlorine insecticides, paraquat, deiquat for instance, were still detectable in surface waters 20 years after their use had been banned (Larson *et al.*, 1997). Few studies have been carried out monitoring the mixtures of pesticides present in our soils. Oldal *et al.* (2006) and Ferencz and Balog (2010) found high concentrations of mixtures of organochlorines and lindane even 20 years after they were forbidden in Hungarian and Romanian soils. Whilst the EC has data available on the herbicide applications per country (Fig. 8.4), no data exist on the actual pesticide concentration in European soils.

We urgently recommend the establishment of European wide monitoring programs with respect to (emerging) pollutants in our soils.

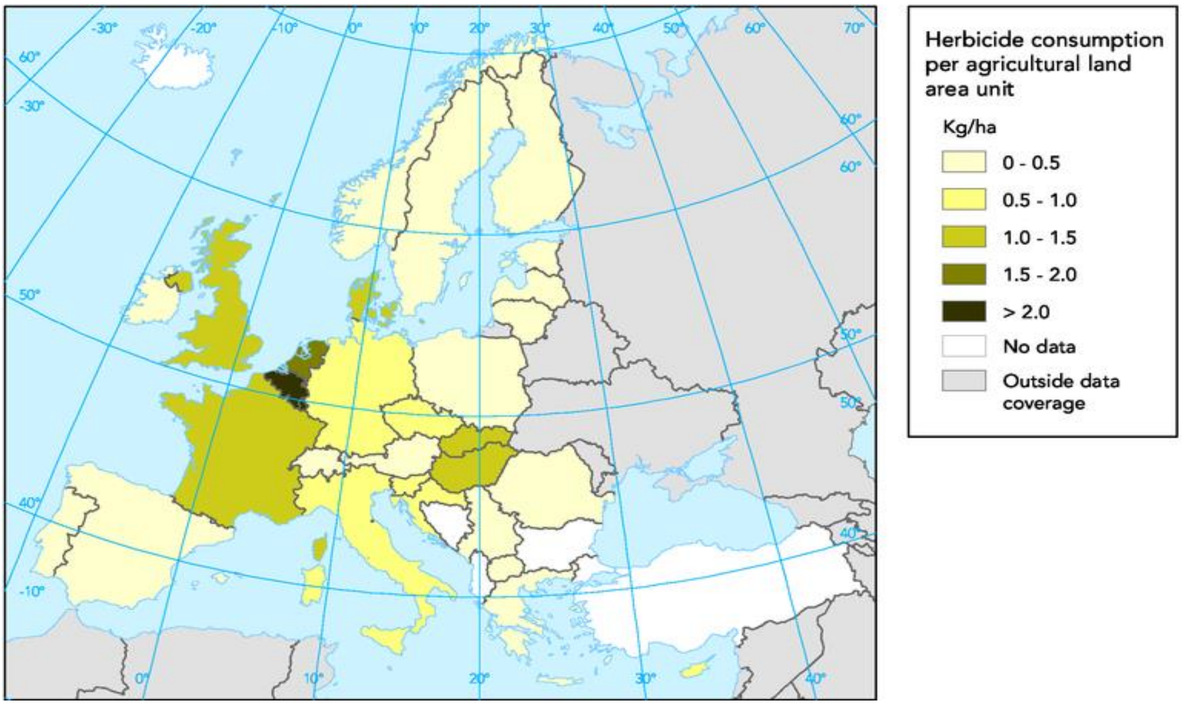

Figure 8.4: Herbicide consumption in the EU countries (source: European Environmental Agency, EEA, 2015).

8.3 Drivers and pressures of soil contamination

The main drivers of soil contamination are anthropogenic in character. They include the main sectors of the economy, such as industry, transport, waste management and agriculture. Manufacturing processes are usually accompanied by certain contaminant release at a level dependent on production intensity, technologies used, and materials processed. The human activities listed by Directive 2010/75/EU of the European Parliament and of the Council of 24 November 2010 on industrial emissions (integrated contamination prevention and control) (IED) posing a risk of contaminant emissions are shown in Table 8.1.
It must be noted that implementation of modern technologies and more strict emission standards have reduced the level of emissions of contaminants. However, historical activities have often left significant contents of various substances in soils.

Transport has been a source of lead compounds, however, the importance of this sector has been reduced as a result of implementing lead-free fuel. The information on impact of intensive road transport on release of organic pollutants is scarce.

Agricultural production as a source of soil contamination is currently relatively less important. In the past, risks were related to the presence of cadmium or lead in phosphate fertilizers or waste liming materials or non-sustainable use of pesticides (Chaney & Oliver, 1996). Uncontrolled application of municipal sewage sludge might cause transfer of contaminants to soil (metals, PCBs, dioxins, etc.).

The major drivers of soil contamination are European or national regulations. Increasing awareness on a risk related to release of pollutants has led to implementation of regulations at EU level aimed at reducing a pressure of urban and industrial development on natural resources, including soil.

Table 8.1: List of human activities posing a risk of contaminant emissions (Directive 2010/75/EU).

Source	Activities
Energy industries	• Combustion of fuels in installations with a high thermal input • Refining of mineral oil and gas • Production of coke • Gasification or liquefaction of coal and other fuels
Production and processing of metals	• Metal ore roasting or sintering • Production of pig iron or steel including continuous casting • Processing of ferrous metals • Operation of ferrous metal foundries • Processing of non-ferrous metals • Surface treatment of metals or plastic materials using an electrolytic or chemical process
Mineral industry	• **Production of cement, lime and magnesium oxide** • Production of asbestos or the manufacture of asbestos-based products • Manufacture of glass including glass fibre • Melting mineral substances including the production of mineral fibres • Manufacture of ceramic products by firing, in particular roofing tiles, bricks, refractory bricks, tiles, stoneware or porcelain
Chemical industry	• Production of organic chemicals, such as simple hydrocarbons (linear or cyclic, saturated or unsaturated, aliphatic or aromatic); oxygen-containing hydrocarbons such as alcohols, aldehydes, ketones, carboxylic acids, esters and mixtures of esters, acetates, ethers, peroxides and epoxy resins; sulphurous hydrocarbons; nitrogenous hydrocarbons such as amines, amides, nitrous compounds, nitro compounds or nitrate compounds, nitriles, cyanates, isocyanates; phosphorus-containing hydrocarbons; halogenic hydrocarbons; organometallic compounds; plastic materials (polymers, synthetic fibres and cellulose-based fibres); synthetic rubbers; dyes and pigments; surface-active agents and surfactants • Production of inorganic chemicals, such as gases, such as ammonia, chlorine or hydrogen chloride, fluorine or hydrogen fluoride, carbon oxides, sulphur compounds, nitrogen oxides, hydrogen, sulphur dioxide, carbonyl chloride; acids, such as chromic acid, hydrofluoric acid, phosphoric acid, nitric acid, hydrochloric acid, sulphuric acid, oleum, sulphurous acids; bases, such as ammonium hydroxide, potassium hydroxide, sodium hydroxide; salts, such as ammonium chloride, potassium chlorate, potassium carbonate, sodium carbonate, perborate, silver nitrate; non-metals, metal oxides or other inorganic compounds such as calcium carbide, silicon, silicon carbide • Production of phosphorous-, nitrogen- or potassium-based fertilisers (simple or compound fertilisers) • Production of plant protection products or of biocides • Production of pharmaceutical products including intermediates • Production of explosives
Waste management	• Disposal or recovery of hazardous waste • Disposal or recovery of waste in waste incineration plants or in waste co-incineration plants • Landfills • Temporary and underground storage of hazardous waste

The Sewage sludge EU directive (Directive on the protection of the environment, and in particular of the soil when sewage sludge is used in agriculture – 86/278/EEC) defines conditions for sewage sludge application to soils. In the past, there were examples of uncontrolled application of municipal sludge of low quality that have caused soil pollution. Therefore, the directive provides threshold trace metals contents in soil and sludge as well as allowed annual inputs of metals. They refer to the following elements: zinc, lead, cadmium, nickel, copper and mercury.

The purpose of the ELD directive (Directive 2004/35/CE of 21 April 2004 on environmental liability with regard to the prevention and remedying of environmental damage) is to establish a framework of environmental liability, based on the "polluter-pays" principle, to prevent and remedy environmental damage. The ELD aims at ensuring that the financial consequences of certain types of harm caused to the environment will be taken by the operator who caused this harm. This prevention instrument refers to various natural resources including protection against soil pollution.

The IE directive (Directive 2010/75/EU of 24 November 2010 on industrial emissions (integrated pollution prevention and control) is aimed at establishing a general framework for the control of the main industrial activities, giving priority to intervention at the source, ensuring prudent management of natural resources and taking into account, when necessary, the economic situation and specific local characteristics of the place in which the industrial activity is taking place. In order to ensure the prevention and control of pollution, each installation should operate only if it holds a permit or is registered. The Directive implements the term of “best available techniques” (BAT), meaning the most effective and advanced stage in the development of activities and their methods of operation. This indicates the practical suitability of particular techniques for providing the basis for emission limit values, and other permit conditions, designed to prevent and, where that is not practicable, reduce emissions.

The Landfill Directive’s (Council Directive 99/31/EC of 26 April 1999 on the landfill of waste) objective is to prevent or reduce negative effects on the environment from the landfilling of waste, by introducing stringent technical requirements for waste and landfills. The Directive is intended to prevent or reduce the adverse effects of landfill on the environment, in particular on surface water, groundwater, soil, air and human health. It defines the different categories of waste (municipal waste, hazardous waste, non-hazardous waste and inert waste) and applies to all landfills, defined as waste disposal sites. Landfills are divided into three classes: landfills for hazardous waste; landfills for non-hazardous waste; and landfills for inert waste.

The Directive 2000/76/EC on the incineration of waste (the WI Directive) is aimed at preventing or reducing negative effects on the environment caused by the incineration and co-incineration of waste. The WI Directive sets emission limit values and monitoring requirements for pollutants to air such as dust, nitrogen oxides (NO_x), sulphur dioxide (SO_2), hydrogen chloride (HCl), hydrogen fluoride (HF), heavy metals, dioxins and furans. Most types of waste incineration plants fall within the scope of the WI Directive, with some exceptions, such as those treating only biomass (e.g. vegetable waste from agriculture and forestry).

One of unresolved problems in Europe is the high number of brownfields. The following definition of brownfields is provided by the Cabernet report (Ferber *et al.,* 2006): “sites that have been affected by the former uses of the site and surrounding land; are derelict and underused; may have real or perceived pollution problems; are mainly in developed urban areas; and require intervention to bring them back to beneficial use”. There is no EU regulation concerning brownfields and only few countries have developed national strategies to deal with such sites. Especially problematic brownfield types are smelter waste deposits that are usually barren due to phytotoxicity of high-metal waste and, therefore, constitute secondary sources of pollution.

Another unresolved soil pollution problem is related to former military sites that received significant inputs of both organic and inorganic compounds over the time of their operation. They currently pose a risk to groundwater quality and biota.

It must be noted that in certain cases elevated metal contents in soil result from natural sources such as metal-rich parent rock material (e.g. high Pb dolomites or high Ni serpentine soils). Such soils are usually less toxic since the metals are mostly in non-bioavailable forms and such cases should not be treated as pollution.

8.4 Key indicators to assess soil contamination

The selection of soil indicator attributes should be based on: (i) land use; (ii) soil function; (iii) reliability of measurement; (iv) spatial and temporal variability; (v) sensitivity to changes in soil management; (vi) comparability in monitoring systems; and (vii) skills required for the use and interpretation (Nortcliff, 2002 cited by De la Rosa and Sobral, 2008).

8.4.1 Point contamination

Direct indicators are used to assess soil contamination. The effects of contaminants can also be measured indirectly by considering the indirect effects on soil functions, such as a decrease in biological activity (Huber *et al.*, 2008, de la Rosa and Sobral, 2008) (Table 8.2). However, these indirect effects are much more difficult to measure. Although there is no EU wide soil protection law, there are several national approaches to establishing indicators and threshold values. In Romania, the assessment of soil pollution is carried out according to Order 756/3.11.1997, which sets the typical values, alert thresholds and action levels for inorganic and organic pollutants, by type of land use. Germany has a similar approach (Bundes-Bodenschutz, 1999). The threshold values take the soil use into consideration, i.e. soils for agricultural use have lower threshold values than soils for industrial use. It is assumed that contents below the threshold values do not affect the soil functions and the environment. Other countries such as the Netherlands concentrate only on point pollution.

Table 8.2: List of indicators for soil pollution, according to Huber et al. (2008) and De la Rosa and Sobral (2008).

Topic	Problem	Indicator
Diffuse contamination by Inorganic pollutants	Which areas show critical heavy metal contents in excess of national thresholds?	Heavy metal contents in soils
Diffuse contamination by Inorganic pollutants	Are we protecting the environment effectively against heavy metal contamination?	Critical load exceedance by heavy metals
Diffuse contamination by nutrients and biocides	What are the environmentally relevant key trends in agricultural production systems?	Area under organic farming
Diffuse contamination by nutrients and biocides	Is the environmental impact of agriculture developing?	Gross nutrient balance
Diffuse contamination by persistent organic pollutants	Which areas show critical concentration of organic pollutants?	Concentration of persistent organic pollutants
Diffuse contamination by soil acidifying substances	How is the environmental impact of soil acidification developing?	Topsoil pH
Diffuse contamination by soil acidifying substances	Are we protecting the environment effectively against acidification and eutrophication?	Critical load exceedance by sulphur and nitrogen
Local soil contamination by point sources	How is the management of contaminated sites progressing?	Progress in management of contaminated sites
Local soil contamination by point sources	Is developed land efficiently used?	New settlement area established on previously developed land
Local soil contamination by point sources	How many sites exist which might be contaminated?	Status of site identification
Filtering function of soil	What is the impact on soil function?	Cation exchange capacity
Filtering function of soil	Is there a loss of organic matter?	Organic matter content
Filtering function of soil	What is the actual availability of pollutants for plants and animals?	Bioavailability of pollutants

Indicators for point contamination of inorganic pollutants, such as heavy metals, are well established taking into consideration the total or plant available content of the specific pollutant in the soil or critical loads (Huber *et al*.., 2008). The content of the pollutant in the soils is compared with the maximum tolerable content established by several national soil legislations and if they exceed the threshold values, soil remediation is required (e.g. Bundes-Bodenschutz, 1999).

It is more difficult to establish indicators for organic pollutants coming from point contamination sources as they are subject to decay which may result in the formation of more or less stabile metabolites. However, threshold values also exist for this kind of contamination in several national legislations; although not all potential organic pollutants are included (e.g. Bundes-Bodenschutz, 1999). For particulate pollutants such as nanoparticles and microplastics no threshold values are available and analytical techniques are still in the process of development.

8.4.2 Diffuse contamination

We distinguish between the small scale applications of agrochemicals to agricultural soils and diffuse contamination caused by deposition of airborne pollutants from industrial and urban sources. Concerning heavy metals, the same indicators can be used as for point contamination: content and critical load.

Organic and particulate pollutants from diffuse sources of industrial or urban activities are very difficult to assess because their distribution and the fall out is not easily known. Screening techniques are required to assess multiple organic pollutants in soils that enter the soil after long distance transport. Furthermore, for many of these pollutants no threshold value is defined and their effect on soil functions is unknown. At the moment, there are no monitoring programs or models available that can assess the actual contamination of soils with these types of pollutants.

The use of agri-environmental indicators play a crucial role in the development of policies aimed at sustainable and multifunctional agriculture (JRC, 2014). A total of 35 agri-environmental indicators have been defined and have provided conceptual background and initial methodological proposals for their development.

A simple methodology was selected for the Pesticide Soil Contamination indicator (JRC, 2014). The selected approach consists of calculating the quantity of herbicides in the soil profile based on the assumption of first-order degradation kinetics. The average annual quantity of herbicides present in soils under cereals, maize and sugar beet cultivation is computed based on an estimated average application rate, herbicide degradation properties, and average monthly temperatures (JRC, 2014). However, no model calibrations are available and the outcomes of the models can only be seen as rough estimations. Furthermore, no information is available about how high the pesticide contents are in European soils after 50 year of application and which mixtures of pesticides and their metabolites are actually present. We are far away from defining direct indicators such as threshold values, and the effect of single pesticides or their mixtures on soil functions is not sufficiently studied.

Europe wide monitoring programs are urgently required to assess the actual state of soil contamination with organic pollutants. The Reports of the *Soil Thematic Strategy: Pollution and Land Management* (Van Camp *et al.,* 2004) stated that there is a general need to achieve a greater harmonization in the quality of the information provided by the indicators, and in the data collection behind these indicators. This can be achieved by using standardised definitions, specifying the data that are required and the standardised methods of sampling and analysis (Van Camp *et al.,* 2004).

8.5 Methods to assess status of soil contamination

Under the EU Soil Framework Directive three steps are defined for the soil status report, which the following details: (a) the background history of the site, as available from official records; (b) chemical analysis determining the concentration levels of the dangerous substances in the soil, limited to those substances that are linked to the potentially polluting activity on the site; and (c) the concentration levels at which there are sufficient reasons to believe that the dangerous substances concerned pose a significant risk to human health or to the environment. The directive sets a common approach for monitoring soil contamination across all Member States but does not specify the methods. It is up to Member States to decide the best method based on local conditions and existing national approaches.

In addition, the Joint Research Center in their recent report “Progress in the management of Contaminated Sites in Europe” described four steps to characterize and assess soil contaminated areas, namely: 1) site identification (or preliminary studies), 2) preliminary investigations, 3) main site investigations, and 4) implementation of risk reduction measures. The first step refers to the mapping of sites where potentially polluting activities have taken place or are still in operation. Preliminary investigations and main site investigations considers the development of inventories and soil contamination assessments (Van Liedekerke *et al.*, 2014).

These assessments must take into account scientific and technical information that are available on each particular soil and climatic conditions, such as rainfall erosivity, length of the growing season, slope, soil infiltration and soil denitrification capacity (Anaya-Romero *et al.,* 2010).

Following Desaules (2012), we can consider two methods for soil contamination assessment, namely statistical and geochemical techniques. Statistical methods are of descriptive nature and summarize, describe and interpret data that is mostly accumulated in a database. Different statistical methods are needed depending on the information a researcher wants to provide, such as univariate analysis, multivariate analysis and geostatistics (GIS). Additionally, geochemical methods involve the measurement of the soil chemistry to determine contamination levels and abnormal chemical patterns (Zingg, 2014).

Furthermore, in order to standardize the information at the European level, it is important to identify the type of information and methods most commonly used in the soil assessment as a guide to data collection. In this sense, emerging technologies in data and knowledge engineering provides excellent possibilities for soil contamination assessment. This involves the development and linkage of integrated databases, evaluation models, and spatialization tools. Within this context, decision support systems for land evaluation such as MicroLEIS DSS (agro-ecological decision support system developed by CSIC-IRNAS and transferred to Evenor-Tech, www.evenor-tech.com) are considered very appropriate tools to include the soil and climatic attributes for a better identification of soil contamination and vulnerable zones and, eventually, for formulation of action programs.

Finally, future efforts in the area of soil contamination assessment will involve integrated methodologies that oncorporate all these different methodologies. Also, screening methods for the identification of pesticides and other organic pollutants resulting from diffuse sources are not available and their development is urgently needed.

8.6 Effects on of soil contamination other soil threats

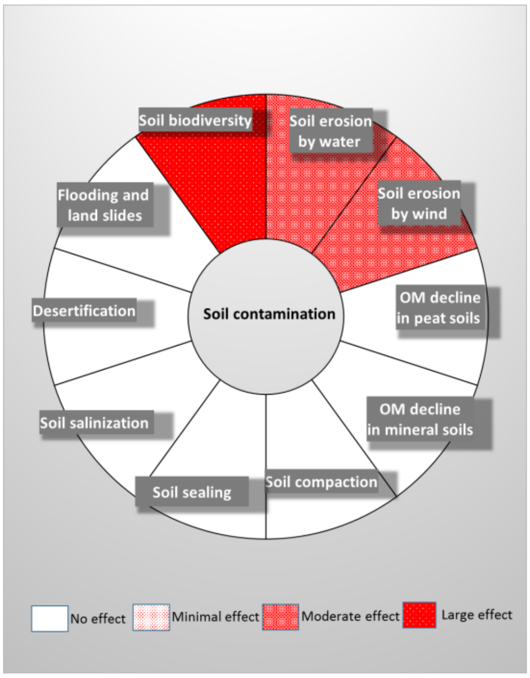

Figure 8.5: Effects of soil contamination on other soil threats. Red is negative effect.

Soil contamination strongly affects the other soil threats mentioned in this report (Fig. 8.5). The effects are based on processes occurring in soils caused by changes in soil properties.

Soil contamination leads to decreased activity of soil biota and to decreased biodiversity (Geissen *et al.,* 2010, Keesstra *et al.,* 2012) and therefore to a decline of aggregate stability and a decline in decomposition

Strong relationsships can also be seen between contamination and erosion. Decline in aggregate stability and organic matter caused by soil pollutants increase the erodibility and therefore the risk of wind and water erosion. On the other hand, pollutants may be transported off site by wind and water erosion related processes as solutes or particles and may pollute the connected aquatic environment or soils downslope. Landslides and flooding may cause the strongest off site transport of polluted soils.

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).

Furthermore, it is important to take into consideration the interactions between the different soil threats and to assess the cumulated pressure on soil degradation.

8.7 Effects of soil contamination on soil functions

Soil contamination affects soil functions in different ways:
1) Biomass production, including in agriculture and forestry. Soil contamination affects biomass production. Obviously, a contaminated soil looses the productivity and the capacity to support plants properly**.** Like all living organisms, plants are often sensitive to the deficiency of some heavy metal ions as essential micronutrient, while for the same ions excess concentrations are strongly poisonous to the metabolic activities. Research has been conducted throughout the world to determine the effects of toxic heavy metals on plant development and biomass production (e.g. Reeves and Baker, 2000).

2) Storing, filtering and transforming nutrients, substances and water; acting as carbon pool. Soil is not only part of the ecosystem but also the survival of the rest of the environment depends on its productivity. Soil functions such as filtering, buffering, storage and transformation systems protect against the effects of contamination. Low decomposition, resulting from harsh climate, acidic conditions, limited supply of essential nutrients and the presence of organic or inorganic pollutants, can lead to an accumulation of organic matter in the soil and to immobilization of essential nutrients (Swift *et al.,* 1979). Soil contamination by trace elements is a potential cause of disturbance of organic matter cycling in terrestrial ecosystems. Several authors have reported that free heavy metal and metalloids present in the ionic form at elevated concentrations in the soil solution may be toxic to the soil microflora (Pérez-de-Mora *et al.,* 2008). Moreover, these metals in the soil solution may inactivate extracellular enzymes responsible for the cycling of many nutrients (Kandeler *et al.,* 1996). They may thus limit the biodegradation of the organic matter and cause nutrient deficiency. In fact, many authors have observed an increase of litter accumulation near the sources of pollutant emission (Cotrufo *et al.,* 1995).

3) Biodiversity pool: habitats, species and genes. The diversity at different scales (from gene to ecosystem) of the organisms living in the soil is strongly affected by contamination. It has been reported that plant biodiversity decreased in polluted soils with high concentration of bioavailable trace elements (Madejón *et al.,* 2013). Soil microorganisms and soil microbial processes can also become disrupted by elevated concentrations of trace elements in soils (Giller *et al.,* 1998). As was indicated above, it is generally accepted that accumulated pollutants reduce the amount of soil microbial biomass (Chander *et al.,* 1995) and various enzyme activities, leading to a decrease in the functional diversity in the soil ecosystem (Kandeler *et al.,* 1996) and changes in the microbial community structure (Pennanen *et al.,* 1998). However, metal exposure may also lead to the development of metal tolerant microbial populations (Ellis *et al.,* 2003).

4) Physical and cultural environment for humans and human activities. Soils affect human health, and in turn humans affect soil health. Both soil and humans must be in a state of well-being with respect to their physical, chemical, and biological characteristics. Much is known about how human activity can improve or detrimentally affect soil health, but how soils can beneficially or adversely impact human health is less well documented (Pepper*,* 2014).

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9 SOIL SALINIZATION

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9.1 Descriptions of soil salinization

Salinization is the accumulation of water-soluble salts in the soil solum (the upper part of a soil profile, including the A and B horizons) or regolith (the layer or mantle of fragmental and unconsolidated rock material, whether residual or transported) to a level that impacts on agricultural production, environmental health, and economic welfare (Rengasamy, 2006). According to van Beek and Tóth (2012), soil (and groundwater) salinity is often used as a comprehensive term to refer to several different salinity forms. These forms are known under the names of, respectively, (1) saline soil, that have elevated salt concentrations, (2) sodic (or alkali) soil, with a disturbed monovalent/divalent cation ratio in favour of the monovalent alkali cations (Na, K), and (3) alkaline soil, for which the chemical composition is disturbed towards alkaline (high pH) compositions and often due to a dominance of (bi)carbonate anions in solution. These three salinity issues may be related, but this needs not be the case (Bolt and Bruggenwert, 1976). According to (Rengasamy, 2006), a soil is considered saline if the electrical conductivity of its saturation extract (ECe) is above 4 dS m1 (Richards, 1954). However, the threshold value above which deleterious effects occur can vary depending on several factors including plant type, soilwater regime and climatic condition (Maas, 1986; Rengasamy, 2002).

Salinity usually becomes a land use issue when the concentration of salt or sodium adversely affects plant growth (crops, pastures or native vegetation) or degrades soil structure. It becomes a water issue when potential uses of water are limited by its salt content. The adverse consequences of salinity generally vary, depending on the form and stage of salinization; in early stages it affects the metabolism of soil organisms and reduces soil productivity, but in advanced stages it kills all vegetation and consequently transforms fertile and productive land to barren land (Chesworth, 2008; Jones *et al.*, 2012; Tóth *et al.*, 2008). As such, it is a major factor limiting crop production and land development in coastal areas (Li *et al.*, 2012; Sparks, 2003).

The mechanisms of the toxic effects of salinity to crop growth can be described by various theories (Guo, 2010) including osmotic inhibition (Koorevaar *et al.*, 1983), plant mineral nutrition imbalance (Verbruggen and Hermans, 2013), saline ion toxic action (Munns, 2005, 2002), and nitrogen metabolism impediment theory (e.g. Lovatt, 1986). Relevant studies have demonstrated that saline ion concentrations in soil can result in physiological hydropenia phenomenon, reduced nutrient absorption, plant dysplasia, output reduction, and death (Bernstein, 1963). Na+ and Mg2+ can destroy cytoarchitecture, restrain plant photosynthesis, and reduce chlorophyll production (Guo, 2010). In addition, soil saline ions can produce some toxic intermediates in the process of nitrogen metabolism, which may hinder metabolic process (Epstein, 1980). For alkaline soil, toxicity and deficiency effects due to altered plant availability of elements is also the main problem.

For sodic soil, the structural degradation caused by too large concentrations of sodium (Na) is generally most important. As sodium salts are leached through the soil, some sodium remains in the soil bound to clay particles, displacing other cations such as calcium. A high proportion of exchangeable sodium attached to clay mineral exchange sites weakens the bonds between soil particles when the soil is wetted. As a result, the clay particles swell and often become detached and disperse. A soil with increased dispersibility becomes more susceptible to erosion by water and wind. Sodic soils become dense, cloddy and structureless on drying because natural aggregation is destroyed. The dispersed clay at the soil surface can act as cement, forming crusts that are relatively dense and hard but typically thin (up to 10 mm thick). The crust impedes seedling emergence and can tear seedling roots as it dries and shrinks. The degree of crusting depends on the soil textural composition, the mineralogy of the clay, the exchangeable sodium content, the energy of raindrop impact, and the rate of drying. Soils with high montmorillonite clay contents will crack on drying. Moreover, the genesis of some soils has resulted in sodic subsoils, often with a columnar structure. Sodic subsoils may be dense, with reduced soil water storage, poor aeration and increased soil strength, and can be susceptible to tunnel erosion. The small clay particles move through the soil, clogging the pore spaces thus reducing hydraulic conductivity.

9.2 State of soil salinization

The Food and Agriculture Organization of the United Nations (FAO) assessment in 2011 showed that saline and sodic soils are widespread and affect millions of hectares of land all over the world. Different estimates have been produced showing that a significant percentage of salt affected soils are human induced. Globally, 34 Mha is believed to be impacted. Major problems have been reported in Pakistan, China, United States, India, Argentina, Sudan and many countries in Central and Western Asia (FAO, 2011; Mateo-Sagasta and Burke, 2011).

According to Stanners *et al.* (1995), salinisation affects around 3.8 Mha in Europe. Using expert judgement, the GLASOD study (van Camp *et al.*, 2004) on soil degradation at global scale assessed that approximately 4 Mha of soils of Europe have a moderate to high level of degradation by salinisation. Naturally saline soils occur in Spain, Hungary, Slovakia, Greece, Austria, Bosnia, Serbia, Croatia, Romania, Bulgaria, Ukraine and the Caspian Basin (Geeson *et al.*, 2003; Jones *et al.*, 2008; Tóth *et al.*, 2008; van Beek and Tóth, 2012; van Camp *et al.*, 2004). On the other hand artificially induced salinisation is affecting significant parts of Italy (e.g. Campania and Sicily), Spain (e.g. the Ebro Valley), Hungary (e.g. Great Alfold), Greece, Cyprus, Portugal, France (West coast), the Dalmatian coast of the Balkans, Slovakia and Romania. In addition, North Europe countries (e.g. Denmark, Poland, Latvia, and Estonia) are facing similar issues. North-western Europe (e.g. Western Netherlands, Belgium, North-eastern France, and South-eastern England) is another territory that is affected by soil salinsation which is mainly caused by sea-level rise and surface seawater seepage (Geeson *et al.*, 2003; Tóth *et al.*, 2008). Several studies have shown that salinisation levels in soils in countries such as Spain, Italy, Greece, Cyprus and Hungary are increasing but, systematic data on trends across Europe are not available. JRC (IES) has recently developed an updated version of the Soil Geographical Database of Europe (SGDBE) which among other threats presents the limitations to agricultural use posed by salinity and sodicity (*Figure 9.*).

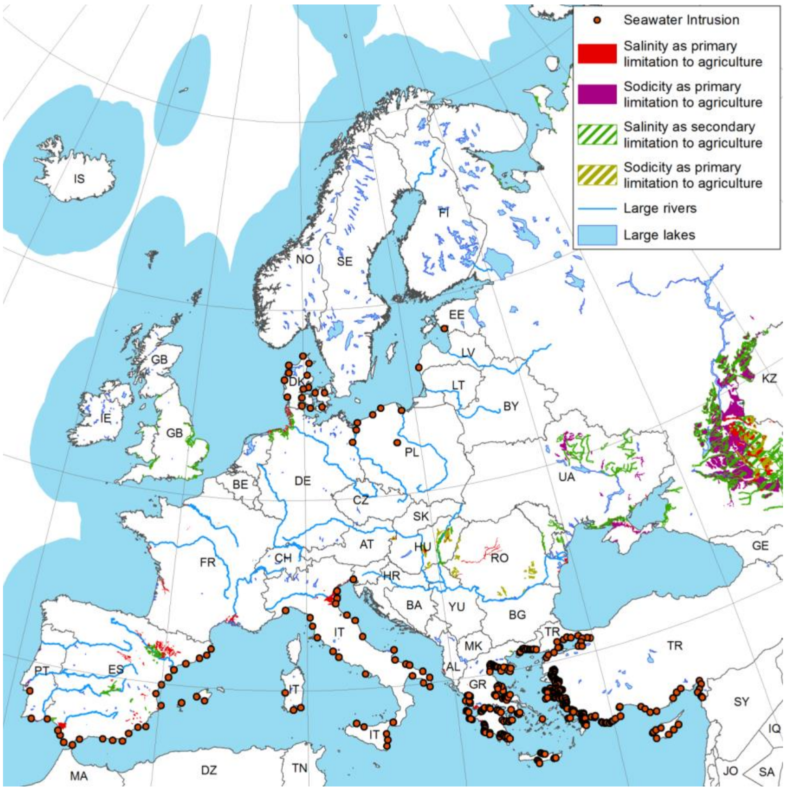

Figure 9.1: Saline (EC > 4 dScm^{-1} within 100 cm soil depth) and sodic (Na/T > 6% within 100 cm soil depth) soils as primary and secondary limitations to agricultural use and areas of seawater intrusion in the European Union. Compiled from Daskalaki and Voudouris (2008), EEA (1999) and Soil Geographical Database of Europe (Le Bas et al., 1998).

Soil salinity that affects mainly the Mediterranean countries is regarded as a major cause of desertification and is therefore a serious form of soil degradation. Along the Mediterranean coast the problem of soil salinity is increasing due to scarcity of precipitation and irrigation with low quality water. Saline soils are present mainly due to human activities, especially with the extension of irrigation and undisciplined use of saline water which has caused over-pumping, and the consequent sea-water infiltration into the groundwater layer. In the Mediterranean area 25% of irrigated cropland is affected by moderate to high salinisation leading to moderate soil degradation (Geeson *et al.*, 2003; Mateo-Sagasta and Burke, 2011). More specific in Spain about 3% of the 3.5 Mha irrigated land is severely affected, significantly reducing its agricultural potential and another 15% is at serious risk. In Greece about 30% of the approximately 0.5 Mha of irrigated land is affected by soil salinization (Geeson *et al.*, 2003; van Camp *et al.*, 2004). Figure 9. depicts the locations of saltwater intrusion, compiled from Daskalaki and Voudouris (2008) and EEA (1999). In addition to sea water intrusion, in several areas like Cyprus, the excess use of fertilizers and municipal wastewater has contributed to the soil salinity (FAO, 2011; Geeson *et al.*, 2003; Huber *et al.*, 2008; Mateo-Sagasta and Burke, 2011). Furthermore, projected temperature increases and changes in precipitation characteristics in the Mediterranean (e.g. Koutroulis *et al.*, 2013) are likely to enhance the problem of salinisation.

Though difficult to estimate, studies in 3 countries (Spain, Hungary and Bulgaria) have demonstrated annual costs of soil salinisation due to mainly agricultural yield losses, but also damages to infrastructure and the environment in the range of 158 – 321 M€ (Montanarella, 2007). A more recent study (Bosello *et al.*, 2012; Richards and Nicholls, 2009) focused on selected rivers and deltas, estimates that the current EC economic impact exclusively in agriculture due salinity is in the area of 600 M€ (mostly borne by Germany, the Netherlands and France), assuming that saline agricultural land is half as valuable as is non-saline land.

9.3 Drivers and pressures

There are two groups of salinization driven by climate and human activities that lead to salinity. (i) Primary salinization involves accumulation of salts through natural processes such as physical or chemical weathering and transport from parent material, geological deposits or groundwater. (ii) Secondary salinization is caused by human interventions such as use of salt-rich irrigation water or other inappropriate irrigation practices, and/or poor drainage conditions (Tóth *et al.*, 2008).

(i) Primary salinization as induced by natural processes

Figure 9.2 shows primary soil salinization as induced by natural processes. Primary salinization of soils is closely r elated to the long-term accumulation of salts in the soil profile. Nevertheless, it can also occur as a result of the one-time submergence of soils under seawater. During this period, seawater fills the voids of the sediments (connate water) and remains trapped inside the marine deposits (e.g. fine-grained sands and clays dating from the Pliocene and Early-Pleistocene), even after the seawater incursion. Soil may also be rich in

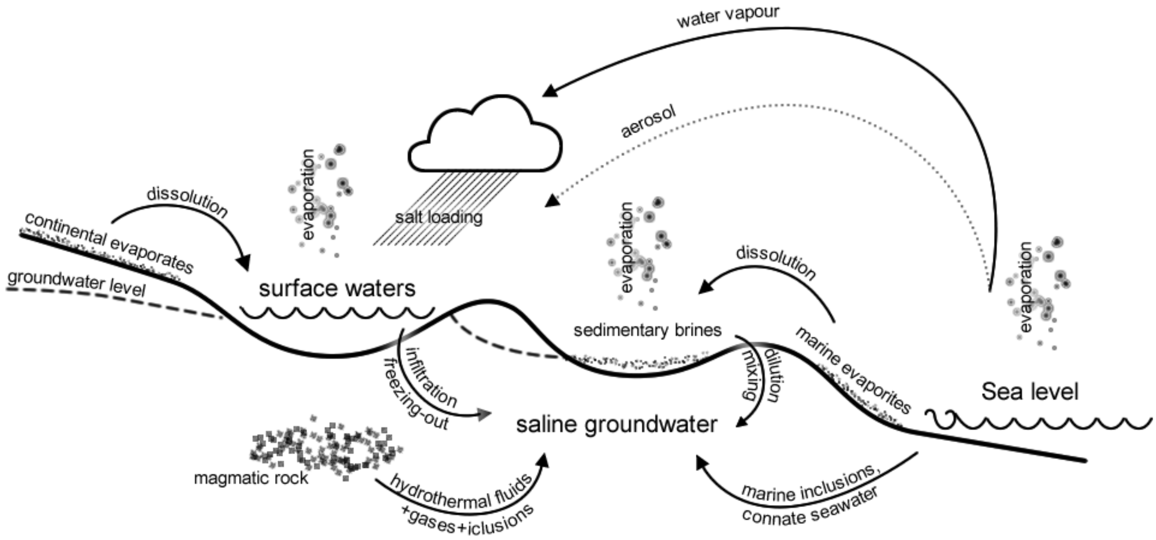

Figure 9.2: Primary soil salinization mechanisms.

salts due to parent rock constituents such as carbonate minerals and/or feldspar. Closely related to this, geological events or specific formations can increase the concentration of salts in groundwater and consequently in soils. This can occur when saline groundwater rises (capillary effects or evapotranspiration) and salts dissolved in the soil moisture remain behind after evaporation of the water and accumulate at or near the surface (Chari *et al.*, 2013; Geeson *et al.*, 2003). These drivers are affecting the soil depending on sequence and thickness of aquifers and the vertical and horizontal transmissibility of geological layers, vertical and horizontal natural drainage conditions to leach the soil, the structure (stability; cracking, shrinkage – swelling characteristics), texture, clay mineral composition; compaction rate – porosity, infiltration rate, water storage capacity, water retention, saturated and unsaturated hydraulic conductivity and finally potential salt content (Chesworth, 2008; Eckelmann *et al.*, 2006; van Beek and Tóth, 2012).

Besides historical marine waters, contemporary sea level rises may cause flooding of coastal land by seawater, either for a long period (marine transgressions) or a short one (storm flood events, tsunamis). In addition, they boost lateral seawater intrusion into coastal aquifers that are hydraulically connected to the sea, fact that in long-term through irrigation causes soil salinisation. Sea level rise also induces seepage into areas lying below sea level (i.e. Netherlands, because of little or no drainage routs) (Geeson *et al.*, 2003; Tóth *et al.*, 2008; van Weert *et al.*, 2009).

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 consequent to the evaporation. This fact is very important in Mediterranean regions in which evaporation reaches 8-10 mm day^{-1}. 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)

A future warmer climate will cause variations in the hydrological circle (Sterling *et al.*, 2012; Vautard *et al.*, 2014) and rising sea levels (Hinkel *et al.*, 2014) and in turn will significantly increase soil salinity resulting in the expansion of the affected areas. Especially areas with reduced rainfall and increased evapotranspiration (e.g. Koutroulis *et al.*, 2013), will face a reduction of the extent of their water courses and a transition to a more arid environment (Koutroulis *et al.*, 2011; Vrochidou *et al.*, 2013). Irrigation water consumption is generally projected to increase with higher global mean temperature (Haddeland *et al.*, 2013) and is likely to have an even higher salt content, due to concentration following evaporation, again promoting soil salinisation and desertification. An intensified hydrological circle may also trigger an increase of floods and flash floods (Hallegatte *et al.*, 2013; Tsanis *et al.*, 2011), thus causing an increased release of dissolved salts into the soil in areas with saline geological substrates (Mateo-Sagasta and Burke, 2011; Trnka *et al.*, 2013; van Weert *et al.*, 2009) .

(ii) Secondary salinization as induced by human activities
Soil salinisation is mainly associated with the over-exploitation of groundwater caused by the demands of society for economic growth and the public policy regarding growing urbanisation, industry and agriculture. However, agriculture plays the major role in driving the soil salinisation phenomenon, by causing high water groundwater consumption and water chemical degradation, but at the same time the socioeconomic sector is the one that urges intense agricultural practices and production (Geeson *et al.*, 2003; Tóth and Li, 2013).

Coastal protection to reduce the encroachment of sea water into the aquifers may block natural drains of discharged water rich in salts. In addition another factor that may lead to soil salinisation in semiarid regions is raising the water table due to filtration from unlined canals and reservoirs, uneven distribution of irrigation water, poor irrigation practices, land clearing, and improper drainage. These allow water to pond for long periods and allowing seepage from irrigation channels, drains and water storages. This increases leakage to the groundwater system, causing the watertable to rise, which may mobilise salt that has accumulated in the soil layers. Poorly drained soils, also allow for too much evaporation leading to salt residuals on the soil surface (Geeson *et al.*, 2003; Mateo-Sagasta and Burke, 2011; van Beek and Tóth, 2012).

Soil salinisation is related to and affected by irrigation when it occurs with waters rich in salts. Excessive irrigation can degrade water bodies and soils by dissolving and transporting chemicals or substances (i.e. salts). Salinisation is often associated with irrigated areas where low rainfall, high evapotranspiration rates or soil textural characteristics impede the washing of salts out of the soil, which subsequently build up in the surface layers. Irrigation tends to increase gradually the salinity levels in soil water, surface water systems and/or aquifers. This is because the crop evapotranspiration leaves a residue of dissolved substances in the soil. These effects are most pronounced under arid conditions (Maas *et al.*, 1999; Tóth *et al.*, 2008). Waterlogging practice has also become a serious cause of soil salinisation. Waterlogging refers to the saturation of soil with water, thus, when the water table of the groundwater is too high. In irrigated agricultural land, waterlogging irrigation or using canals especially in arid and semi-arid regions is often accompanied by soil salinity as waterlogged soils prevent leaching of the salts imported by the irrigation water (Chesworth, 2008; Eckelmann *et al.*, 2006). Soil salinisation from irrigation depends on the quality and salt concentration of the water used and the nature of the soils, i.e. the damage caused by using saline water increases in particular in high clay-content soils. By contrast, the rainfall pattern in the rainy season is very important because it may be conducive to the leaching of the salts from irrigation water. The excessive use of water for irrigation in dry climates, with heavy soils, causes salt accumulation because they are not washed out by rainfall. The process occurs in cultivated areas where irrigation is associated with high evaporation rates and a clay texture of the soil. In this context salt leaching is scarce or absent and sodium magnesium and calcium ions accumulate in the soil surface layers. Moderate soil salinisation is reported even in areas irrigated with "good" quality water depending on irrigation methods and aridity conditions. In reality, constant or increasing soil salinity is chiefly caused by the use of highly saline irrigation water, compounded by excessive evapotranspiration in dry areas (Dubois *et al.*, 2011; Geeson *et al.*, 2003; Mateo-Sagasta and Burke, 2011; van Camp *et al.*, 2004).

Salty groundwater may also contribute to salinisation. When the water table rises (e.g. following irrigation in the absence of proper drainage), the salty groundwater may reach the upper soil layers and, thus, supply salts to the root zone. In addition use of fertilisers and other inputs in association with irrigation and insufficient drainage cause soil salinisation, especially where land under intensive agriculture has low permeability and limited possibilities of leaching. Finally soil salinisation occurs through irrigation via vegetation growth. When crops use water, salts are left behind in the soil and eventually begin to accumulate unless there is sufficient seasonal rainfall (usually in the winter months) to flush out the salts (Chesworth, 2008; Maas *et al.*, 1999; Mateo-Sagasta and Burke, 2011).

Factors leading to soil salinisation associated to waste disposal are the subsurface injection of saline water from industrial operations, the operation of waste disposal sites, use of wastewaters rich in salts for irrigation, salt-rich wastewater disposal on soils, contamination of soils with salt-rich waters and industrial by-products. Other human factors that can induce soil salinisation are the discharge of saline water to rivers from industries and mining activities. In addition, periodic application of de-icing agents in snow-belt regions of industrialized countries contributes to the accumulation of salt in the soil and water (Eckelmann *et al.*, 2006; Jones *et al.*, 2012; van Beek and Tóth, 2012).

Policy drivers

It is a common fact that policies permitting unsustainable use of resources and lack of infrastructures are major contributors to land degradation. Policy instruments against soil salinization can be applied at different levels of authority and management. At the European level, the 5th Environment Action Programme (EAP) legislation in the late 90s has set environmental objectives that are built up on scientifically sound-based action plans that integrate scientific disciplines, policies, and stakeholder consultations and has helped ensure that these objectives are backed by environment legislation. Within the 5th EAP, the Water Framework Directive (WFD) 2000/60/EC[1] established a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater, including relevant information on the superficial deposits and soils at catchment scale, thus illustrating the importance of the holistic approach of soil and water management as well as the data collection. Nevertheless, the WFD treats soil merely as a medium of achieving "good status" of all waters, mostly concerning point and diffuse pollution sources which may affect the aquatic ecosystems (Quevauviller and Olazabal, 2003), thus overlooking the essential functions and services it provides. Tackling this shortcoming, the 6th EAP (2002 - 2012)[2] established the Soil Thematic Strategy[3] aiming specifically at preventing and diminishing the soil degradation and threats. Later, the Thematic Strategy for Soil Protection[4] recognised salinisation among all other soil threats and in 2012[5] the EU recognised the increasing soil degradation trends and structured its strategy on the pillars of awareness raising, research, integration, legislation. It is important to note that salinisation can pose a major risk for the long-term objectives of the Common Agricultural Policy (CAP) ("viable food production, sustainable management of natural resources and climate action and balanced territorial development") and provision in the new CAP's "targeted agri-environment schemes" has been proposed by several NGOs (BirdLife International *et al.*, 2009). Nevertheless, salinisation is never mentioned but only implied even in the current CAP's Good Agricultural and Environmental Conditions (GAEC)[6]. In the 7th EAP that came into force in 2014 and will be guiding the European environment policy until 2020, fertile soil and the productive land are considered part of the "natural capital" to be managed sustainably and adequately protected, while action for the remediation of contaminated areas, reduce soil erosion and increase soil organic matter is encouraged. These policy and soft

[1] Council Directive 2000/60/EC establishing a framework for Community action in the field of water policy, OJ L 327, 22 December 2000, p.72
[2] Official Journal of the European Communities, Od L 242, 10 September 2002, p.81
[3] Communication on Towards a Thematic Strategy for Soil Protection, COM(2002)179 final
[4] Communication on the Thematic Strategy for Soil Protection (COM(2006) 231 final)
[5] Communication on the The implementation of the Soil Thematic Strategy and ongoing activities (COM(2012) 46 final)
[6] REGULATION (EU) No 1306/2013 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 17 December 2013 on the financing, management and monitoring of the common agricultural policy and repealing Council Regulations (EEC) No 52/78, (EC) No 165/94, (EC) No 2799/98, (EC) No 814/2000, (EC) No 1290/2005 and (EC) No 485/2008

law texts indicate the intention of EU for further and more specific protection of the soil, nevertheless a hard law text (directive, regulation) is vitally important in order to set the limit values of the salinisation soil threat.

9.4 Key indicators of soil salinization

The ENVASSO Project (Contract 022713) identified three major indicators for soil salinisation: (a) Salt profile where Soil Salinisation is assessed in Total Salt Content [%] and Electrical Conductivity [dS m-1], (b) Exchangeable Sodium Percentage (ESP) [%] to assess sodification and (c) the potential salt sources (groundwater or irrigation water) and vulnerability of soils to salinisation/sodification measured in Salt content [mg l-1] or SAR [dimensionless]. Here we discuss some of them and also consider additional indicators:

Electrical Conductivity of solution, (EC): Electrical conductivity (EC) is a measure of the concentration of all the soluble salts in soil or water. EC is measured in deci-Siemens per meter (dS m−1) at 25°C to avoid the influence of temperature. From the saline soil definition point of view, it is most consistent to measure EC at field capacity (EC_f) as this provides the soil's true salt concentration. However, determining salinity in a standard saturation extract (EC_e) obtained by adding water to a dry soil is more practical than extracting sufficient soil water from soil samples at field capacity. The relationship between EC_f and standard saturation extract EC_e depends on soil structure. Typically, a 1:n solution (1 part soil n parts distilled water were n is also typically 1, 2.5 or 5) is prepared from field soil samples using one many standard procedures (He *et al.*, 2012) and EC_(1:n) measurements are converted to EC_e depending on soil texture using tables (e.g. Dahnke and Whitney, 1988) or regression equations (e.g. Sonmez *et al.*, 2008). The derived EC_e can be used to compare across different soils and is classified depending on the salinity hazard and its effects on the yield of field crops according to the general scheme of Richards (1954) presented in Table 9.1.

Table 9.1: Classification of electrical conductivity with regard to salinity effects on crops (Source: Richards (1954)).

EC (d S m^{-1})	Class	Effect
0-2	Non saline	Negligible
2-4	Mildly saline	Yield reduction of sensitive crops
4-8	Medium saline	Yield reduction of many crops
8-12	Very saline	Normal yields for salt tolerance crops only
>16	Extremely saline	Reasonable crop yield for very tolerance crops only

Sodium Adsorption Ratio, (SAR) and Exchangeable Sodium Percentage, (ESP): Sodicity is a measure of sodium ions in soil or water relative to calcium and magnesium ions. It is expressed either as sodium adsorption ratio (SAR) or as exchangeable sodium percentage (ESP). SAR is a measure of the suitability of water for use in agricultural irrigation, as determined by the concentrations of solids dissolved in the water and a measure of the sodicity of soil, as determined from analysis of water extracted from the soil (Shahid *et al.*, 2013; van Beek and Tóth, 2012), given by:

$$SAR = \frac{Na^+}{\sqrt{\frac{1}{2}(Ca^{2+}+Mg^{2+})}} \qquad [9.1]$$

where the concentrations of Na^+, Ca^{2+} and Mg^{2+} are in milliequivalents per liter (meq L^{-1}) in soil extract from saturated paste, and SAR is expressed as (mmoles $L^{-1})^{0.5}$.

Every soil has a definite capacity to adsorb the positively charged constituents of dissolved salts, such as calcium, magnesium, potassium, sodium, etc., termed as the cation exchange capacity. The various adsorbed cations can be exchanged one for another and the extent of exchange depends upon their relative concentrations in the soil solution, the valency and size of the cation involved, the nature and amounts of other cations present in the solution or the exchange complex, etc. ESP is the amount of adsorbed sodium on the soil exchange complex expressed in percent of the cation exchange capacity in milliequivalents per 100 g of soil. The ESP is calculated by the relationship, (Shahid *et al.*, 2013):

$$ESP = \frac{\text{Exchanagable } Na \left(\text{mC}F/_{100\text{ g } soil}\right)}{\text{Cation exhange capacity} \left(\text{mC}F/_{100\text{ g } soil}\right)} \cdot 100 \qquad [9.1]$$

If the SAR of the soil equals or is greater than 13 (mmoles L−1)0.5 or ESP equals or is greater than 15, the soil is termed sodic (Richards, 1954).

Total Dissolved Solids, (TDS): TDS is a measure of the total ionic concentration of dissolved minerals in water. This indicator is directly related to EC in irrigation water. An important classification of EC and TDS is that of USDA Salinity Laboratory (Richards, 1954; **Error! Reference source not found.**), that is still commonly used.

Table 9.2: Classification of the electrical conductivity (EC) and (TDS) of water with regard to the salinity hazards (Source: Richards (1954))..

EC (d S m^{-1})	TDS (ppm)	Salinity Hazard
0-0.25	<160	Low – water use is safe
0.25-0.75	160-480	Medium – water quality is marginal
0.75-2.25	480-1470	High – water unsuitable for use
>2.25	>1470	Very High

pH: pH is a measure of the acidity or alkalinity of the soil. Specifically, if pH is greater than 8.5 the soil is more likely to be saline – alkaline. **Error! Reference source not found.** represents salinity/alkalinity/sodicity classification schemes for the above commonly used indicators.

Table 9.3: Salt-affected soils classification scheme (van Beek and Tóth, 2012).

Soil type	Soil property			
	EC (d S m^{-1})	SAR	ESP	pH
Non saline, non-sodic	<4	<13	<15	<8.5
Saline	>4	<13	<15	<8.5
Sodic	<4	>13	>15	>8.5
Saline – Sodic	>4	>13	>15	>8.5

Remote Sensing indices for detecting salinity: Under salinity stress, plant health is hindered showing symptoms similar to that of water deficit (Hamzeh *et al.*, 2013). Numerous studies have shown that vegetation reflectance and hence remote sensing vegetation indices may be used as a proxy for soil salinity estimation. Special attention is given to specific water absorption bands that can determine the leaf and canopy water content and relate it to the soil salinity. Literature shows the ability of the indices that include the water absorption bands in the SWIR (short-wave infrared wavelength bands) and NIR (near infrared wavelength bands) in detecting water and salinity stress in agricultural fields (Ceccato *et al.*, 2001; Leone *et al.*, 2007; Poss *et al.*, 2006; Zhang *et al.*, 2011). High salt concentrations can be identified through the existence of characteristic vegetation types (e.g. halophytes) and growth patterns or by the salt efflorescence and crust that are present on bare soil. Similar to vegetation indices, different salinity indices exist for detecting and mapping soil salinity from multispectral (low cost or free) and hyperspectral (higher resolution information) satellite sensors (Dehaan and Taylor, 2002). Nevertheless, surface reflectance is highly affected by soil moisture content, salt content, color and roughness. Hamzeh *et al.* (2013) and Allbed and Kumar (2013) have documented a range of remote sensing indices relevant to salinity stress, some of which date since the advent of remote sensing in 1975 and others appear in very recent publications. These indices have been applied with a varying degree of success, thus demonstrating that a single selected index may not be suitable for all cases. A no-regrets and robust index seems to be the Normalized Difference Vegetation Index (NDVI) that can quickly be used to assess vegetation health spatial patters. Using NDVI as a first indicator, statistical methods such as principal component analysis (PCA) can be used to correlate soil properties and different indices.

9.5 Methods to assess status of soil salinization and state of degradation

(i) Monitoring and Mapping methods

Monitoring and mapping soil salinity is crucial for effective adaptation and mitigation through land reclamation actions. The appropriate mapping methods are directly related to the spatial scale of interest. Monitoring at farm or field scale can be accomplished through local salinity sensors and sampling or non-invasive techniques like electromagnetic induction, however regional or greater level assessments are based on remote sensing and geographic information systems coupled with ground measurements

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Remote observations using satellite sensors and aerial photography offers efficient techniques for salinity mapping and monitoring, outperforming traditional ground methods at large spatial scales. The remote detection of soil salinity can be performed directly through salt features on the soil surface in the visible spectrum (Farifteh *et al.*, 2008), or through multispectral/hyperspectral remote sensing indices that depict soil properties or vegetation health that can serve as a proxy. Geophysical measurements (Metternicht and Zinck, 2003) such as airborne electromagnetic, magnetic, and gamma-ray sensors also have the ability to directly map subsurface soil information when combined with ground data. Multi-scale integrated assessments that uses a combination of remote sensing, field data and various modeling approaches can improve the development of soil salinization risk maps useful to land managers and users (Bouksila *et al.*, 2013; Douaoui *et al.*, 2006; Farifteh *et al.*, 2006; Metternicht and Zinck, 2003). Finally, macroscopic maps of salt affected soils at global scale (Li *et al.*, 2014; Szabolcs, 1985) illustrate the extent of the environmental problem, especially in coastal areas.

The need of harmonized soil mapping and monitoring (Eckelmann *et al.*, 2006; Morvan *et al.*, 2008; van Beek *et al.*, 2010) at a Pan-European level motivated the initiation of several projects (Kibblewhite *et al.*, 2008; van Beek and Tóth, 2012). The ENVASSO – Project (Contract 022713) was a recent coordinated effort for monitoring harmonization, by defining and evaluating (Stephens *et al.*, 2008) top indicators (Huber *et al.*, 2008) for salinization at field level and geo-statistical upscaling at regional, national (Arrouays and Forges, 2008) and European level (Morvan *et al.*, 2008) based on specific procedures and protocols (Jones *et al.*, 2008).

(ii) Modelling

Salinity is a dynamic and transient condition in saline soils. Chemical reactions in root zone (solubility, precipitation, cation-exchange reactions) in irrigated fields affect soil salinity and sodicity and salt contribution to drainage water. Many studies used models to evaluate salinity, sodicity, and environmental hazards of drainage water that resulted from irrigation (Oster and Rhoades, 1975; Rhoades and Suarez, 1977; Shahid *et al.*, 2013) and others calculate the effect of chemical reactions in the soil solution composition for transient conditions within the root zone (Jury *et al.*, 1978; Robbins *et al.*, 1980). Models could be simple or of great complexity. A major constraint to these models is usually the lack of input data (Ranatunga *et al.*, 2008).

An additional challenge when using modelling under saline conditions is due to the dynamic nature of salinity problems which should be clearly understood by the model users. Physically based models simulating water and solute transport represent an essential tool for predicting soil salinity and/or sodicity. These models enable different options to be compared to develop strategies for sustainable irrigation in the short and in the long term. However, calibration and validation of these models against soil and crop field data is needed to check accuracy of the predicted values before these models can be used to develop reliable management scenarios (Shahid *et al.*, 2013).

In principle, it is quite easy to develop an equation that calculates the Leaching Requirement (LR). Leaching requirement can be defined as the fraction of infiltrated water that must pass through the root zone to keep soil salinity from exceeding levels that would significantly reduce crop yield under steady-state conditions. This concept can be formulated in terms of easily measurable properties (Rhoades, 1974), such as the water content of soil at field capacity and in the saturated paste, which are quite robust measures. Hence, also LR is quite a robust risk assessesment method for soil salinization and can be defined by the following equation (van Beek and Tóth, 2012):

$$LR = \frac{D_{DW}}{D_{IW}} \approx \frac{w_{FC}}{w_{SP}} \cdot \frac{EC_{IW}}{EC_e} \qquad [9.3]$$

where D denotes an amount of water (mm/year), w stands for water content by weight, and ECe is the soil salinity electrical conductivity. Subscripts DW, IW, FC, and SP denote drainage water, irrigation water, field capacity of soil, and saturation extract, respectively. Finally, the asterisk denotes that the electrical conductivity of the saturated paste may not exceed this particular value (Corwin *et al.*, 2007).

Despite its simplicity, the LR concept is a robust way to convince stakeholders of the need for drainage and has motivated research on drainage improvements. In some cases, for instance if more factors of interest need to be taken into account, the complexity of required modelling is higher and for that purpose, various models have been developed. Corwin *et al.* (2007), considered the leaching requirement as defined above, with more complicated models such as WATSUIT and TETrans. They found that transient modelling with the mentioned models may lead to leaching requirements that are smaller than the steady state LR concept given above. Strongly focussed towards salinity/sodicity type of problems is UNSATCHEM (Shahid *et al.*, 2013;

Šimůnek *et al.*, 1996). This advanced code has been used successfully to understand both salinity and sodicity process dynamics at a very local scale (Jalali *et al.*, 2008). Advantage of this code is that boundary conditions can be variant in time, whereas flow and transport are both transient. Obviously, the demands regarding computation, model parameterization, and expertise of the modelers are much larger than for applying the LR concept. Other software, such as LEACHM, PHREEQC, HYDRUS, and ORCHESTRA are less focussed to soil salinity issues (van Beek and Tóth, 2012).

In the First Expert Consultation on Advances in Assessment and Monitoring of Salinization for Managing Salt-Affected Habitats (Aquastat, 2009), it was concluded that salinity models could be of limited use if they are not well designed and some models can be very vulnerable to particular parameters if not properly developed.

Models that incorporate all governing elements of nature such as soils, water, crops, and agrometeorology produce better results as they represent the nature to a large extent. The limitations of above software are commonly the same: they are complex, much a priori knowledge is required from the modeler, and important feed backs have been ignored. In a recent overview of the state-of-the-art of modelling a serious gap is identified between model complexity and the demands for application by the irrigation and drainage community (Shahid *et al.*, 2013). As soil salinisation is a long-term process, long duration experiments, as well as robust comprehensive are required for long-term predictions. Models that can be used for a variety of irrigation systems, soil types, soil stratification, crops and trees, water application strategies (blending or cyclic), leaching requirement, and water qualities are lacking. The SALTMED model has been developed to meet these challenges. The SALTMED model includes the following key processes: evapotranspiration, plant water uptake, water and solute transport under different irrigation systems, drainage and the relationship between crop yield and water use (Ragab, 2002).

9.6 Effects of salinization on other soil threats

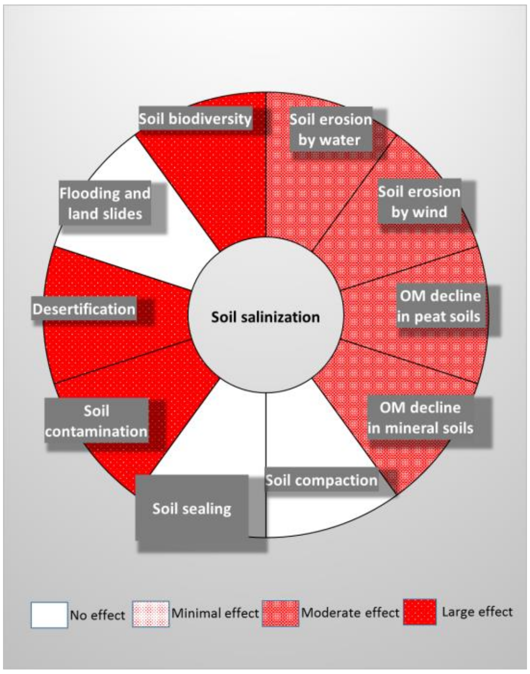

Figure 9.3: Effects of decline in OM for mineral soil on other soil threats. Red is negative effect.

As salinity is responsible for the structural collapse of soil aggregates into their components (Li *et al.*, 2012) it is closely linked to other soil degradation issues (Fig. 9.3). Salinity is often associated with prolonged wetness and lack of surface cover and therefore increases the vulnerability of soils to erosion. Scalding can occur when wind and water erosion removes the top soil and exposes saline or sodic soils. However, water or wind moving over the surface will remove more soil, and contribute to sheet, rill and gully erosion. Erosion also tends to remove the lighter, smaller soil particles first (such as clay and silt), leaving fine and coarse sand behind. A combination of large amounts of fine sand and small amounts of clay at the surface means that the soil tends to seal and set hard, which limits infiltration (Prager *et al.*, 2011; Wong *et al.*, 2010). The effect of sodium, specifically the exchangeable sodium percentage (ESP), could also enhance soil erosion. Many studies show that the linear relation between runoff and soil loss becomes exponential at higher levels of ESP (Mamedov *et al.*, 2002). In addition, for the salinized soil, soil organic matter improves soil structure by increasing aggregate stability and decreasing bulk density, which is primarily due to a reduction in ESP (García-Orenes *et al.*, 2005).

Salt interacts with biota (animals and plants), changing the ecological health of land, streams and estuaries. The greatest threat to biodiversity is from the loss of habitat-both on land and in water (Squires, 2009; van Beek *et al.*, 2010). Moreover, soil texture is a very important environmental factor for salinization. The salt content of the soils shows a strong negative correlation with the sand content. However, sandy soils store lower amounts of plant available water and have lower contents of plant nutrients than loamy soils. Generally, the loamy saline soils contain more soil organic matter and more microbial biomass C than the sandy non-saline soils. The negative effects of further increasing salinization lead to a decrease in the soil organic C level in the saline-sodic soils. These interactions between salinization and texture, presumably also lead to a unimodal expression between salinity and microbial biomass. A decrease in salinity and sodicity would improve the accessibility of soil organic matter to the soil microbial community. However, this implies a threat of further reduction in soil organic matter levels if the C input is not improved at the same time (Muhammad *et al.*, 2008).

Besides these links, as the data presented in Figure 9.1 shows, salinity and sodicity frequently overlap each other. Indeed, in many areas in Europe the soil has been characterized not only as saline but also as sodic which means that this soil is no longer suitable for any agricultural use. In this sense, salinization can be viewed as a type of soil contamination. Salinity 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 feedback of organic matter loss, erosion, and desertification.

9.7 Effects of salinization on soil functions

Salinization primarily affects ecological soil functions. Soils in salt-affected landscapes are less fertile and produce less biomass than non-saline soils resulting in less SOC and in turn more erosion, which further accentuates SOC losses due to the dominance of plant inputs in the accumulation of organic matter. Currently, in terms of C accounting, data on how these salt-affected areas relate to C stocks are almost non-existent while data related to C dynamics is contradictory (Wong *et al.*, 2010).

Furthermore, soil biodiversity and microorganism activity declines as EC increases, thus impacting important soil processes such as respiration, residue decomposition, nitrification, and denitrification. As a reciprocal effect of ecological functions, salinization affects a series of environmental interactions leading to reduced water infiltration and retention resulting in increased water runoff and erosion. Regarding non-ecological soil functions, salinization can also lead to damages to water supply infrastructure as well as transport infrastructure from shallow saline groundwater (Montanarella, 2007) thus hindering the functions of soil a physical medium for build development. As a general outcome, land value depreciates, with some studies estimating agricultural land depreciation at 50%, and supply of raw material such as sand, gravel and peat being hindered. Finally, cultural value is also affected thus affecting tourism as well as local people's livelihoods.

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10 DESERTIFICATION

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10.1 Description of desertification

Desertification and Biodiversity are the only two soil threats that have United Nations Conventions. The United Nations Convention to Combat Desertification (UNCCD) was adopted in June 1994 and entered into force on 26 of December 1996. The European Union is one of the signatories to the Convention (UNCCD, 2014a). According to the text of the 1994 convention, desertification means "land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities" (United Nations General Assembly, 1994). These three environments together comprise the drylands, which are susceptible to experiencing full desert conditions (Thomas, 1993). Thomas (1993) noted that hyperarid environments were excluded because "it would be difficult to make them more desert-like than they naturally are."

Although the intuitive view of desertification pictures the advance of sand dunes over previously fertile lands, and the burial and/or abandonment of settlements as the desert margin advances, the UNCCD defined desertification in a less dramatic way, and one that may be important for many parts of southern Europe. Land degradation in dryland areas means the reduction or loss of biological or economic productivity and complexity for rain-fed croplands, irrigated croplands, or range, pasture, forest and woodlands. It is considered to result from land uses that have become unsustainable through the action of one or a combination of processes, many of them arising from or aggravated by human induced activities and habitation patterns, and potentially aggravated by climate change (UNCCD, 2014b).

Although the above standard definitions of desertification are widely applicable, other types of desertification have been identified, and a broader, and perhaps more meaningful definition encompasses "any progressive and unsustainable reduction in the ecosystem services provided by the soil." Note that we do not specify any climatic regions. Clearly, desertification extends beyond the drylands, although most desertification takes place in drylands. Thus, this working definition includes, for example, the wind erosion of fragile volcanic soils in Iceland, as a result of unsustainable grazing pressures that damage the vegetation cover. These processes have much in common with damage linked with overgrazing in the Sahel, despite the contrasting climatic differences. Desertification is thus a very broad term, which has been defined in many different ways (Thomas, 1997, counted more than 100 definitions), and which is expressed though many different degradation processes.

The three most important of these processes for induced desertification are generally considered to be soil erosion, loss of soil fertility and long-term loss of natural or desirable vegetation. Kosmas *et al.* (1999) identified loss of soil by water erosion and the associated loss of soil nutrient status as the dominant desertification process for European Mediterranean environments. In more arid areas, there is greater concern with wind erosion and salinisation problems. Soil erosion by wind mobilises materials from the soil surface, usually re-depositing coarser fractions locally and removing the fine particles of (silt and clay) and organic matter that supports most of the nutrient in the soil and helps to store rainwater. The degraded surface in turn grows only sparse vegetation, further increasing the intensity of wind shear at the surface that eventually leads to the expansion of desertification.

On the other hand, in wet climates, where there is enough rain to generate significant runoff, erosion by water tends to become dominant, washing away fertile topsoil, so that less water is held in the soil to support vegetation growth. At progressively wetter sites, increasingly dense vegetation protects the surface from crusting that seals the surface, and so decreases runoff. The conflict between increasing torrential rainfall and increasing natural vegetation cover leads to a maximum of water erosion in semi-arid climates. However, cultivation generally exposes the soil at the beginning of the rainy season when crops must be planted, and water erosion can then be severe, irrespective of climate.

Long-term loss of natural or desirable vegetation can also degrade the potential land uses of an area, often due to the invasion of rangeland by unpalatable species, in many cases replacing grass with shrubs. Overgrazing exacerbates this process, giving a selective advantage to the unpalatable plants, but climate change and the introduction of non-endemic species may also play a part (Thornes, 1990). Shrub encroachment is commonly associated with desertification (MEA, 2005). Unequal distribution of nutrients, with the shrubs forming fertile patches in a degraded bareland, could lead to desertification (e.g., Kefi *et al.*, 2007). However, Maestre *et al.* (2009) found that shrub encroachment in 13 degraded *Stipa* grassland sites in Spain (260-500 mm annual rain) was linked with greater soil fertility and N mineralization rates, throughout the area. They suggest that the link between shrub encroachment and desertification is not universal. The review of Naito and Cairns (2011) indicates that the expansion of shrubs into rangelands is more common or more researched in the Americas than in the Mediterranean region. In a worldwide analysis of 244 studies from the literature, Eldridge *et al.* (2011) reported that under shrub dominance, soils tended to have (i) lower pH levels, (ii) greater soil C and N pools, and greater potential N mineralization and (iii) higher levels of exchangeable Ca. They found no evidence that encroachment of woody plants/shrubs into semiarid grassland shrubs leads to functionally, structurally or contextually degraded ecosystems. However, in general, ecosystems closer to the more arid end of the climate gradient have a greater likelihood of experiencing increasing ecosystem degradation.

Today, combating desertification is generally one aspect of an integrated development approach that targets sustainable land use through a number of physical measures. Such measures aim to prevent or reduce land degradation, with rehabilitation of partly degraded lands, and perhaps reclamation of more severely desertified areas.

10.2 State of desertification

Globally, 1.5 billion people are said to be directly affected by land degradation. Every year, 12 million hectares of land become unproductive through desertification and drought alone. In the same period, 75 billion tons of soil are lost forever (UNCCD, 2014c). Global land degradation has been mapped by Middleton and Thomas (1997). However, as demonstrated by Vogt *et al.* (2011) and D'Odorico *et al.* (2012), estimates of areas affected by desertification continue to vary widely as a result of different definitions and different methodologies used for estimation. The review of D'Odorico *et al.* (2012) summarized desertification assessment estimates ranging between 10 to 53% of dryland areas.

Although there are no integrated maps for desertification in Europe, sensitivity to desertification has been recently mapped by the DISMED project (Domingues and Fons-Esteve, 2008), based on soil quality, climate and vegetation parameters (Fig 10.1). The map indicates that 8% of the territory in southern, central and eastern Europe shows very high or high sensitivity to desertification, corresponding to about 14 million ha, and more than 40 million ha if moderate sensitivities are included (EEA, 2010).

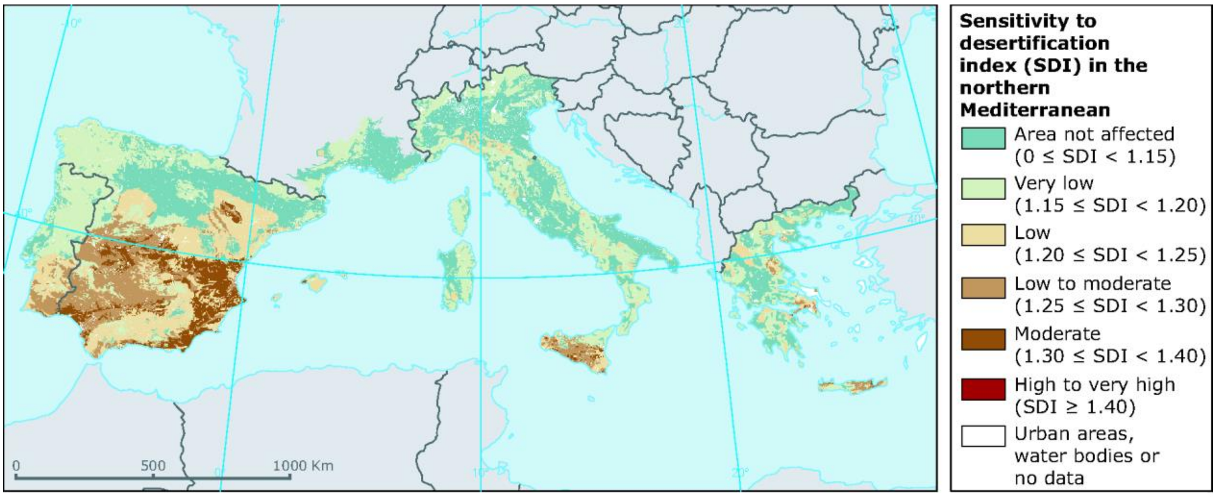

Figure 10.1: Sensitivity to desertification mapped by the DISMED project (Domingues and Fons-Esteve, 2008) (http://www.eea.europa.eu/data-and-maps/figures/sensitivity-to-desertification-in-the-northern-mediterranean).

10.3 Drivers/pressures

Figure 10.2 shows a conceptual framework for desertification, as developed in the FP6 DESIRE project (Hessel *et al.*, 2014). The starting point of the DESIRE conceptual framework is socio-economic and bio-physical drivers which both influence desertification processes. The influence of socio-economic drivers is assumed to be via land use and land management. Some of the drivers might affect both land use and land management, while others might only influence management. Desertification processes in turn affect rural livelihoods, which causes (local) stakeholders to develop or adapt sustainability goals, resulting in the adoption of sustainable land management (SLM) strategies.

Since the concept of desertification was identified for the Sahel in the 1980's, a strong link has always been made between climate and loss of soil quality. Although subsequent work suggested that the regional droughts were a recurrent natural phenomenon. Several climate change modelling studies have been conducted to explain the Sahel droughts. Yoshioka *et al.* (2007) found that changes in sea surface temperatures may explain up to 50% of the observed reduction in Sahel precipitation, while the increase of North African dust can explain up to 30% and vegetation loss in the Sahel region may explain about 10%. It is important to realise that drought is not the same as desertification, and that aridity should not be confused with land degradation (Hessel *et al.* 2014). Aridity and the occurrence of droughts are basic characteristics of dryland climates. However, if dryland climates are changing this can drive soils towards desertification

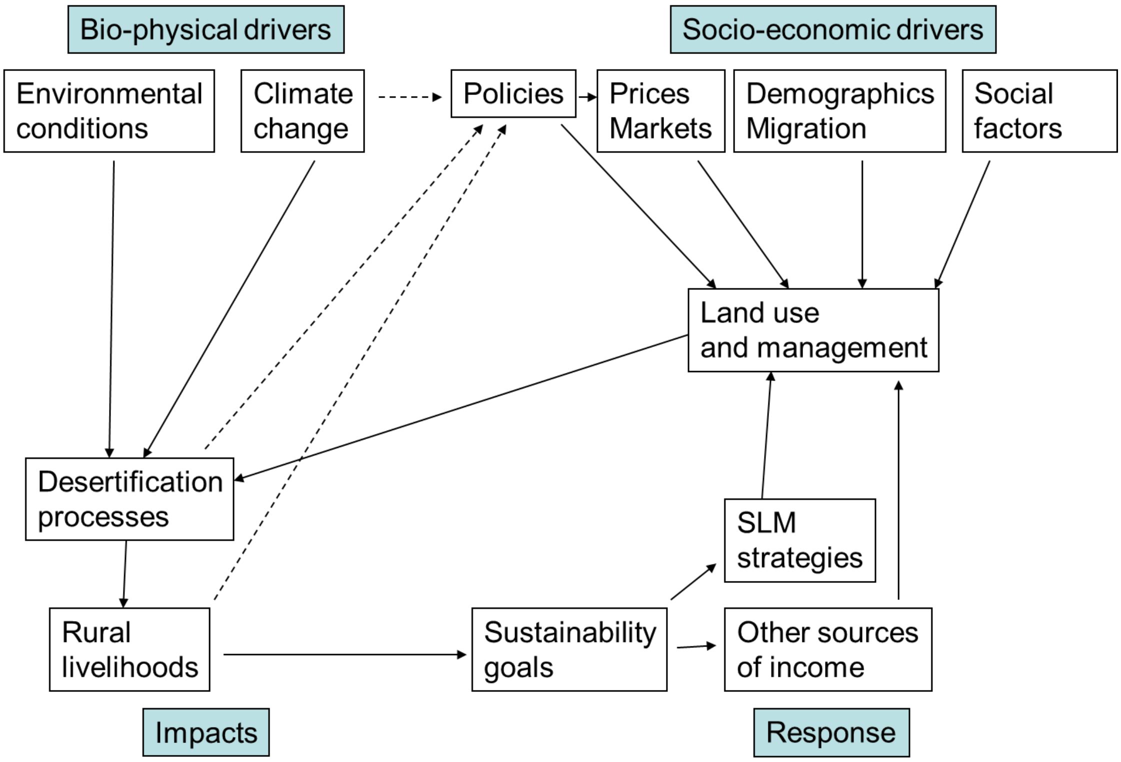

Figure 10.2: Conceptual framework of desertification, as used in DESIRE project (Hessel et al., in press).

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. Projected changes in annual mean surface soil moisture (upper 10 cm) in the Coordinated Modelling Intercomparison Project Phase 5 (CMIP5) ensemble at the end of the 21st century show a consistent drying in the Mediterranean region. This large-scale drying in the Mediterranean appears across generations of projections and climate models and is deemed likely as global temperatures rise (IPCC, 2013). The impact of climate, however, is seen to interact strongly with movements of population. For the Sahel, increases in regional population through improved public health and nutrition, and the imposition of less porous political boundaries hindered migration in response to fluctuating climatic conditions, so that, for example, nomadic herders could not readily move south to wetter areas in times of drought.

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. The area affected by a wildfire increases with wind speeds and with the available dry fuel load near the ground and in the tree canopy. Wild fires in dry land areas reduce soil quality and enhance the threat to desertification.

Widespread population growth has resulted in migration from rural areas to the cities. Although this process has taken many people off the land, net rural populations commonly remain high in areas of subsistence farming and it is only in more developed countries that there has been substantial rural depopulation. In the many mountainous Mediterranean areas, net rural depopulation has led to abandonment of terrace systems that have been laboriously maintained for thousands of years, releasing large volumes of stored sediment (e.g., Arnaez *et al.*, 2011). Population is thus a highly significant driver of desertification, although the impacts vary widely according to local conditions.

Immediate pressures on the soil and vegetation system are posed through inappropriate land use, poor management or overuse of the available resource. One of the most widespread examples of over-exploitation is through increasing grazing density of sheep, goats or cattle in rangeland areas. Marginal areas too arid or too steep for cultivation can be used for stock grazing provided the density is kept low enough to allow recovery of the vegetation cover and from excessive trampling and topsoil compaction, although the soils may benefit from the animal manure if this is not removed for application elsewhere or for fuel, directly or via biogas production. Model results suggest that the optimum animal nutrition occurs when 5-15% of the available biomass is consumed: greater intensities of grazing lead to reduced nutrition and a severe loss of cover. Where unpalatable plants are present or introduced, they may also come to dominate the surviving vegetation, further reducing the value of grazing land. The shift towards less palatable species may also help to drive a conversion of grassland to shrubs, as has been documented in the US south-west, leading to reduced total crown cover, reduced species diversity and significant increases in erosion and desertification.

Van Auken (2000) argues that chronic, high levels of grazing by domestic animals and the concomitant reduction in grassland fires are the main reasons for the increase in density or cover of local shrubs and woody species in the semiarid grasslands of southwestern North America. Eldridge *et al.* (2011) also identified the contribution of higher levels of atmospheric CO_2 and an increase in N deposition as drivers for shrub encroachment. However, these authors noted that the impact of shrub encroachment on ecosystems functioning is mixed. Traits of the encroaching woody plants can influence the overall impact on ecosystem function in soils and especially in community structural attributes, such as biodiversity.

Potential prices for products often influence crop choice more than the concept of sustainable land management. As Tiffen and Mortimore (2002) note, poverty is a more important factor than ignorance in contributing to soil degradation. Although the local population can be well aware of what should be done, they might not always have the means to actually do so. Lack of funds or cash flow barriers can prevent them from buying fertilizers or from applying certain conservation measures. Lack of transport can block their access to regional markets. Therefore, incentives and financial means are needed to allow land users to invest in their land. The existence of an accessible market infrastructure is also crucial, and the undulations of (global) market prices of agricultural commodities also affect land users in desertification prone areas. Land tenure, i.e. land ownership and land use rights also influence the willingness of land users to invest in (sustainable) land management.

Social factors (such as culture, education, religion and ethics) are assumed to influence both land use and management. It goes beyond the scope of this paper to describe these effects in detail. However, it is clear that these issues are at the core of any society, and that they determine to a large extent how populations look upon their surroundings and how they then manage the natural resources (Warren, 2002; Imeson, 2012).

Finally, political context and policies also influence land use and management, with either stimulation or constraint. Degradation is better understood in its political context. Policies influence prices, but also land use and management. For example, the EU Common Agricultural Policy (CAP) that was a.o. meant to provide farmers with a reasonable standard of living, to ensure reasonable prices and to preserve cultural heritage (e.g. de Graaff *et al.*, 2008) also resulted in intensification or abandonment of agriculture (Onate and Peco, 2005; Juntti and Wilson, 2005; de Graaff *et al.*, 2008). National and international policies do directly affect the choices land users make, and in many cases even prescribe certain land management practices.

The importance of these different drivers is different for different locations. In an European context, other drivers might apply than in e.g. an African context. Also, within Europe, there will be variations in the importance of the different drivers. Generally speaking, desertification is caused by an interplay of different causes that operate at different organizational levels and different spatial and temporal scales (Hessel *et al.* 2014). Thus, to understand desertification, and to find ways to combat it, a good understanding of local bio-physical, socio-economic and political conditions is necessary.

10.4 Key indicators of desertification

Through a series of EU and other projects on desertification, a comprehensive list of desertification indicators has been developed. One of the most complete lists is that developed in the EU FP5 DesertLinks project (DIS4ME, 2004). The 148 candidate indicators have been categorised according to their relevance to ecological, economic, social or institutional aspects of desertification (Table 10.1).

Table 10.1: Desertification indicators from DIS4ME.

Physical indicators	
Climate	Air temperature, Aridity index, Climate quality index, Drought, Drought index, Effective precipitation, Potential evapotranspiration, Rainfall, Rainfall erosivity, Rainfall seasonality, Wind speed
Water	Groundwater depth (change in), Water quality
Runoff	Dam sedimentation, Drainage density, Erosivity (RDI), Flooding frequency, Floodplain and channel morphology, Impervious surface area, Rainfall-runoff relationship, Runoff threshold (RDI), Soil permeability
Soils	Acidified area, Drainage, Erosion risk (RDI), Infiltration capacity, Organic matter in surface soil rs, Organic matter in surface soil, Organic matter mixing with depth, Parent material, Rock fragments, Salinization potential, Slope aspect, Slope gradient, Soil crusting, Soil depth, Soil erosion (USLE), Soil erosion (measured), Soil loss index, Soil quality index, Soil stability index, Soil structure, Soil surface stability, Soil texture, Soil type, Water storage capacity
Vegetation	Area of matorral, Biodiversity conservation, Deforested area, Drought resistance, Ecosystem resilience, Erosion protection, Forest fragmentation, Vegetation cover,Vegetation cover type, Vegetation quality index
Fire	Burned Area, Fire Frequency, Fire Risk, Forest and wild fires, Fuel models, Wild fire incidence
Economic indicators	
Agriculture	Expenditure on water, Family size, Farmer's age, Farm ownership, Farm size, Forest productivity, Fragmentation of land parcels, Gross margin index, Traditional agricultural products, Net farm income, Parallel employment
Land management	Agri-enviromental management,Fire Protection, Forest management quality, Management quality index, Organic farming, Reclamation of affected soils, Reclamation of mining areas, Soil erosion control measures, Soil water conservation measures, Sustainable farming, Terraces (presence of)
Land use	** Area of cultivated & semi-natural vegetation (rs), Area of marginal soil used, Land abandoned from agriculture, Land use evolution, Land use intensity, Land use type, Natural vegetation, Period of existing land use type, Shannon's diversity index, Urban sprawl
Cultivation	Area of hillslope cultivated, Fertilizer application, Mechanisation index, Tillage direction, Tillage depth, Tillage operations
Husbandry	Grazing, Grazing control,Grazing impact, Grazing intensity, Husbandry intensity
Water use	Aquifer over exploitation, External water resources, Groundwater exploitation, Hydrological regulation (artificial), Irrigated area, Irrigation intensity and seawater intrusion, Irrigation percentage of arable land, Irrigation potential realised Runoff water storage, Water consumption by sector, Water leakage, Wastewater recycling, Water scarcity, Water availability
Tourism	Penetration of tourist eco-labels, Tourism contribution to local GDP, Tourism change, Tourism intensity
Macro economics	Employment index, GDP per capita, Accessibility, Unemployment rate, Value added by sector

The ENVASSO Project (Kibblewhite *et al.*, 2008) proposed three indicators for monitoring of desertification at EU level: (i) land area at risk of desertification, (ii) land area burnt by forest fires, (iii) soil organic carbon content in desertified areas. Only the first indicator was tested, using the MEDALUS model at different scales in tree pilot areas. However, this indicator is again an amalgamation of a number of indicators. The authors concluded that input data harmonization and development of standard procedures for data integration into GIS is needed.

As part of the FP6 DESIRE project (Kosmas *et al.*, 2014; Kairis *et al.*, 2014) a list of candidate desertification indicators was developed, based on the DIS4ME indicators, literature and input from stakeholders in 16 DESIRE study sites. They investigated which of 70 candidate indicators, related to biophysical environment, socio-economic conditions, and land management, were most effective in assessing the level of desertification risk for 6 different desertification processes, namely water erosion, tillage erosion, salinization, water stress, overgrazing and forest fires. The analysis indicated that 8-17 indicators were needed to determine desertification risk for these processes. The most important indicators were found to be: (i) rain seasonality affecting water erosion, water stress, and forest fires, (ii) slope gradient affecting water erosion, tillage erosion and water stress, and (iii) water scarcity affecting soil salinization, water stress, forest fires, and water scarcity. Implementation of existing regulations or policies on resources development and environmental sustainability was identified as the most important effective indicator on land protection due to various processes or causes of land degradation identified in the study field sites.

Sommer *et al.* (2011) and Cowie *et al.* (2011) reviewed the nine indicators of the UNCCD framework. Cowie *et al.* (2011) summarized these as indicators for assessing: (i) trends in condition of physical resources (including carbon stocks); (ii) the condition of affected populations; and (iii) responses (sustainable land management, policies). They stressed the importance of recognizing the dynamic nature of dryland systems, with cycles of change strongly affected by rainfall. Monitoring strategies should also consider the non-linearity of the system and thresholds for system collapse. Sommer *et al.* (2011) also noted that the causes and consequences of desertification vary within space and scale, thus, requiring different sets of indicators. The change in dominant processes as a result of up-scaling was also recognized by Kirkby *et al.* (1998) in the development of two soil erosion models for the MEDALUS project.

Soil organic carbon has been recognized as a key indicator of soil health (e.g., Lal, 2003; Cardoso *et al.*, 2013). Cardoso *et al.* (2013) also emphasize the critical role of microbial biomass and diversity. These authors proposed the use of an integrative approach for maintaining soil health and productivity, based on physical, chemical and biological indicators.

It is recommended that the RECARE project focus on physical state indicators that are directly related to the degree of desertification and its effect on soil functions. Of these perhaps the most central, and which appear in all proposed lists of indicators are organic matter content, soil depth and electrical conductivity. Although these vary consistently with climatic regime and between undisturbed and arable land, differences within an area provide an excellent first indication of the state of soil health and productivity, and how it may be changing over time.

10.5 Methods to assess the status of desertification

Vogt *et al.* (2011) reviewed desertification assessment methods that have been used since the United Nations Conference on Desertification (UNCOD) in 1977, such as GLASOD (Global Assessment of Human-Induced Soil Degradation), LADA (Land Degradation Assessment in Drylands), and ROSELT/OSS (Long-Term Ecological Surveillance Observatories Network of the Sahara and Sahel Observatory). Both GLASOD and LADA used expert opinion, while in ROSELT monitoring of bio-physical parameters is combined with surveys for socio-economic data (Vogt *et al.*, 2011)

Another widely applied method for assessing the detailed state of desertification within an area is the Environmentally Sensitive Area (ESA) method developed by Kosmas *et al.* (1999). For a large region, a nested approach can be adopted, in which a range of indicators is used to identify areas of 500-5,000 km^2 at greatest risk. Within these areas, the ESA methodology can be applied, using a more restricted range of 15 indicators that were considered most relevant within the individual area.

The ESA method was applied within the MEDALUS projects to the island of Lesbos and to the Agri Basin of southern Italy (Kosmas *et al.*, 1999; 2000). For these areas, the soil indicators chosen were *rock type, soil stoniness, soil depth, slope gradient* and *decline of organic matter with depth*. In addition climatic indicators were also used, *precipitation, aridity* and *aspect*; vegetation characteristics reflecting, *cover, fire resilience, drought resistance* and *crown cover*. Land use factors including *fire frequency, grazing intensity* and *terracing*. These indicators, obtained from GIS, maps and field survey, were mapped across the study area. Various methods have then been applied to combine then, for example using principal component analysis to identify the key independent components of the desertification for the study area, which could then be mapped at scales of 1:25,000 -1:50,000.

In the DESIRE project, Kostas *et al.* (2014) and Kairis *et al.* (2014) developed multiple linear regression models for 6 desertification processes, based on indicators. Results from 16 DESIRE study sites, located around the world in semi-arid climates, were used to develop these equations. AUA (2011) implemented these equations in a decision support tool that allows users to calculate desertification risk based on indicators.

Methods based on satellite images are also being used to assess desertification. The most commonly used method is the Normalized Difference Vegetation Index (NDVI), which assesses greenness and is thus a measure of Net Primary Productivity (NPP) (Higginbottom and Symeonakis, 2014). As mentioned by Higginbottom and Symeonakis (2014), two main assumptions are made, namely 1) that degradation results in a decrease of NPP, and 2) that NDVI variation is able to capture the change in NPP. These assumptions might not always be met, for example because degradation does not necessarily need to result in a decrease of NPP (Vogt *et al.*, 2011). For example, bush encroachment might increase greenness rather than decrease it, and processes like soil erosion, salinization and nutrient depletion might not cause changes in NPP (Vogt *et al.* 2011). Bai (2011) performed NDVI trend analyses using GLADA methodology for 14 DESIRE sites that are known to have degradation problems, but found greening in 8 out of these. Interestingly, Ivits *et al.* (2014), who analysed an NDVI time-series found that Northern and Mediterranean ecosystems were more resilient to droughts in terms of vegetation phenology and productivity than Eastern Europe and Western Atlantic regions. There are also various other technical pitfalls and problems with the use of NDVI in dryland areas (Eisfelder et al, 2011) and to assess desertification (Higginbottom and Symeonakis, 2014), but nevertheless, satellite based methods have the potential to provide data about large areas, and for various moments in time. Hence, they can provide useful information if used in combination with other methods.

Alternative methods typically map individual components of desertification, focusing primarily on the dominant processes (for Europe) of water erosion, loss of fertility and loss of desirable vegetation types, using, for soil erosion, either physically based modelling (eg PESERA or RUSLE) factor-based assessments (e.g., CORINE or MESALES) or questionnaire-based approaches (e.g., GLASOD, LADA or WOCAT).

Vogt *et al.* (2011) noted that for desertification assessment there is a lack of standardized procedures, and that an integrated framework is needed to enable meaningful, repeatable and comparable assessment of desertification. Such a framework should address both bio-physical and socio-economic dimensions to identify which are the key variables that should be monitored. The Dryland Development Paradigm (Reynolds et al, 2007, 2011) is a step in that direction.

Given the focus of RECARE on soil properties, and how these can be used to assess degradation as well as ecosystem services, an indicator based approach is proposed for RECARE, at least for work at case study level.

10.6 Effects of desertification on other soil threats

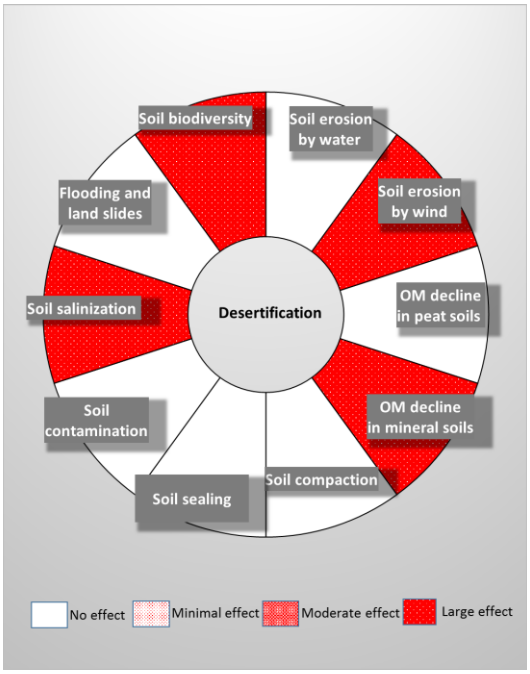

Figure 10.3: Effects of desertification on other soil threats. Red is negative effect.

All soil threats could contribute to an increase in desertification. For example, Belnap (1995) found that soil compaction and disruption of cryptobiotic soil surfaces (cyanobacteria, lichens, and mosses) caused by livestock, people, and off-road vehicles resulted in increased vulnerability to desertification at five study sites in Utah. The other way around, desertification also affects other soil threats (Figure 10.3). A reduction in desertification will improve biomass production and thereby soil organic matter and nutrient cycles. The increase in vegetative cover and plant roots will also reduce the risk of wind and water erosion.

10.7 Effects of desertification on soil functions

When soils are degraded, they lose their capacity to capture and store water, nutrients and carbon and to support microbiological processes. Considering the slow natural formation of soils, the loss of soil functions due to desertification is often irreversible (Van-Camp *et al.*, 2004). Table 10.3 summarizes the effect of a decline in desertification on soil functions. Clearly, desertification will affect all soil functions. However, the impact on biomass and food production, biological habitat, and environmental services will be the greatest.

Table 10.3: The effect of a decline in desertification on soil functions (+ slight, ++ moderate, +++ strong positive effect, i.e. the soil function is improved).

Soil function	Desertification
Food and other biomass production	+++
Environmental interaction: storage, filtering, buffering and transformation (including carbon pool)	+++
Biological habitat and gene pool	+++
Physical and cultural heritage	+
Platform for man-made structures: buildings, highways	+
Source of raw materials	+

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11 FLOODING & LANDSLIDES

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11.1 Description of flooding and landslides

Flooding can be defined as the overflowing by water of the normal confines of a watercourse or water body and/or the accumulation of drainage water over areas which are not normally submerged (WMO, 2012). In addition to inundating and degrading soils, the flooding of soils and the subsoil itself can represent the source areas of floods which can have impact on areas located downstream (Jackson *et al.*, 2008). A landslide is defined as the movement of a mass of rock, debris, artificial fill or earth down a slope, under the force of gravity, causing a deterioratilunaon or loss of one or more soil functions (Huber *et al.*, 2008). Landslides are usually classified on the basis of their type of movement (fall, topple, slide, lateral spread, and flow) and the type of material involved like rock or fine/coarse soil (Varnes, 1978; Cruden & Varnes, 1996; Hungr *et al.*, 2001). In some specific cases of land degradation, despite of the different mechanism of flow of both agents (water and soil), a gradual transition from landslides to floods and vice versa can also be observed in areas with high soil erosion and local flooding potential (muddy floods) (Stankoviansky *et al.*, 2010). This chapter will focus on shallow landslides, usually characterized by a sliding or flowing type of movement and involving soil, not rocks.

The intensity of flooding can be divided into a number of fast and slow flow generating, controlling, and concentrating processes. As well as infiltration and saturation excess overland flow, the subsurface storm flow is nowadays generally recognized as another dominant factor in flood/landslide generation (e.g., Bachmair & Weiler, 2014). Preferential flow in the soil/subsoil/hill slope systems through subsurface networks also contributes to the transport of fine particles, water and solutes and finally to soil degradation (Band *et al.*, 2014). Overland and subsurface storm flow on hill slopes lead to runoff into spatial flooding and floods in a stream network during particular events. Small and large scale stagnant, flooded soils can suffer significant soil deterioration. Slow flow-controlled processes such as the gradual thinning of the soil by erosion, increases the portion of overland flow. Amplification of subsurface storm flow by subsoil compaction contributes to slow changes in runoff/landslide generation regimes over several decades. Flooding can be accompanied by landslides with mass movements, which are sources of coarse and fine sediments in river networks (Butzen *et al.*, 2014).

Landslides are dominantly considered as a local soil threat in mountainous regions and on slopes. Their major driving force is gravity, but local management and controls can be responsible for triggering/preventing them. Among the most common local factors interacting with landslides are topography and the related relief characteristics; soil and bedrock and their specific mechanical and hydrogeological properties; soil depth; hydrological and hydrogeological conditions; vegetation; and anthropogenic activities. However, the most important triggering factor for landslides remains climate and, in particular, precipitation.

Flooding and landslides represent significant threats to man's activities, property and infrastructure. The rich spatial and temporal heterogeneity of control conditions (e.g., the state of the soil and storage controls), and the variety of active flow processes during particular events makes it difficult to arrive at generalized descriptions of the genesis of particular types of events, to define specific local risk factors, and to design generally applicable mitigation schemes (e.g., Bachmair & Weiler, 2014; Fiener *et al.*, 2011). This, together with the potential changes in climate and precipitation regimes (Petrow *et al.*, 2009; Lehtonen, *et al.*, 2014), represents a particular challenge.

11.2 State of flooding and landslides

Floods and landslides are major natural hazards, costing millions of Euros in property damage and claiming many lives each year in almost all areas of Europe (EEA, 2010; EM-DAT, 2003). However, despite of the fact that both have clear impacts on the soils on which they occur, and it is therefore justified to recognise them as soil threats, floods and landslides were generally not considered as threats to soil at sites in the recent past, but more as triggers of threats to societal activities in susceptible areas and natural disasters Numerous studies have been undertaken at the European and regional levels in this respect for past, present and future climates (e.g., Lehner *et al.*, 2006; Lugeri *et al.*, 2010; Kundzewicz, 2012; Hall *et al.*, 2014). A number of Europe-wide, interregional and national flood research programs have contributed to a deeper and valuable understanding of the problems of flooding and landslides (see http://www.crue-eranet.net/ for an overview and the database). The societal perception of the state of floods in Europe can be illustrated by the combination of the potential damage and the risk of flooding, as shown in Figure 11.1.

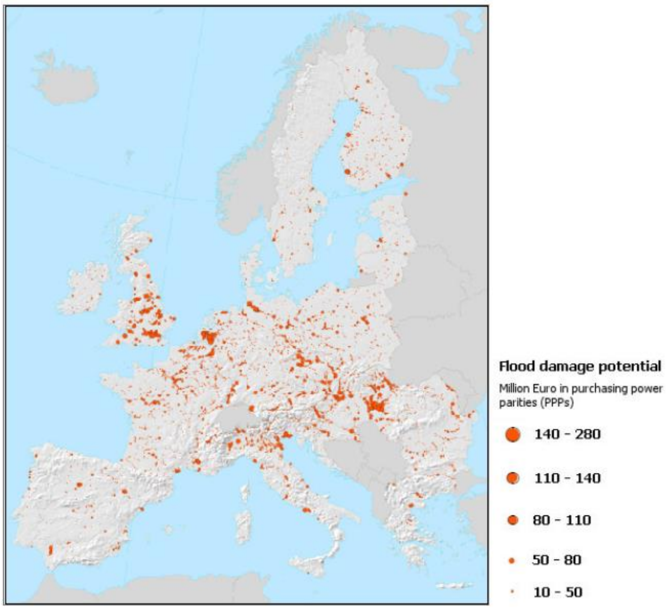

Figure 11.1: Flood damage potential in the European Union. (Source: http://floods.jrc.ec.europa.eu).

Landslides are dominantly considered as a local soil threat in mountainous regions and on slopes. Landslides account for a high number of dedicated EU FP, interregional and national projects (e.g., LAMPRE, SafeLand, DORIS, ENSURE, CHANGES, and KULTURisk). These projects range from advances in mapping, monitoring and forecasting on the basis of developing earth observation techniques, to the development of new strategies and policies for landslide risk management. Specific studies on hazards, vulnerability and risk modelling have been included too, as well as analyses of expected changes in the climate, land use and population.

The assessment of the relative importance of climate and land use changes on flooding across Europe varies from study to study. One has also to take into consideration that messages have been prevailingly derived from data and modelling hotspots and applied to flooding at sections and along river reaches and not necessarily region-wide with respect to soil deterioration. Hazards posed by flooding and landslides have generally been considered to be rising in the last century in Europe with regionally different intensities (e.g., Van Beek & Van Asch, 2004; Kundzewicz, 2012). Kundzewicz *et al.* 2013b) showed an increasing trend during a 25-year period in the number of reported floods exceeding the severity and magnitude thresholds. On the other hand, Mudelsee *et al.* (2003) analysed long-term records from the two main European rivers, (Elbe and Oder) and did not find any increase in flood occurrence rates in recent decades. According to EEA more than 325 major river floods have been reported for Europe since 1980, of which more than 200 have occurred only during the last 15 years (EEA, 2012a). The rise in the number of floods was attributed to better reporting, as well as land-use changes. On the contrary, Kundzewicz *et al.* (2013a) has noted that it was not possible to attribute rain-generated peak stream flow trends to anthropogenic climate change over the past several decades despite the fact that economic losses from floods have greatly increased. The latest IPCC assessment (IPCC, 2014) attributed only a few changes in flood trends as a result of climate change, partly due to lack of sufficiently long term records. According to IPCC (2014), increased peak flows over the past 30-50 years have been observed in parts of Germany, the Meuse river basin, parts of Central Europe, Russia, and north-eastern France. No changes were observed in Switzerland, Germany, and the Nordic countries. River regulation possibly masked increasing peak flows in the Rhine River. A general schematic summary of the flood changes observed in Europe, was reported by Hall *et al.* (2014) and is shown in Figure 11.2.

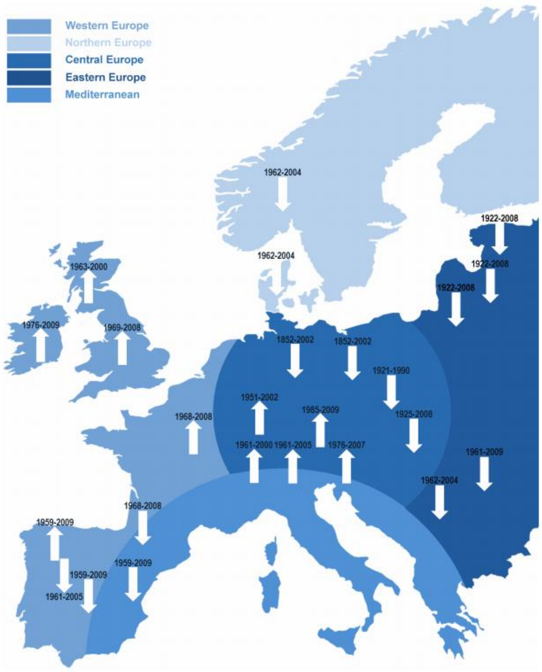

Figure 11.2: Arrows in the schematic indicate the majority of flooding trends, including regions with weak and/or mixed change patterns. Areas with no/inconclusive studies due to insufficient data (e.g., Italy) and inconclusive change signals (e.g., Sweden) are not shown – Hall et al. (2014).

Petrow *et al.* (2009) established a link between trends in circulation patterns and trends in the flood magnitudes for 122 meso-scale catchments in Germany for the period 1951–2002, and showed that significantly increasing frequency and persistence of flood-prone circulation patterns intensified flood hazards during the winter season throughout Germany. Moreover, a trend towards a reduced diversity of circulation patterns was found, causing fewer patterns with longer persistence to dominate the weather over Europe.

With respect to climate change, an EEA report (EEA, 2012b) concluded that global warming is projected to intensify the hydrological cycle and increase the occurrence and frequency of flood events in large parts of Europe. Flash floods and pluvial floods, which are triggered by local intense precipitation events, are likely to become more frequent throughout Europe. In regions with a projected reduction of snow accumulation during the winter, the risk of early spring flooding could decrease. However, quantitative projections of changes in flood frequency and magnitude remain highly uncertain.

Hazards posed by landslides are accidental and dynamic. Landslides are increasingly recognised as a severe problem, as evidenced by the numerous studies that try to assess the most susceptible areas all over Europe (Van Den Eeckhaut *et al.*, 2012; Günther *et al.*, 2013). Figure 11.3 shows landslide susceptibility map of Europe (Panagos *et al.*, 2012; Günther *et al.*, 2013).

Due to expected global warming and the related increase in extreme precipitation (IPCC, 2014), on a theoretical basis landslide activity is expected to increase as well (Crozier, 2010). However, a realistic projection of how landslide activity can evolve should include the role of human activity and a geomorphologically evolving background as well (Crozier, 2010).

The regime of precipitation events and weather patterns may be changing (e.g., IPCC, 2014; EEA, 2012a), which might be attributed to changes in climate. Will this be a decisive factor in the increase in flood and landslide risks on slopes and to what extent? How can these soil threats be reduced by managing and improving the water regulation function of the soil and landscapes? Such questions have so far not been answered in a spatially coherent way all over Europe, especially in relation to other soil threats and respecting diverse hydroclimatic and physiographic regions, agro-ecological systems, land management practices and socio-economic situations.

11.3 Drivers and pressures

The driving forces/pressures for flooding and landslides are of natural, social, economic, and ecological origins. They interact in complex ways; therefore, the analysis of their impacts requires respecting synergies (Crozier, 2010).

(i) Climate drivers

Climate and climate change control precipitation and snowmelt (frequency, intensity and magnitude, seasonality, cyclonality and the respective changes), which manifest their impacts both locally and regionally. Both of these factors are the most important external drivers for landslides and flooding (e.g., Iverson, 2000; Crosta & Frattini, 2003).

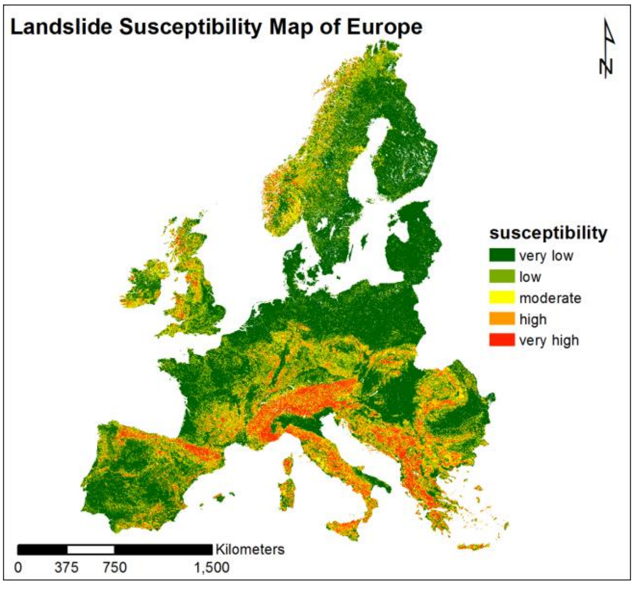

Figure 11.3: Landslide susceptibility map of Europe (Panagos et al., 2012; Günther et al., 2013).

The spatio-temporal variability of rainfall, especially on fine temporal and spatial scales, can significantly affect flooding and trigger landslides, and can lead to great variability in responses and uncertainty in their prediction (Paschalis *et al.*, 2014). A study covering the whole of Europe was conducted by Van den Besselaar *et al.* (2013) who analysed the trends in rainfall extremes between 1950 to 2010 for short duration (1 day) and for long duration (5 days) events in different seasons in northern and southern Europe regions. The result showed that the frequency of extreme events is increasing in all regions for all the seasons and for both durations considered. The northern part of Europe is generally more affected than Southern Europe as the winter months are showing the highest rate of change in the frequency of rainfall events, indicating an increase of flood and landslide risks. Nied *et* *al.* (2013) studied the impact of soil moisture as a driver for flooding in the Elbe River Basin. They concluded that 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 cases 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).

(ii) Human induced land use changes
One of the main socio-economic drivers for flooding and landslides are changes in land use. Slow changes in land-use/management due to economic reasons and change of political priorities (e.g. changes in agricultural policies (Baartman *et al.*, 2012) complement alterations of vegetation due to feedbacks to climate and ecological changes (Brown *et al.*, 2005). Among these are factors such as the abandonment of land (Lozano-Parra *et al.*, 2014), changes in forest cover (Robinson *et al.*, 2003), protection of ecosystems and natural water retention measures (Salazar *et al.*, 2012; Burek et.al., 2012). Specific and sudden local factors such as forest fires (DeBano, 2000; Granged *et al.*, 2011), ecological and wind-related calamities (Novák & Šurda, 2010), and sealing due to urban development (Ranzi *et al.*, 2002), which are, to a large extent, unpredictable, should also be considered. Changes in agricultural practices have received particular attention because of their role in flood formation. Brath *et al.* (2006) concluded that the flood flow regime in Italy was remarkably sensitive to the land use changes in the catchment, which, amongst others, consisted of a reduction in meadows and pastures. On the other hand, Marshall *et al.* (2014) found that runoff volumes varied greatly according to land use cover in experimental plots in the UK, with improved grazed land having significantly higher runoff volumes than ungrazed and tree-planted plots. Increased sub-surface drainage of both agricultural fields and forested areas can also increase flood risks (Wheater & Evans, 2009). Influence of various forest types/managements on associated flood generation mechanisms have also been explored by experiments and modelling and extensively reviewed in the literature. The experiments were on a hillslope scale with deforestation and afforestation, as well as on established forest stands covering various physiographic contitions (Andréassian, 2004; Cosandey *et al.*, 2005; Kostka & Holko, 2006; Wahren *et al.*, 2009; Butzen *et al.*, 2014). Afforestation and reforestation in general reduced floodings due to increased evaporation (Wheater & Evans 2009; Černohous *et al.*, 2014), but contradictory results were reported as well.

Increases in vegetation/forest cover reduces landslide activity and soil loss (García-Ruiz & Lana-Renault, 2011), and improves the mechanical characteristics of the soil because of root-cohesion (Bathurst *et al.*, 2010). The abandonment of the lands (Nadal-Romero *et al.*, 2012) in the terraced slopes in the Mediterranean environment of southern Europe has led to an increase in shallow landslide activity. Often terraces are retained by dry-stone walls that, if not well maintained, can lose their drainage function and develop saturated horizons at their back slope that can result in their collapse and the triggering of superficial landslides (Camera *et al.*, 2014). Although the impact of land use changes on flood and landslide responses has been investigated by several authors, this topic is still significantly under-researched and unclear (e.g., Deasy *et al.*, 2014).

(iii) Policy drivers
European and national policies targeting flooding/landslides provide a broad interlinked framework for mainstreaming flooding and landslide risk management mainly through agriculture, water and climate change mitigation policies. At EU level a comprehensive set of policies addressing such risks exists, which are implemented into national policies/legal frameworks. The most relevant in the EU are the EU Water Framework Directive (WFD), the EU Floods Directive (FD), the EU Common Agricultural Policy (CAP) and Structural and Cohesion Funds. The WFD recommends to take climate change into account in River Basin Management Plans and the FD requires flood risk management plans and flood risk assessments to be undertaken. While the CAP does not directly address flood and landslide risks, its recent reforms present mainstreaming opportunities through cross-compliance regulations that require on-farm measures (e.g. small retention ponds, shelter belts which can reduce runoff and changes in tillage practices to maintain soil moisture). The Agri-Environment Program plans to compensate farmers for implementing on-farm water-retention and other ecological investments with indirect impacts on flooding and landslides. The European Commission also stresses the need of mainstreaming climate change mitigation into flood/landslide risk policy. A Blueprint to Safeguard Europe's Water Resources stressed the importance of natural water retention measures and planned policy integration tools for the 2014–2020 Multiannual Financial Framework (MFF) that could greatly enhance the take-up of green infrastructure. However, lacking a European Soil Framework Directive, soil conservation policies have been in the past, and still usually are, developed on a national or

local basis. For this reason, assessments of their effects on flooding and landslides are difficult to be found in literature.

11.4 Key indicators on flooding/ landslides

Given the large amount of diversity in cause and effect relations, indicators (derived by office, field work and modelling) (Smith & Redding, 2012) for flooding/floods and landslides need to reflect the following:

- runoff and soil mass movements generation and environmental factors controlling their dynamics;
- the sensitivity of flooding, flood runoff and soil mass movements to disturbances;
- the varying biophysical and socio-economic factors for signalling changes (FAO, 2003);
- policy relevance, analytical soundness and measurability (OECD, 2004); and
- reliable data from harmonised soil monitoring networks (Huber *et al.*, 2008).

The critical and threshold values of these are by nature local, as they are dependent on the hydrology of the plot/hillslope/river basin in question.

11.4.1 Flooding

Plot scale: Climatic indicators comprise critical rainfall and snowmelt intensities for flooding, which may include the influence of vegetation and slopes with respect to soil moisture conditions. Soil quality indicators may include typical physical, chemical, and biological indices such as soil depth (topsoil depth), soil bulk density, soil and subsoil permeability, available water holding capacity and saturated hydraulic conductivity, soil compaction, porosity, soil texture and structure, parent material, organic matter content, soil moisture etc. A list of potential indicators for flooding and floods is given in Table 11.1. Less obvious runoff indicators can complement these such as water repellency and hydrophobicity, surface roughness, soil crusting, and macropore structures.

Hill slope scale: On the hill slope scale, indices for morphology, runoff generation mechanisms, and land use have to be added. Standard indicators would include regionally valid frequencies of precipitation intensities in relation to scales (e.g., runoff concentration time), slope, slope exposure (aspect) and gradients, topographic convergence/divergence, organic soil depth, mineral soil depth, infiltration capacity, surface runoff coefficients, runoff pathways, groundwater depths, vegetation and land use patterns. Less used, but recently intensively studied, are the distribution of macro pores and preferential flow paths, bedrock permeability and the connectivity of runoff generation areas.

Catchment scale: Typically, morphometric parameters (such as stream density, catchment shape, size and distribution of wetlands and sealed surfaces), anthropogenic disturbances (cross-drain frequency or density of roads, density of stream crossings, drainage systems), and runoff concentration indices are used to express the characteristics of the floods formation. Recent research suggests including indices for the distribution/dynamics of runoff-producing zones, digital terrain indices, residence times, the proportion of old and new water, and wetness-related vegetation indices.

For river floods, typical flood indicators include water level thresholds for water gauges. Furthermore, flood frequency and duration, the extent of land that is inundated at any given water level, economic losses or vulnerability to floods are also often used as indicators.

11.4.2 Landslides

The main set of indicators listed in the ENVASSO project (Huber *et al.*, 2008) for landslides are:

- occurrence of landslide activity;
- volume or mass of displaced material;
- landslide hazard assessment

The indicators for landslides can also be divided into two main categories: indicators of susceptibility/hazard and indicators of activity. When talking of susceptibility, the scale of reference is regional, while when analysing activity, the scale is local. The susceptibility of landslides in a certain area depends on the condition of the materials; topographical setting; structural setting; and land use (Guzzetti *et al.*, 2005; Fressard *et al.*, 2014). Lithology, soil texture and soil structure can be considered as good proxy indicators for porosity, hydraulic conductivity, and mechanical resistance, as they have a direct influence on landslides. Soil depth, can also be useful indicator for the available volume to store water. In terms of topographic characteristics, a commonly used indicator is the slope angle, as it directly influences the component of the forces acting on the soil mass. The proximity to faults (Guzzetti *et al.*, 2005) is the main structural indicator as it can be the cause of diffuse weakness and earthquake activity in seismic areas.

Table 11.1: Summary of the main indicators for flooding and floods.

Indicators	Rationale	Methods	Outputs (expected)
Seasonality, magnitude and frequency of precipitation, rainfall intensity	Flooding and flood generation potential of soils, hill slopes and catchments	Statistical analysis of precipitation measurements	Water inputs which affect the potential for flooding and flood runoff
Standard soil quality/property indicators	Flooding and flood generation potential of soils and hill slopes	Standard laboratory and field methods	Indication of occurrence of infiltration-excess overland flow, water ponding on the soil surface, depth of the percolation, the potential for saturation of overland flow, lateral flow in the soil, groundwater recharge
Special soil quality/property indicators	Hill slope and catchment-scale spatial differentiation	Satellite and airborne remote sensing, dye tracers, TDR, rain simulation	Spatial distribution of areas with high flooding/flow generation/erosion potential
Spatial flow generation and flooding	Hill slope and catchment-scale spatial integration of factors/processes	Rainfall-runoff modelling, GIS terrain and spatial analysis	Spatial distribution of areas with high flooding/flow generation/erosion potential
Anthropogenic disturbances to flow paths	Potential for changes to flooding and flood regimes	Office and field surveys	Potential for amplification of natural flooding/flood hazards
Catchment flood regime descriptors	Climatic and environmental controls of runoff dynamics	Hydrological statistics, field mapping, GIS analysis	Understanding of seasonality, frequency and magnitude of flooding/floods
Water level thresholds exceeded	Provides easy understanding to stakeholders	Well-known hydrological methods	Flooding threshold identification
Extent of inundated area	Combined with land use of inundated area, this will give info on potential soil degradation	Flood zone mapping	Potential area of soil degradation due to flooding
Flood frequency	Quantitative estimate of natural hazards	Statistical analyses	Potential soil degradation due to floods/flooding
Flood duration	As above	Flood duration analyses	Potential soil degradation due to floods/flooding
Soil anaerobic conditions	Plant productivity, organic matter and nutrient dynamics.	Duration of flooding	Direct measure of soil degradation
Loss of crops due to inundation of fields	Economic losses may be easy to monitor	Questionnaires, surveys	Indirect measure of soil degradation and changes in ecosystem services
Loss of crops due to siltation of fields	Economic losses may be easy to monitor	Questionnaires, surveys	Indirect measure of soil degradation and changes in ecosystem services

Table 11.2: Main indicators of landslide susceptibility and activity.

Indicators	**Rationale**	**Methods**	**Outcomes/outputs (expected)**
Displacement of mass	Moving mass	Field measures, remote sensing	Direct measure of landslide activity
Elevation	Influence on erosion	Topographical surveys, remote sensing	Indirect measure of soil depth and soil cover/vegetation
Groundwater depth	Water availability/Reduction of effective strength	Field measures	Indirect measure of the influence of available water
Infiltration capacity	Water availability/Reduction of effective strength	Field measures	Indirect measure of the influence of available water
Land abandoned from agriculture	Modification of water redistribution processes	Field mapping, remote sensing	Indirect measure of land degradation
Landforms	Moving mass	Field mapping, remote sensing	Indirect measure of landslide activity
Land use evolution	Water redistribution/Soil properties	Field mapping, remote sensing	Landslide susceptibility
Land use type	Water redistribution/Soil properties	Field mapping, remote sensing	Landslide susceptibility
Parent material	Influence on soil mechanical and hydrogeological properties	Field mapping	Indirect measure of soil resistance
Presence of terraces	Topographical disturbance	Field mapping, remote sensing	Indirect measure of modified hydrological conditions
Potential evapotranspiration	Water availability	Field measures (direct or indirect)	Indirect measure of the influence of available water
Proximity to faults	Area weakness	Field mapping, remote sensing	Indirect measure of bedrock weakness
Rainfall	Water availability/reduction of effective strength	Intensity-duration thresholds, monitoring	Indirect measure of the influence of available water/Landslide threshold identification
Rainfall-runoff relationship	Water availability	Field measures (indirect)	Indirect measure of the influence of available water
Slope aspect	Influence on soil moisture	Derived from DEM	Indirect measure of the influence of available water
Slope gradient	Influence on acting forces	Derived from DEM	Indirect measure of acting forces
Soil depth	Water storage capacity	Field mapping/indirect methods (geophysics)	Indirect measure of the influence of available water
Soil structure	Influence on soil mechanical and hydrogeological properties	Field mapping, laboratory analyses	Indirect measure of soil resistance
Soil texture			Indirect measure of soil resistance
Vegetation cover	Reinforcement of soil through root system	Field mapping, remote sensing	Indirect measure of soil resistance
Water storage capacity	Potential water availability/reduction of effective strength	Field and laboratory measures (indirect)	Indirect measure of the influence of available water

11.5 Methods to assess risk of flooding/landslides

The main tools for analysing flooding and landslides are monitoring, experimental research and modelling (e.g., Bronstert *et al.*, 2002; Robinson *et al.*, 2003, Salazar *et al.*, 2012).

11.5.1 Floods

In the case of flooding, a number of innovative methods using new technologies have recently been introduced to aid data/monitoring-based assessments and modelling. Numerous remote sensing methods have been developed for plot, hill slope and catchment scales monitoring of floods (Robinson *et al.*, 2008). Advances in airborne, satellite and radar-based remote sensing have increased areal coverage, refined temporal resolution and the reliability of information (Moore *et al.*, 2012), (e.g. for integrated estimates of soil moisture in near-surface layers by microwave radiometers and scatterometers (e.g., De Jeu *et al.*, 2008), and thermal imagery technology (e.g., Su *et al.*, 2003). The exploration of geophysical methods for hydrological/pedological applications has shown that ground-penetrating radar (GPR) can be used to describe soil moisture in the shallow subsurface using scales from one meter up to a kilometre (e.g., Grote *et al.*, 2003), and multi-channel GPRs can reach lower layers of soil (Gerhards *et al.*, 2008; Pan *et al.*, 2012). The potential of electrical resistivity surveys for soil moisture estimation (Samouëlian *et al.*, 2005) in combination with other field methods (e.g., tracer methods) has been demonstrated (Uhlenbrook *et al.*, 2008). Microwave links can also be used for estimating precipitation (e.g., Leijnse *et al.*, 2007).

Precipitation, runoff, soil moisture and other data have been subject to spatial heterogeneity assessments using GIS methods and geostatistics, mapping, regional frequency analyses, and other means of dealing with spatial heterogeneity (Szolgay *et al.*, 2009).

As to catchment scales, revival of methods of comparative hydrology has led to conceptual and process-based hydrological models for flood typology (Viglione *et al.*, 2010; Gaal *et al.*, 2012; Salinas *et al.*, 2013) and documentation of flash floods (post-flood surveys e.g., Borga *et al.*, 2007; Blaškovičová *et al.*, 2011). For models of river floods/flooding, remotely sensed water levels and inundated areas have provided spatiotemporal patterns of flooding (e.g., Smith & Pavelsky, 2008) for the identification of flood zones (e.g., Pappenberger *et al.*, 2012; Alfieri *et al.*, 2013).

11.5.2 Landslides

Methods are required for the assessment of the place and time of occurrence of landslides, the estimation of the volumes, which can be released, and the place of their accumulation with regard to climatic or human-induced triggers. Pradhan (2011) and Piacentini *et al.* (2012) recognize different approaches that have been used to carry out landslide susceptibility assessments. These entail: (i) heuristic (index); (ii) statistical and deterministic (geotechnical, physically-based) methods; (iii) artificial neural network models, fuzzy logic; and (iv) advanced data mining methods.

Statistical and data mining methods, as well as artificial neural networks, are mainly applied on medium-regional scales (200-500 km^2) and have been developed in a GIS environment (Clerici *et al.*, 2006). As it has been well summarized by Clerici *et al.* (2010), with methods relying on the assumption that landslides are likely to occur under the same conditions (predisposing and triggering factors) as those of the past. Under these assumptions, susceptibility studies to landslides are mainly composed of four steps (Vijit & Madhu, 2008):

Step 1: the mapping of past landslides to be used to train and test the model;
Step 2: the definition and mapping of the predisposing factors (indicators);
Step 3: the definition of the relationships between the occurrence of landslides and the predisposing factors, along with the test of their validity; and
Step 4: the use of these relationships to divide the study area into different classes of susceptibility.

On a small catchment or slope scale (< 100 km^2), the approach that is used to detect the possible sources and propagation areas of landslides (mass movements) is usually deterministic. To recognize the areas where landslides originate, the effects of infiltration and ground water redistribution processes on stability are often studied by coupling hydrological models with an infinite slope stability analysis (Von Ruette *et al.*, 2013). The degree of stability of the soil mass is often expressed through the Safety Factor, which is given by the ratio of the resistant and acting forces. Many models are available (Table 11.3); some incorporate steady state hydrological processes, such as SHALSTAB (Dietrich & Montgomery, 1998) and SINMAP (Pack *et al.*, 1998), while others have implemented a dynamic spatial-temporal code, the evolution of the volumetric water content in the unsaturated zone, and the water level in the saturated horizon. Among these models are SHETRAN (Burton & Bathurst, 1998), STARWARS (Van Beek, 2002), TRIGRS (Baum *et al.*, 2002), and GEOtop-FS (Simoni *et al.*, 2008).

Table 11.3: Models employed to detect landslides at catchment/hillslope scale.

Models	Explanations (briefly)	References
SINMAP	Steady state hydrologic models+infinite slope	Pack *et al.* (1998)
SHALSTAB	Steady state hydrologic models+infinite slope	Dietrich & Montgomery (1998)
SHETRAN	3D physically based hydrologic model for surface and groundwater flow+solute and sediment transport in river catchments	Burton & Bathurst (1998)
STARWARS	3D physically based model for saturated and unsaturated flow	Van Beek (2002)
TRIGRS	1D Vertical flow for both saturated and unsaturated conditions+infinite slope	Baum *et al.* (2002)
GEOtop-FS	3D physically based hydrologic model for surface and groundwater flow+infinite slope	Simoni *et al.* (2008)

11.6 Effects of flooding and landslides on other soil threats

11.6.1 Effects of flooding

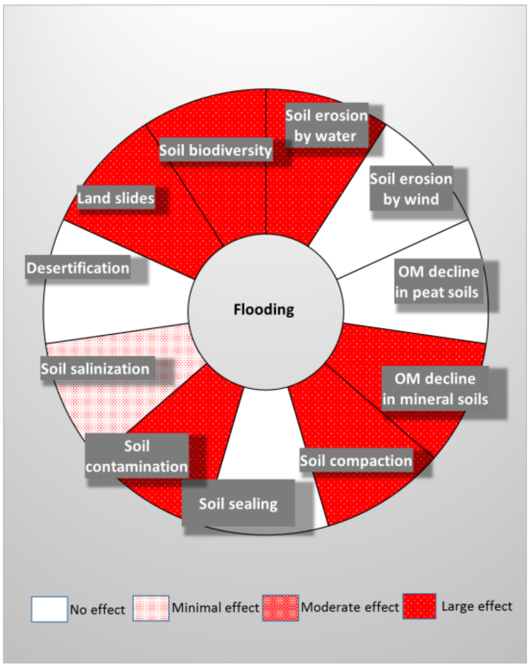

Figure 11.4: Effects of flooding on other soil threats. Red is negative effect.

Small and large-scale temporary flooding of soil can cause significant soil deterioration effects. Floods over slopes in the form of overland flow, sheet flow, return flow, groundwater ridging, etc. are obviously connected to soil erosion and landslides. Changes in soil structure and compaction due to flooding is a more problematic topic since it depends on the soil type, as well as a range of other factors. The flood water along with saturated conditions may destroy soil macro pores and the soil organisms that create a soil's structure. Under such conditions, the soil can be more susceptible to compaction, crusting, and high bulk-density problems (USDA, 2008). Soils with a high clay content can become compacted and form a surface crust after heavy rainfalls and flooding (DAFF, 2013). Soil compaction may increase due to floods if, for example, tilling and grazing is done on land that is not yet sufficiently dry (Hamza & Anderson, 2005). When agricultural practices become more industrialized, more land is often owned by fewer farmers, which means that heavy machinery is most likely to be used during conditions that are not optimal for e.g. soils with high water content. This condition in combination with increased flood frequencies and durations may enhance the compaction problem in some regions.

Provin *et al.* (2014) report on microbiological, pesticide, hydrocarbon and heavy metal releases as well as the movements caused by the flooding or inundation of containment systems, residential storage sheds and garages, chemical storage warehouses, industrial complexes, various machinery service centres, industrial areas, sewage handling and treatment systems, and livestock feeding operations. Also, high water discharges can lead to failures in wastewater treatment plants or the erosion of contaminated deposits upstream. During floods, eroded sediments are often deposited on floodplains. Therefore, soil contamination can increase as a result of floods. Specific local problems can be caused on urban soil by urban flooding induced by heavy rainfall, since this in many cases entails the flooding of combined sewer systems. Such floodwaters are likely to be contaminated and also may pose potential health risks to citizens exposed to pathogens in these waters and soil (ten Veldhuis *et al.*, 2010).

Kozlowski (1997) lists many adverse effects of flooding on plant growth with detailed references. During the growing season, all the developmental stages of flood-intolerant plants are affected (whereas, short flooding during the dormant season may have little effect). Generally known impacts include damage, the inhibition of seed germination, vegetative and reproductive growth, and changes in plant anatomy. Ausden *et al.* (2001) reported on the effects of the flooding of lowland wet grasslands on the soil's macro-invertebrates and found that unflooded grasslands contained high biomasses of soil macro-invertebrates, but grasslands (with a long history of winter flooding) contained much lower biomasses of soil macro-invertebrates, that were mainly comprised of a limited range of semi-aquatic earthworm species. Winter flooding also expelled large numbers of overwintering arthropods from the soil.

Floods may lead to a decline in soil biodiversity if anaerobic conditions prevail. Provin *et al.* (2014) lists causes such as the death of plant vegetation due to oxygen depletion in the rooting zone (seeds may not germinate in saturated rooting zones, and most plants will not grow) and the loss of plant-available soil nitrogen due to leaching or volatilization (the biological conversion to nitrogen gas by soil microbes). Flood-related water logging may potentially lead to local salinization (however, such waterlogging and salinity can be highly variable, both spatially and temporally). Other soil threats, including soil erosion by wind and desertification, may be only marginally impacted by floods, or not at all.

11.6.2 Effects of landslides

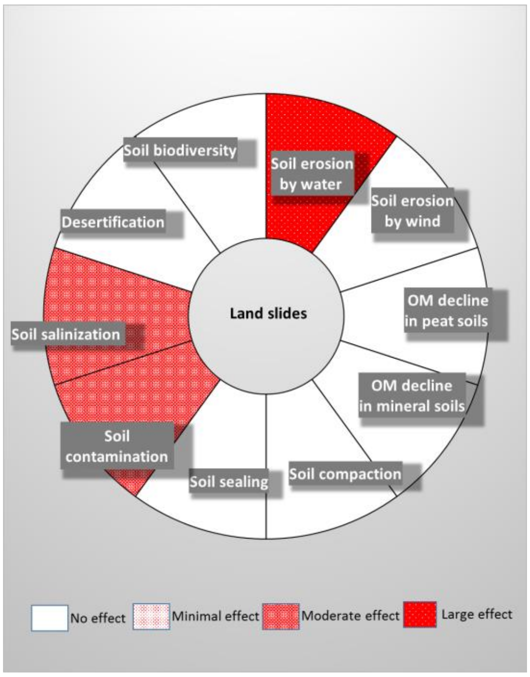

Figure 11.5: Effects of landslides on other soil threats. Red is negative effect.

The linking of landslides to soil erosion by water is fairly intuitive and evident as landslides can be seen as a primary source of punctual erosion by increasing the sediment yield in the drainage basins where they occur (e.g., Borrelli *et al.*, 2014). On the other hand, landslides can also be seen as a secondary source of erosion, since the material that accumulates in the deposition area is looser than in the neighbouring areas. In the first stages, it is evidently not covered by vegetation, so it is very prone to erosional processes driven by different factors (water and wind). As an example, de Vente *et al.* (2007) reported an increase in sediment yields due to the delivery of sediments in river channels linked to landslide activity. Malamud *et al.* (2004) stated that after some years, it is often almost impossible to recognize an area affected by small landslides due to erosion processes and re-vegetation.

Landslides transform substrates in complex ways, by mixing up soil, vegetation and the superficial level of the underlying bedrock (Geertsema & Pojar, 2007). Such a sudden and complex rearrangement of a landscape can also differentiate the movement of organisms and the development of ecosystems in areas affected by the landslide, including those in its proximity and undisturbed ones (Hupp, 1983), thereby leading to a rejuvenations of the soil (Wilcke *et al.*, 2003) and ecology (Geertsema *et al.*, 2009). Restrepo *et al.* (2009) succinctly summarizes how landslides can influence the richness of species, plant abundance, and soil nutrients over both short (< 5 year) and long (> 5 years) timescales. In human-modified environments, especially industrial sites and areas that have been intensively cultivated, if a landslide occurred, it would most probably lead to the increase of erosion potential and release and transport of contaminating substances (Ohlson & Serveiss, 2007). Very few studies have dealt with this type of multi-risk scenario, which is particularly critical when the contaminated sediment can reach rivers (Göransson *et al.*, 2012), even if the risk of landslides occurring in polluted areas is increasing due to climate change and unsustainable development (Göransson *et al.*, 2014). Some landslides actually flow and might therefore travel further. In that case, they have an obvious impact on the site where the sediment is deposited.

Short-lived landslide dams that form and fail within the duration of a rainfall-induced flood event in mountainous environments (i.e., a few hours to some days) can generate flash floods or aggravate flooding in a basin (Catane *et al.*, 2012). They occur in many steep land areas but are often small in volume (Costa & Schuster, 1988). Muddy floods are a common phenomenon in the Loess areas of Europe, though with much lower sediment content. Floods are not always harmful, floods are useful particularly in arid lowlands where agriculture is possible by diverting small to medium-sized floods to irrigate adjacent fields. These seasonal floods spring from highland and mountainous areas and contain fertile sediments and nutrients.

11.7 Effects of flooding and landslides on soil functions

The complexity of the interactions of flooding and landslides with soil functions is difficult to assess. Flooded soils can lead to significant soil deterioration. Among the many direct effects of temporary flooding on soil health (including soil texture, structure, water holding capacity, fertility and nutrient availability, etc.) are erosion, mudflows, deposition of sediments and debris, soil crusting, nutrient leaching, changes in microbial and fungi populations, changes in soil chemical properties, deterioration of soil aggregation, and temporary water logging (e.g. Kozlowski, 1997; Barrett-Lennard, 2003; Parent *et al.*, 2008; Unger *et al.*, 2009a,b; Stankoviansky *et al.*, 2010).

Of the six soil functions, the most obvious and often recognised impacts by floods are probably those on food and biomass production (e.g. O'Connell *et al.*, 2004; SG, 2009). Floods will affect food production either through soil erosion and the leaching of nutrients (usually upstream), or by the inundation and siltation of agricultural land (usually downstream) (e.g., Larson *et al.*, 1997; Dotterweich, 2008). The gradients and variability of soil flooding and variations in plant traits associated with their tolerance of flooding can have a major impact on the distribution and abundance of plant species in certain natural ecosystems (Voesenek *et al.* 2004; Vashisht *et al.*, 2011). The flooding of soil varies in depths and durations and ranges from the flooding of roots only (waterlogging) to the submergence of plants (almost dark conditions in turbid waters) (Colmer & Voesenek, 2009). Although floods may harm the biological life in ecosystems, many ecosystems are well adapted to the occurrence of floods.

With respect to the preservation of cultural heritage the loss of archaeological sites presently preserved close to the ground surface can be caused by flood risk alleviation schemes and flooding. Certain artifacts are only preserved in waterlogged/anaerobic/anoxic conditions. Flood events demonstrate devastating effects on the behaviour of the foundations of cultural heritage sites in their interaction with subsoil. Herle *et al.* (2010) gives an overview of the different phenomena, which arise, in subsoil and on the foundation level during flooding, which causes groundwater to rise; the overview is accompanied by several case histories relating to cultural heritages and discusses possible geotechnical measures.

Soil as a platform for man-made structures (such as buildings and highways) is affected by floods, but most often the impact is not attributed to the soil itself. The soil is, in these cases, usually covered by an infrastructure (road, foundations, and urban sealed surfaces). Foundations exposed to (repeated) flooding are not supported by subsoil from the bottom or cannot reach their design bearing capacity due to the lack of soil overburden. Erosion can lead to the loss of a significant soil volume below foundation structures, thus producing deformations and cracks in the superstructure. An uneven settlement or a collapse of the whole structure can occur.

Many soil functions are affected in areas prone to landslides. The effects of landslides are similar to those listed for floods, affecting the stability and functionality of the structure and sometimes completely destroying it. The effects of landslides on other main soil functions (food production, biological habitats, environment interaction, physical and cultural heritages, sources of raw materials) are related exclusively to the actual occurrence of mass movements. In the case of landslides that affect cultivated or natural areas, food, biological and environmental functions are lost in a very short period. However, landslides can lead to a rejuvenation of soils favouring the development of new biological and ecological systems and the restoration of soil functions in a short time period (< 5 years) (Restrepo *et al.*, 2009).

In countries with an extensive cultural heritage and complex topography (e.g., Italy), it is possible to find archaeological sites threatened by landslides and hydrogeological hazards in general (e.g., Canuti *et al.*, 2000; Sdao & Simeone, 2007). In order to avoid the loss of cultural heritage sites, site-specific remediation measures, depending on the local geological and geomorphological setting and landslide type, are usually suggested.

Di Baldassare *et al.* (2014) called attention to the fact that state-of-the-art methods for risk assessment typically treat natural and social systems (and ecological) separately (e.g., hydrology focuses on flood hazards, whereas socio-economics and ecology focus on exposure, vulnerability, and resilience to floods). This also applies to flooding, landslides, and soil functions. Therefore, the interconnections and feedbacks between the different components of risk may remain unknown and risk prevention costs and ecosystem services may be incorrectly valued. As a solution, the dynamics of risk should be investigated further.

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12 DECLINE IN SOIL BIODIVERSITY

Mark Tibbett

12.1 Description of decline in Soil Biodiversity

Biodiversity is a relatively recent concept first used in 1988 (Wilson and Peter, 1988) and has been defined in many and varied ways but most simply put is the variety of life. Soil biodiversity is generally defined as the variability of living organisms in soil and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems (UNEP, 1992). For the purposes of soils and its complexity of habitats a more detailed definition needs to encompass the full range of the variety and variability of living organisms and the ecological complexes in which they occur, and encompasses ecosystem or community diversity, species diversity, phenotypic, genetic and functional diversity (after Jensen *et al.,* 1990). Soil biodiversity can be described in many ways including:

- **Ecosystem diversity** which encompasses the variety of habitats that are in the soil.
- **Species diversity** is the variety and abundance of different types of organisms which inhabit a soil. This is akin to taxonomic diversity.
- **Genetic diversity** is the combination of different genes found within a population of a single species, and the pattern of variation found within different populations of the same species. This can also be assessed across the whole community of organisms.
- **Phenotypic diversity** is based on any and/or all of the morphological, biochemical or physiological aspects of the organism in the soil and is a result of genes and environmental factors.
- **Functional diversity** is the variety of functions performed by the soil biota such as nitrification and litter comminution.

Soils are a globally important reservoir of biodiversity. They contain at least one quarter to one third of all living organisms on the planet yet little is known about them, as only ca. 1% of soil microorganisms have been identified compared to 80% of plants (Jeffery *et al.*, 2010).

The threat - decline in Soil Biodiversity - is generally considered as the reduction of forms of life living in soils, both in terms of quantity and variety (Jones *et al.,* 2005). The ENVASSO project (Huber *et al.*, 2008) proposed the following description of the threat decline in soil biodiversity; “reduction of forms of life living in soils (both in terms of quantity and variety) and of related functions”.

Wherever soil biodiversity decline occurs it can significantly affect the soils’ ability to function normally and respond to perturbations and on the capacity to recover. Decline in soil biodiversity is usually related to other deteriorations in soil quality and can be linked with other threats like erosion, organic matter depletion, salinization, contamination and compaction. Soils are remarkably complex and dynamic environments and hence typically comprise a wide range of habitat types for organism over a range of dimensions from micrometre to the landscape scale. It is this highly heterogeneous nature of soil, particularly at the microhabitat level, that is responsible for its considerable biodiversity (Jeffery *et al.*, 2010). At its simplest the vast biodiversity of the soil can be divided into four major groups. These are the microbes and microfauna with body widths of less than 100 micrometres, the mesofauna with body widths between 100 micrometres and 2 millimetres, and the macrofauna which are larger than 2 millimetres (Wurst *et al.*, 2012; Swift *et al.*, 1979). While the size boundaries for classification into micro, meso, and macro are universally agreed, published groupings and classifications can vary as some taxa cross size boundaries, and may be interpreted in terms of body width or length (eg. Swift *et al.*, 1979; Coleman and Crossley 1996).

The microbes are the smallest group in physical dimension yet the most abundant and, despite their size, may comprise the largest biomass. The microbial community is the most diverse group of organisms, not only in the soil but arguably on the planet. The major taxa comprise bacteria, archaea, fungi and viruses (Fierer *et al.* 2007). The biodiversity of soil bacterial communities alone is enormous where one gram of soil may contain anything from ten thousand to ten million taxa (Torsvik *et al.*, 2002, Gans *et al.*, 2005, Tringe *et al.*, 2005). Most microorganisms are heterotrophic (particular in aerobic soils), while others are autotrophs, with the chemoautotrophs forming a particularly important group in nutrient cycles and climate change biogeochemistry.

The microfauna consist of tiny soil animals that are dominated by three main groups: the Protozoa (include the amoebae); Nematoda; and Rotifera (Wurst *et al.*, 2012) (Figure 12.1). They usually required water films or water filled pores to move around the soil (Coleman and Crossley 1996).

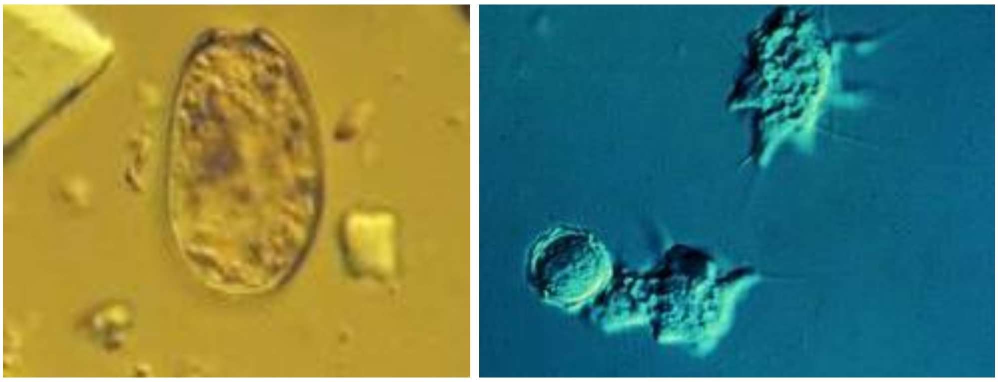

Figure 12.1: Soil microfauna, including a protozoa (left panel) and amoebae (right) panel

The protozoa are the most diverse group and are single celled eukaryotes. They are primary and secondary consumers, predominantly feeding on bacteria and fungi with some saprophytic taxa also present in the soil. The multicellular nematodes, which are small round worms, and have a wide range of feeding strategies and may be microphagous as well as plant root pests. The Rotifera, which are also multicelled, primarily feed on bacteria and algae (Wurst *et al.*, 2012).

The mesofauna include arthropods, such as mites, collembola (springtails) and enchytraeids (Figure 12.2) and many other groups (Jeffery *et al.*, 2010). They tend to occupy air-filled pores in soil and litter and feed on the microbes and microfauna as well as plants and algae.

Figure 12.2: Enchytraeid worm (top left panel), collembola or "springtail" (top left panel), Acari (red velvet mite) (bottom left) and a pseudoscorpion (bottom right).

The macrofauna includes snails, slugs, earthworms, ants, termites, millipedes, woodlice and larger animals, such as moles, badgers and rabbits. Burrowing animals, such as earthworms, ants and millipedes create their own living space by burrowing into the soil and as such can alter the soil. These groups are sometimes referred to as "ecosystem engineers" (Jones *et al.*, 1996).

12.2 State of soil biodiversity decline

As described above, soil biodiversity is so extensive that, when compared to other components of the global ecosystem, it seems to be in good health. Belowground biodiversity can often be much higher than above ground biodiversity. Soil biodiversity, however, does not decline independent of other factors and is usually related to some other deterioration in soil quality. This represents a decline in the quality and/or number of biological habitats in the soil that support soil biodiversity. In general and geographical terms, the state of soil biodiversity has been well described in the European Atlas of Soil Biodiversity (Jeffery *et al.*, 2010). This unique resource for Europe, while written from a European point of view, includes soil biodiversity assessments globally particularly from extreme environments. The Atlas tries to address a fundamental problem with soil biodiversity: if we do not know what is out there, how do we know if it is in decline? Even with this resource it is challenging to gauge at national, European and global scales.

At local levels it is clear that biodiversity is in decline. For example, soil sealing (the permanent covering of soil with hard surfaces, such as roads and buildings) causes the death of the soil biota by cutting off water and carbon and nutrient inputs (Turbé *et al.*, 2010). In this extreme case, not only is all biodiversity lost but practically all biology. In other cases, soil biodiversity decline can be linked with erosion, organic matter depletion, salinization, contamination and compaction (Montanarella, 2007). Wherever soil biodiversity decline occurs it is of concern as it can significantly affect the soils' ability to function normally and respond to perturbations.

Soil biodiversity is subject to considerable disturbances through any number of threats. The soil biota has its own unique capacity to resist events that cause disturbance or change and a certain capacity to recover from these perturbations. The capacity to recover from change is considered a key attribute of biodiversity. Figure 12.3 provides a simple schematic that describes the concept of *Resistance* and *Resilience*.

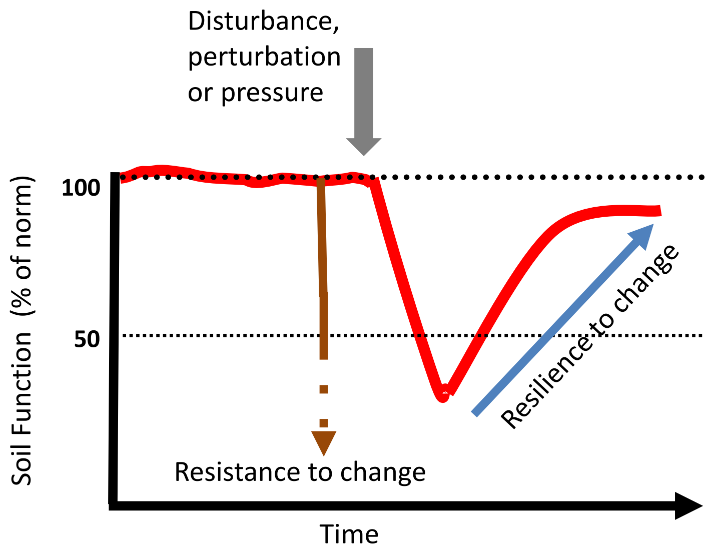

Figure 12.3: Simple model showing the effect of a perturbation on the resistance and resilience of a soil biological function or property. Higher biodiversity is thought to correspond to high resistance and resilience. A loss of biodiversity is thought lead to a soil with lower resistance to a perturbation and lower capacity to recover.

Soils with higher biodiversity are thought to have an innate resistance and resilience to change. A loss of biodiversity is thought lead to a soil with lower resistance to a perturbation and reduced capacity to recover (Allison & Martiny, 2008; Downing *et al.,* 2012).

12.3 Drivers of soil biodiversity decline

There are many pressures on soils that can cause a loss in biodiversity and these include human activities with local land management matters, socio-political factors as well as more universal climate change effects.

Land management can have varying effects on belowground biodiversity. A primary driver of this is the close link between soil biodiversity and soil organic matter, although the relationship is not fully understood (Six *et al.,* 2006). Loss/decline of soil carbon has been a general feature of tillage agriculture (Janzen, 2006) and carbon losses have been found to be occurring at a national scale in the UK for example (Bellamy *et al.*, 2005). Gobin *et al.*, (2011) estimated that 13-36% of the current soil carbon stock in European peat lands might be lost by the end of this century. As the source of energy underpinning food-webs, carbon losses may lead to reduced biodiversity. Coupled to this, the general use of fertilizers, pesticides and herbicides as part of agricultural intensification are a significant cause of soil biodiversity loss. The pressures that cause agricultural intensification include: population growth, food production disparities, urbanization, and a growing shortage of land suitable for agriculture, are the underlying drivers for local and regional soil biodiversity loss. This, of course, must be coupled with the effect of agricultural policies, which may also lead to land management practices that degrade soils and their biological diversity. The agricultural techniques/management that lead to loss of soil biodiversity are monoculture cropping, removal of residues, soil erosion, soil compaction (both due to degradation of the soil structure) and repeated application of pesticides (Wachira, 2014).

Climate is considered a potential important factor in driving future soil biodiversity decline (Suárez *et al.*, 2002). Generally, soil organisms have a relatively wide tolerance to temperature variations, and the warming (or cooling) of soils which are buffered diurnally and seasonally (particularly at depth) (Tibbett & Cairney, 2007) means that the direct effects of temperature change are unlikely to be a key factor in itself. It is the global ecosystem-scale effects that bring-to-bear strong changes to 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 as described above.

For Europe, the main pressures have been recognised for the three levels of biodiversity: ecosystem, species and gene (Jefferey *et al.*, 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 regimes and change of geochemical properties. At the level of species of organism in the soil, the main pressures on biodiversity where thought to derive from a change in environmental conditions of geochemistry, competition with invasive species and ecotoxins. At the genetic level, the main pressures where thought to derive from a change of environmental conditions, ecotoxins and “Genetic pollution” (Jefferey *et al.*, 2010).

12.4 Key indicators of decline in soil biodiversity

Measurement of biodiversity is often fiendishly challenging. In the soil, the difficulties encountered are compounded as the assessment must be done in an opaque medium and where the relative importance of taxa is unclear or unknown. Rapidly changing environmental factors, such as water content and soil air chemistry combined with the soil’s innate heterogeneity only help to compound matters.

While the ENVASSO project’s (Huber *et al.*, 2008) indicators for decline in soil biodiversity were divided into indicators for species diversity and indicators for biological functions, there are established sub-disciplines within ecology that deals with the biodiversity and its measurement (Magurran, 2004). There are several commonly used ways of assessing diversity that are based on calculated indices. The most simplistic measure is *Species Richness* which is simply the number of species present in the soil, or more accurately in the sample(s) of soil taken. The problem with this index is that common species are found with little sampling effort and more with greater sampling effort and it is no longer recognised as a comprehensive measure of biodiversity. In order to avoid this potential misinterpretation of diversity numerous indices have been developed that suit a variety of environmental and community circumstances (see Magurran, 2004 for a comprehensive discussion). Two biodiversity indices that are used commonly calculate relative abundance from proportions (p_i) of each species (i) within the total number of individuals. Simpson’s Index:

$$D = \frac{1}{\sum p_i^2} \quad [12.1]$$

Where D equals diversity. For any number of species in a sample (S), the value of D can range from 1 to S. As D increases diversity decreases so this index is usually expressed as 1-D or 1/D. In contrast, the Shannon-Weaver Index (H) is a logarithmic measure of diversity:

$$H = -\sum p_i log_e p_i \quad [12.2]$$

The higher H, the greater the diversity. Because H is roughly proportional to the logarithm of the number of species, it is sometimes preferable to present data as e^H, which is proportional to the actual number of species.

The scale at which biodiversity is considered is also important and the type of biodiversity measured is dependent on whether comparisons are made with or between soils (habitats). *Alpha-diversity*, or within-habitat diversity, refers to a group of organisms interacting and competing for the same resources or sharing the same environment or soil. This is measured as the number of species within a given area. *Beta-diversity*, or between-habitat diversity, refers to the response of organisms to spatial heterogeneity. High beta-diversity implies low similarity between species composition of different soils or habitats. It is usually expressed in terms of similarity index between communities between different habitats in same geographical area. *Gamma diversity*, or landscape diversity, refers to the total biodiversity over a large area or region. It is the total of α and β diversity.

Functional redundancy and diversity
Functional diversity considers the variety and number of taxa that undertake contrasting functional roles in the soil. Measuring diversity in this way allows an emphasis on what the biodiversity does rather than the species that comprise the diversity.

Species diversity is thought to be important because it is synonymous with ecosystem health, as this leads to ecosystem supporting functions and ultimately ecosystem services. However in soil, primarily due to its tremendous heterogeneity, there is often considerable functional redundancy (Walker, 1992; Wellnitz & Poff, 2001). Functional redundancy within a soil is where certain species contribute in equivalent ways to precise functions such that one species may substitute for another. In effect this means that the loss of taxonomic diversity may not necessarily lead to the loss of soil functions as more than one species may be doing the same job. However, these organisms may not necessarily occupy the same ecological niche, one species may not survive flooding, whereas another may not survive freezing. In this way, taxonomic diversity of species may remain important.

12.5 Methods to assess status of soil biodiversity

Monitoring soil biodiversity has been encouraged as a method of assessing soil quality and health and to inform management and policy (Jeffery *et al.*, 2010). It also allows for the detection of biodiversity decline and to enable remedial measures. To best characterise the soil biota, the protocols from sampling to analysis should enable representation of both the complexity and the high temporal and spatial variability. While a detailed discussion of methods is beyond the scope of this paper, these should be based on standardised, quantitative and repeatable protocols of sampling and estimation of soil biodiversity (e.g International Organisation for Standardisation - ISO 23611 series).

A recent study has evaluated numerous separate methods that assess the soil biota with a view to finding the best single measure, or multiple measures, to provide indictors of soil health, all of which have a role in diversity indices discussed (Black *et al.,* 2011; also see Ritz *et al.,* 2009). The main methods considered were:

- Multiple enzyme fluorometric assays to profile the activity of soil enzymes
- Multiple substrate induced respiration
- Genetic profiles of soil microbial community structure
- Lipid profiles soil microbial community structure and biomass
- Assessment of soil nematode communities
- Assessment of soil microarthropod communities

The authors concluded that there was no universal method that provided an overall measure of soil biological health. Instead they settled on a suite of soil biological methods to provide an informative approach to monitoring changes in soil biology, which would be particularly robust when multiple threats to the soil system interact, or where the pressures influencing soil are unknown. The recommended suite included:

- Phospholipid fatty acids (PLFAs), these are signature lipid biomarkers of soil organisms that are widely used to study soil microbial communities (c.f. Zelles, 1999).
- Multiplex terminal restriction fragment length polymorphism (TRFLP) for rapid and simultaneous analysis of different components of the soil microbial community, including fungi, bacteria and archaea (Singh *et al.*, 2006).
- Multiple Substrate induced respiration (MSIR), also known as community level physiological profiling (CLPP), now developed in Microresp™ (Degens & Harris 1997; Campbell *et al.*, 2003)
- The extraction and assessment of microarthropods using established techniques (van Straalen 1998; Tullgren 1918).

It is worth noting that 454 pyrosequencing which enumerates and contrasts microbial diversity in soil (Roesch *et al.*, 2007), was an important technique omitted from the above study. This technique may hold great promise for the future, but is currently an expensive option.

Black *et al.* (2011) provided a useful framework for classifying methodological approaches as indicators of soil diversity. Using their recommended suite of methods above these are:

- **Genotypic** – Including the assessment of actinomycetes; ammonia oxidisers; Archaea; denitrifiers; eubacteria; fungi; methanogens; methanotrophs using TRFLP
- **Phenotypic** – that would include PLFA profiling and microarthropod assessment
- **Functional** – Employing multiple substrate-induced respiration

These fit well into the categories of diversity outlined at the beginning of this chapter and form a useful framework to link methods with indicators.

12.6 Effects of soil biodiversity decline on other soil threats

The decline in soil biodiversity is usually related to other deteriorations in soil quality and can be linked with other threats like erosion, organic matter depletion, salinization, contamination and compaction as described in 12.1. An expert group at JRC has, illustrated in Figure 12.4, weighted the potential threat – for a selection of possible soil threats - to soil biodiversity (Jeffery *et al.* 2010). This illustrates that soil biodiversity is highly influenced by the other threats.

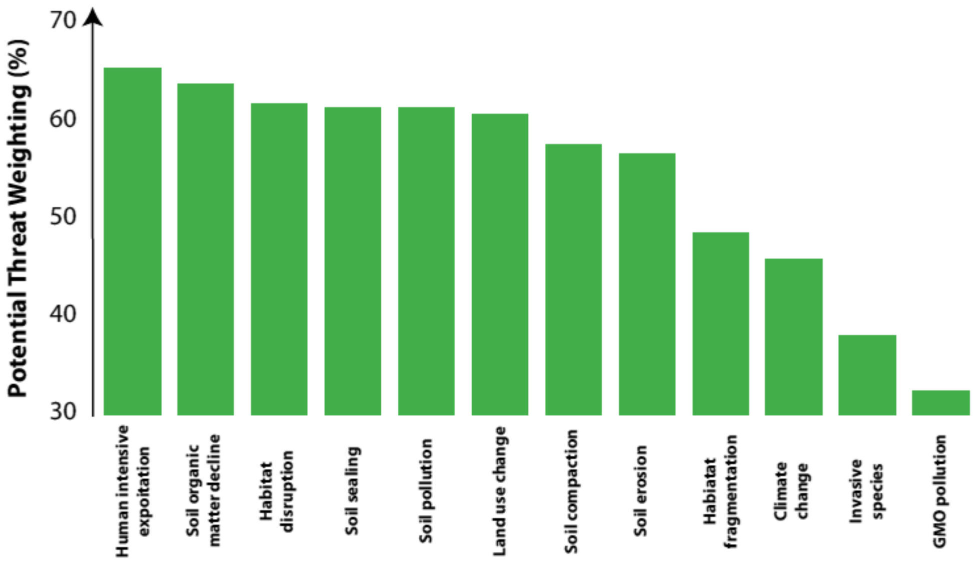

Figure 12.4: The potential threat weigthing given to a selection of possible soil threats to soil biodiversity by the expert working group at the JRC on 2nd March 2009 (after Jefferey et al., 2010).

Although less obvious, there are ways that a decline in biodiversity can affect other soil threats. One can think, for example, of climate change or soil management being the pressure for causing a loss of a species (=decrease in biodiversity), which can lead to loss of function when there is little functional redundancy, meaning for example reduced breakdown of some xenobiotic compounds, i.e. increased soil contamination.

Even when another soil threat causes a decline in soil biodiversity, it can, in turn, have effects on other soil threats. Imagine antibiotics in manure reducing microbial activity, which reduces soil respiration, thereby increasing SOC. Most of these effects are, however, very poorly understood.

12.7 Effects of soil biodiversity decline on soil functions

Activities of the soil biota are essential to most of the soil functions These stretch much beyond supporting the production of food and fibre and extend into functions, such as erosion control and pollution attenuation.

The soil functions that will be considered in RECARE are the ones identified in the Soil Thematic Strategy.

- Food and other biomass production.
- Environmental interaction; Storing, filtering, buffering and transformation.
- Biological habitat and gene pool.
- Physical and cultural heritage.
- Source of raw materials
- Platform for man- made structures; buildings, highways

The primary services include (i) nutrient cycling; (ii) regulation of water flow and storage (iii) regulation of soil and sediment movement; (iv) biological regulation of other biota (including pests and diseases); (v) soil structural development and maintenance; (vi) the detoxification of xenobiotics and pollutants; and (vii) the regulation of atmospheric gases.

Numerous conceptual models have been designed that attempt to capture the link between ecosystem function, ecosystem services and the soil biota (e.g Brussaard, 2012; Kibblewhite *et al.*, 2008). They all show that the activities of the soil biota are essential to provide most of the ecosystems that are considered typical of the wider landscape (Figure 12.5).

The ecosystem services concept provides an understandable and translatable outcome of the role of soil biodiversity in a manner that allows people to recognise its impacts on their lives.

Ecosystem goods and services are delivered by the functions of the soil biota. These services are many and varied and dependant on different components of the biological community.

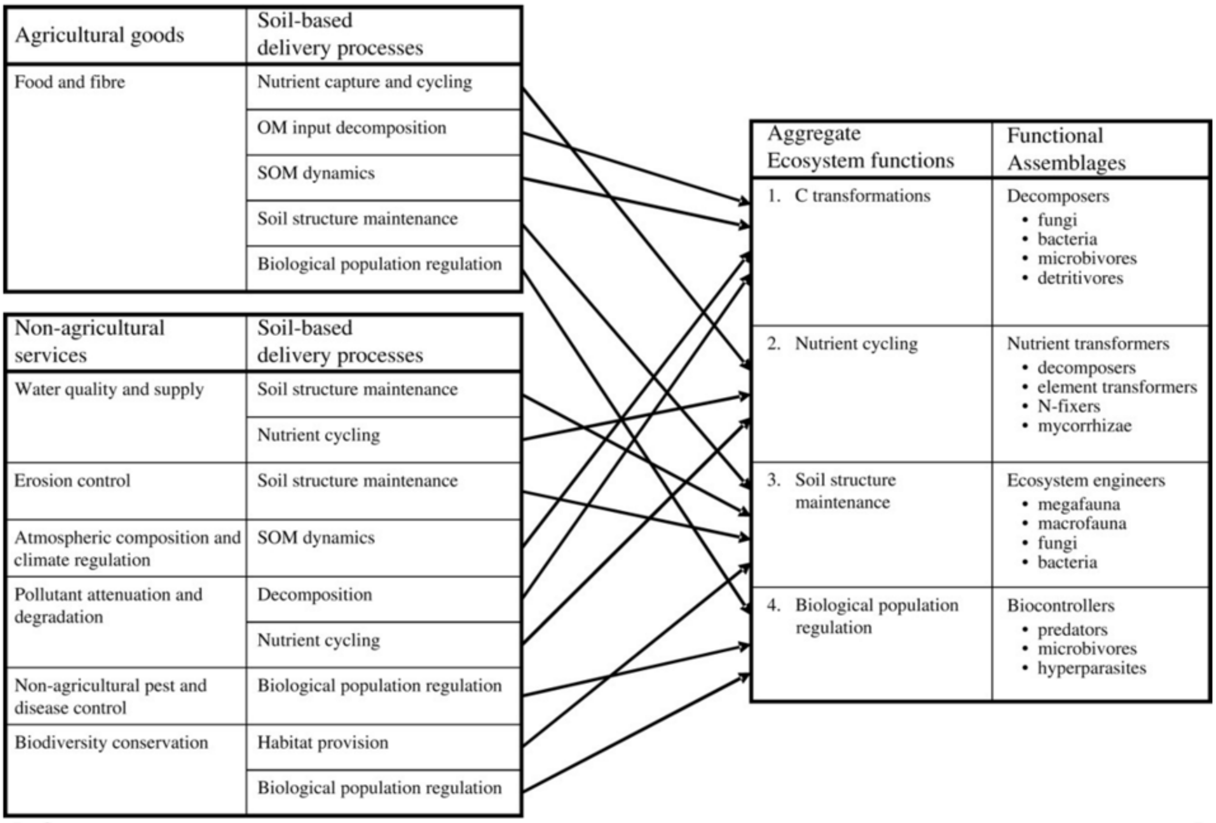

Figure 12.5: Relationship between soil biota and ecosystem goods and services (after Kibblewhite et al., 2008).

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13 SOIL FUNCTIONS & ECOSYSTEM SERVICES

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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”.

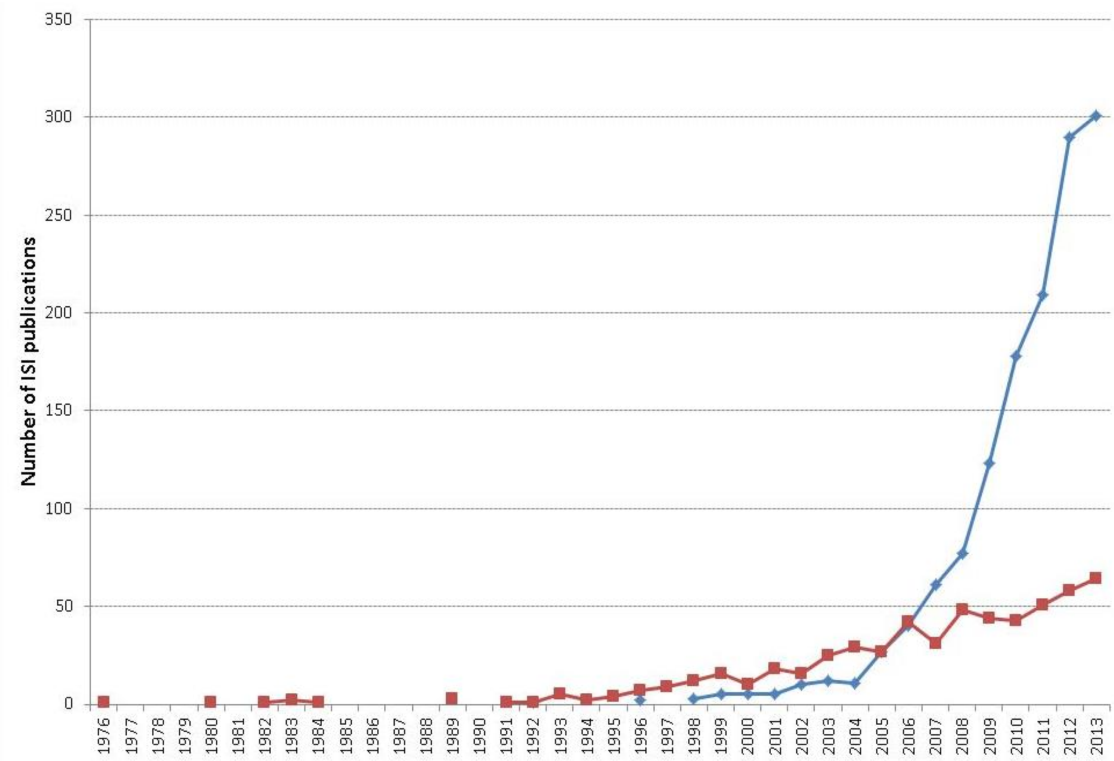

Figure 13.1: Temporal trends in ISI papers on “soil functions” and “ecosystem services”. In red are ISI papers with “soil function” in the title, abstract, or key words. In blue are ISI papers with “soil” and “ecosystem service” in the title, abstract, or key words. All searches were done in SCOPUS (25-09-2014).

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).

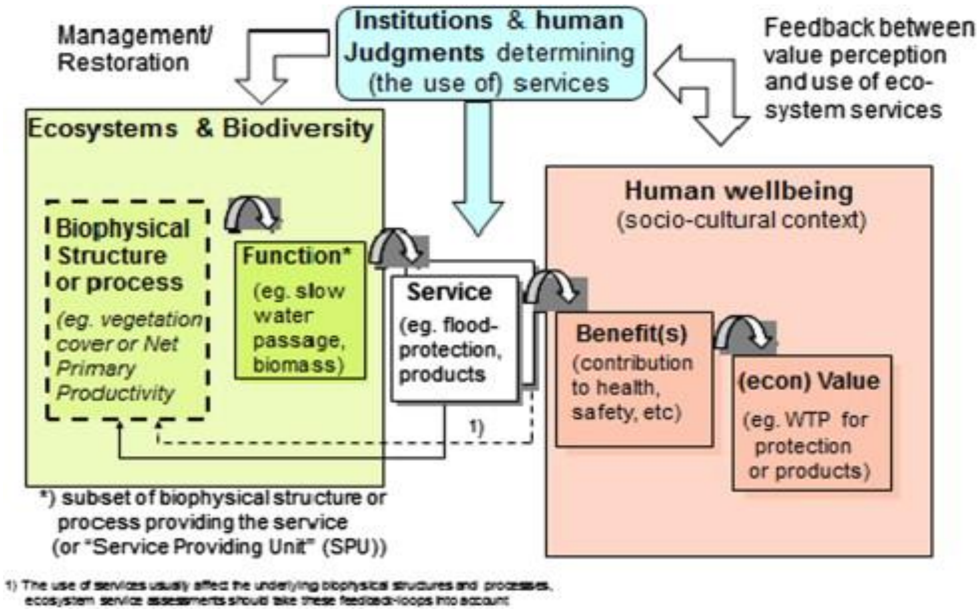

Figure13.2: The Economics of Ecosystems and Biodiversity (TEEB) overview diagram. Braat and de Groot et al. (2012), adapted from Haines-Young and Potschin (2010). As this framework was designed for economic valuation purposes it focuses mainly on economic values without considering other value systems.

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.

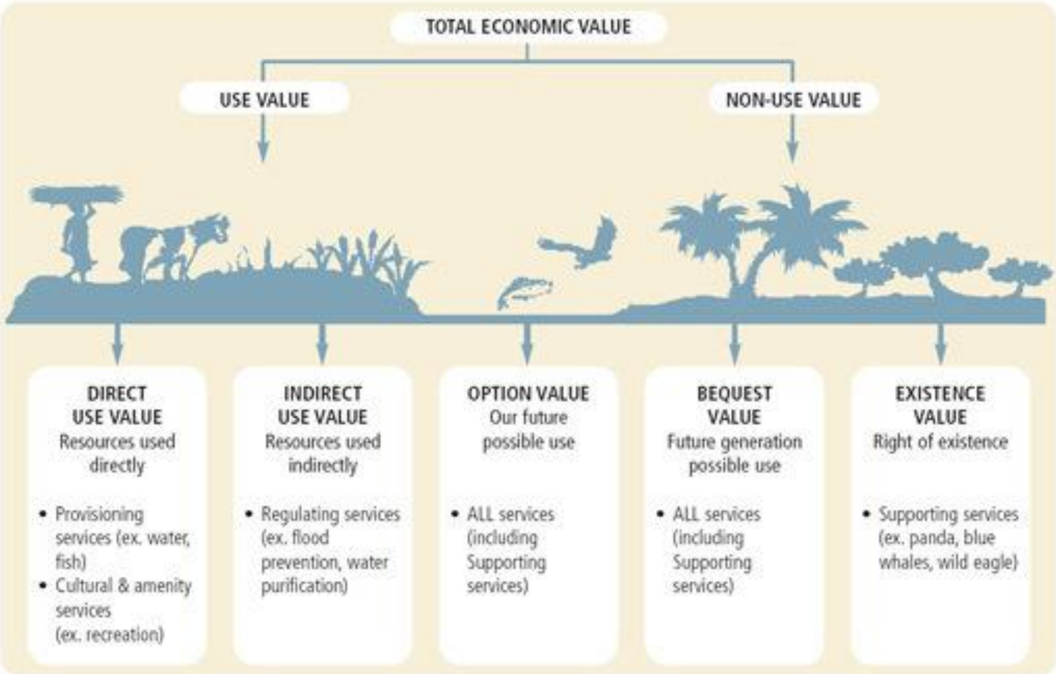

Figure 13.3: Decomposition of the Total Economic Value (TEV) of ecosystems (Smith et al., 2006).

'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.

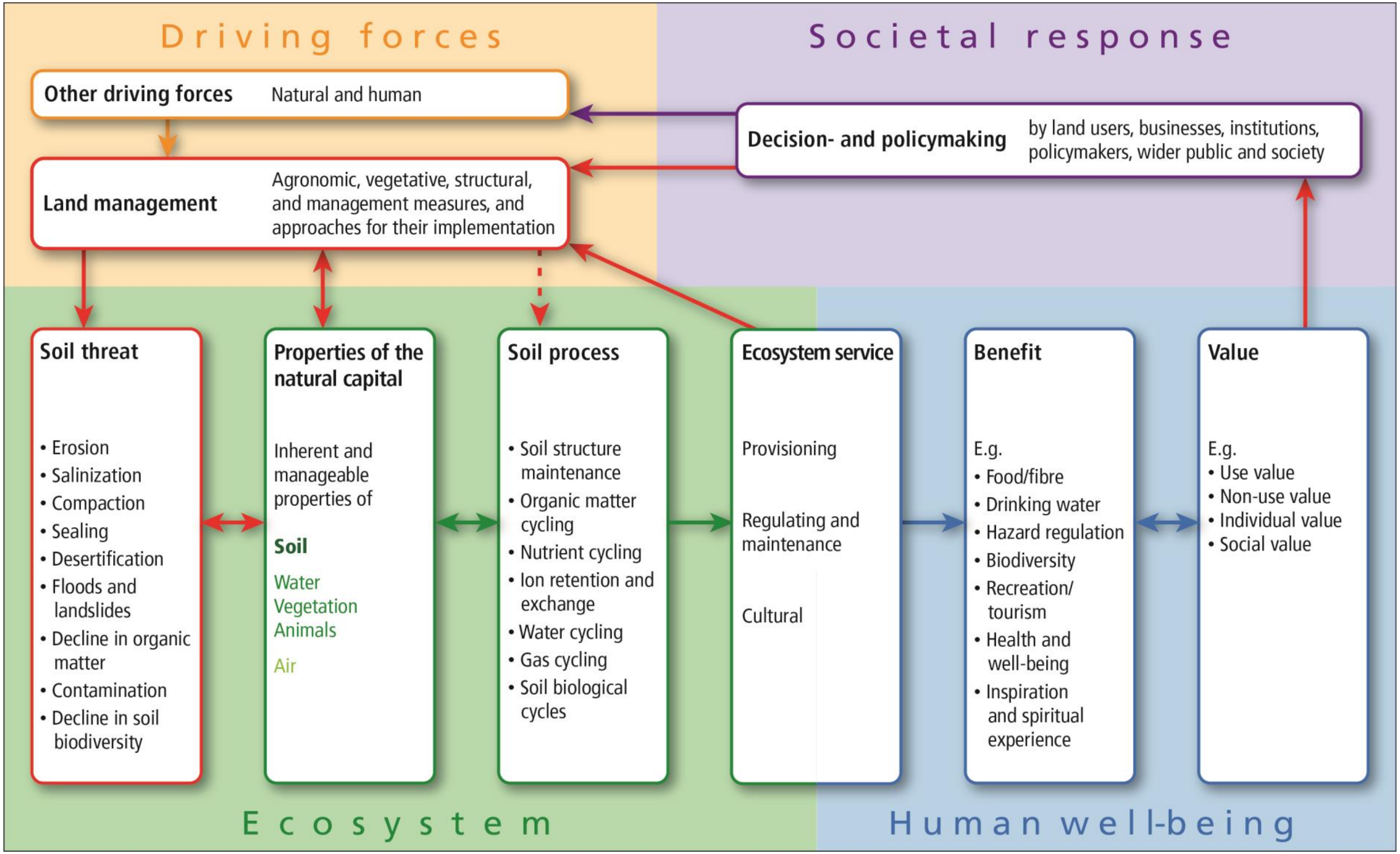

Figure 13.4: Proposed ES Framework for RECARE.

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.

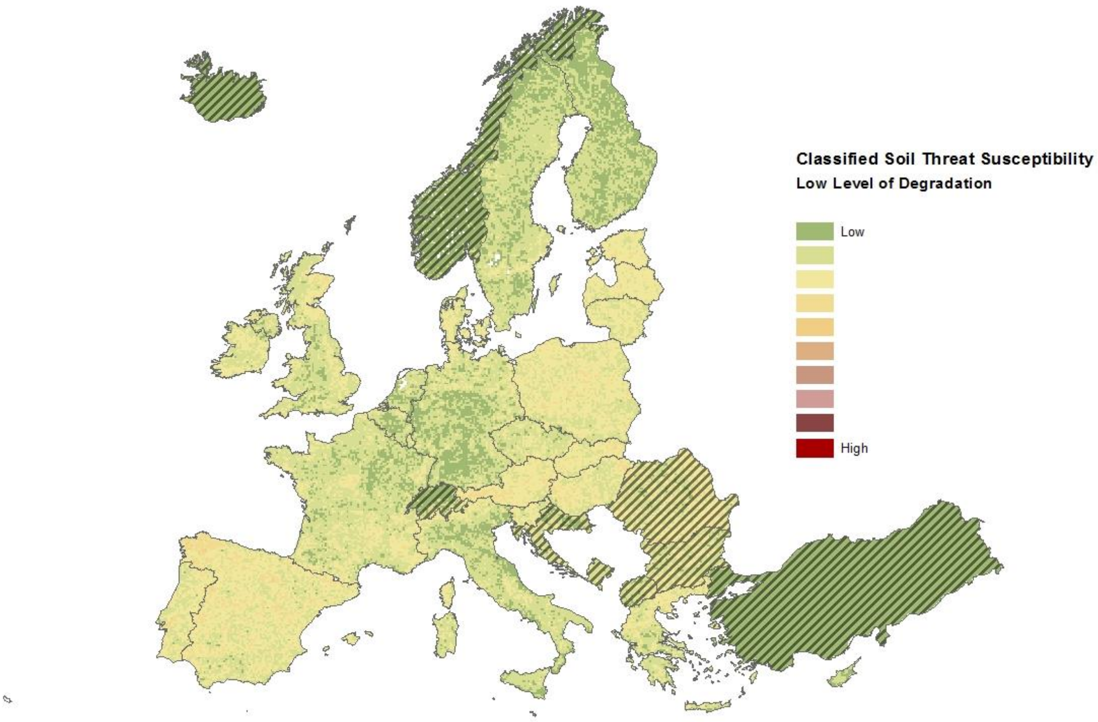

Figure 14.1: Soil threat map of Europe for the low category of degradation. For the shaded areas, not all threats are mapped.

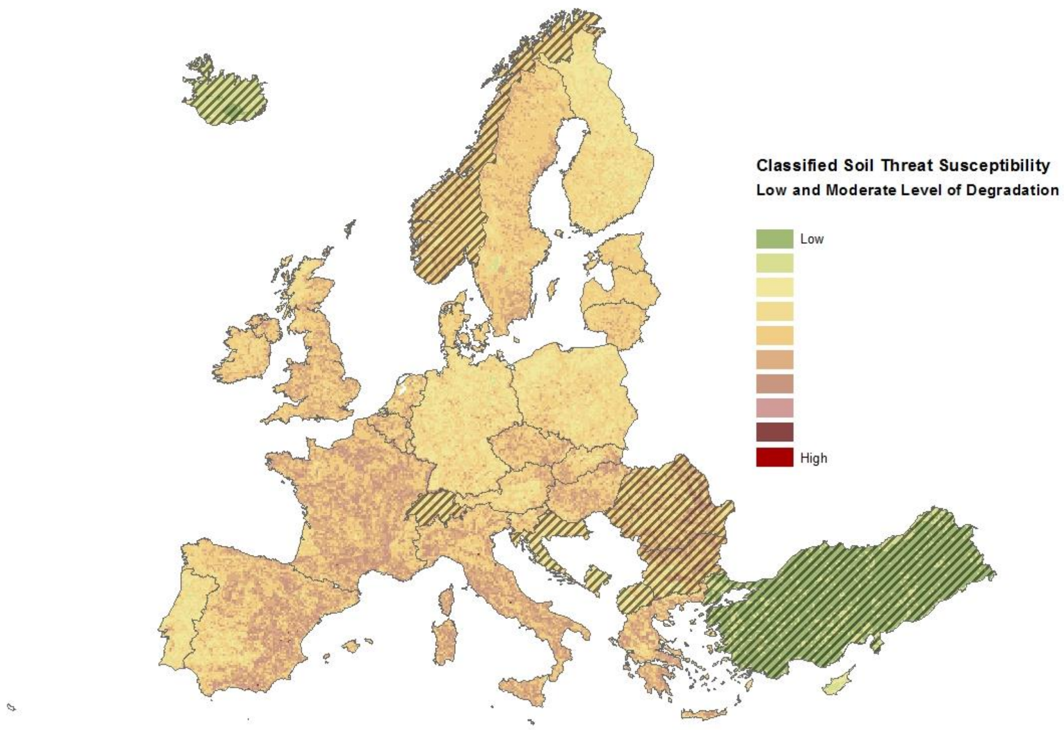

Figure 14.2: Soil threat map of Europe, summarized for the low (weighing coefficient 1) and moderate (weighing coefficient 2) category of degradation. For the shaded areas, not all threats are mapped.

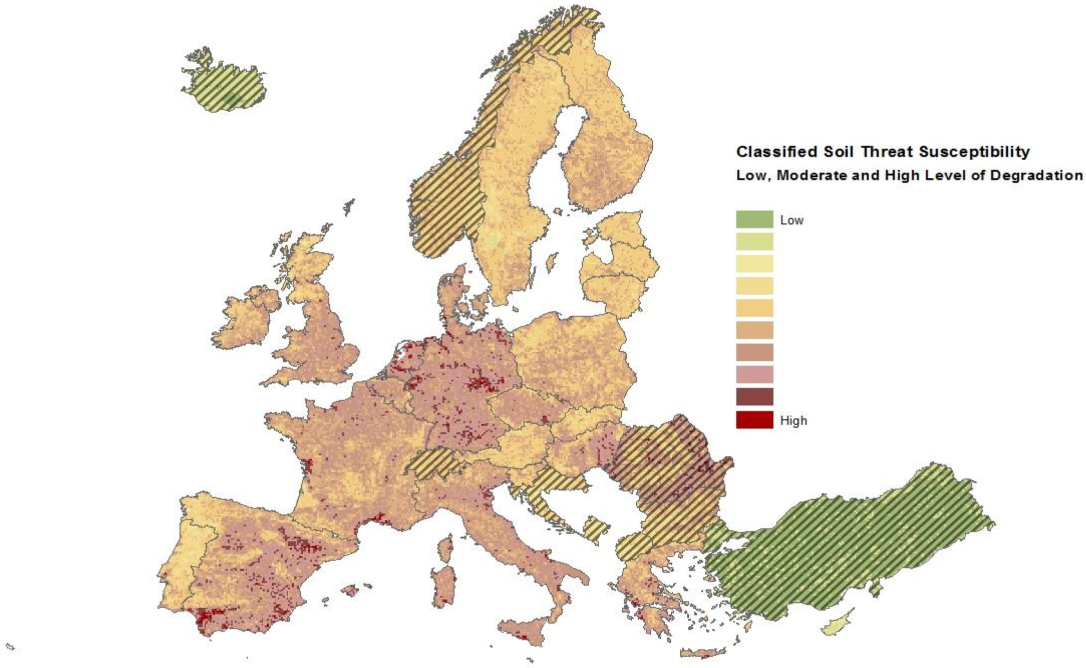

Figure 14.3: Soil threat map of Europe summarized for the low (weighing coefficient 1), moderate (weighing coefficient 2) and high (weighing coefficient 3) category of degradation. For the shaded areas, not all threats are mapped.

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	Salinizatio-n	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	$\mathrm{TI} = \frac{\%\text{ of soil class 'n' in new buil}}{\%\text{ of soil class 'n' in new whole u}}$	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/IAssessment1152619898983/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 lan	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 dono	The Economics of Ecosyste	TEEB	Draw attenti	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 n	http://www.e	yes	yes	yes	yes	yes	yes
Environmental Change Ne	NERC	Environmental Change Net	The Centre fo	Long term m	http://www.e	yes	yes	yes	no	no	no
EPSRC grant EP/F007604/	EPSRC	An evidence based metho	Loughboroug	Examining im	http://gow.ep	yes	yes	yes	no	no	no
UKPopNet Linking biodive	NERC	The UK Population Biodive	?	Funded/parti	http://www.n	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://ecofin	yes	yes	no	no	yes	yes
RUBICODE (036890)	FP6	Rationalising Biodiversity C	?	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, S	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 01LR08	Germany Federal Resear	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 1103	Academy of Finland		The Finnish F	Understandin	http://www.n	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 ef	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.o	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 - e	University of	The use of in	http://users.u	yes	yes	yes	yes	yes	no
CLIMES	Academy of Finland	Climate change impacts o	Finnish Envir	Spatial mode	http://www.a	no	no	no	yes	yes	yes
EuroDiversity AgriPopes	European Science Founda	Agricultural Policy Induced	Swedish Univ	Examine eco	http://www.a	yes	yes	yes	no	no	no

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