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POLICY DEPARTMENT
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Agriculture and Rural Development

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**Research for AGRI
Committee -
Preserving agricultural
soils in the EU**

STUDY



DIRECTORATE-GENERAL FOR INTERNAL POLICIES
Policy Department for Structural and Cohesion Policies

Agriculture and Rural Development

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Abstract

This study explains how threats to soils and soil services are linked to agricultural soil management, how threats can be mitigated, and which barriers complicate this. It highlights trade-offs and synergies that exist between different interests affected by soil management, such as climate change mitigation, water and air quality, biodiversity, food security and farm income. Conservation of peatland and extensive agro-forestry systems, and protecting soils against sealing, erosion and compaction are ranked as highest priorities. Potential policy elements are suggested.

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LIST OF ABBREVIATIONS

AKIS	Agricultural knowledge and innovation system
CAP	Common Agricultural Policy
CCCGM	Catch and cover crops and green manures
C/N ratio	Carbon to Nitrogen ratio
EAP	Environmental Action Programme
EFG	EIP-AGRI Focus Group
EIP-AGRI	European Innovation Partnership for Agricultural Productivity and Sustainability
ESS	Ecosystem services
ETS	Emissions Trading System
GAEC	Good Agricultural and Environmental Condition
GHG	Greenhouse gases
GMO	Genetically modified organism
Gt	Gigaton (10^9 ton, 10^{12} kg, 10^{15} g, 1 Pg)
IPM	Integrated pest management
ISMP	Integrated soil management plan
LTE	Long term experiment
LUCAS	Land Use/Land Cover Area Frame Statistical Survey
Mg	Megagram, ton, 10^3 kg
MS	EU Member states
Mt	Megaton (10^6 ton, 1 million ton, 10^9 kg)
NEC	National Emissions Ceilings (under NEC Directive)
NPP	Net primary production

OCTOP	Organic Carbon in Topsoil database
RSG	Reference Soil Group
SCAR	Standing Committee on Agricultural Research
SDG	Sustainable Development Goals
SGDBE	Soil Geographical Database of Eurasia
SMR	Statutory management requirement
SMU	Soil mapping unit
SOC	Soil organic carbon
SOM	Soil organic matter
STU	Soil typological unit
WLCC	Wheel load carrying capacity
WRB	World Reference Base for soil resources

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EXECUTIVE SUMMARY

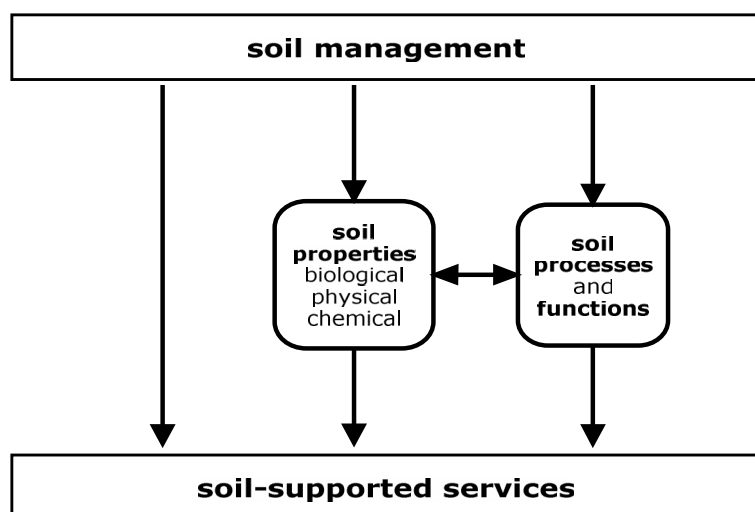
Soils and soil services

Soil, as defined in the EU Thematic Strategy for Soil Protection (COM(2006) 231), is 'the top layer of the earth's crust, formed by mineral particles, organic matter, water, air and living organisms. It is the interface between earth, air and water and hosts most of the biosphere'. Soil is differentiated into **horizons** of variable depth, which differ from the material below in morphology, physical make-up, chemical properties and composition, and biological characteristics. The most fertile portion of soil, generally the top 0-30 cm, is called '**topsoil**'. Soil fertility results from soil characteristics (e.g. texture), nutrient inputs and other management practices.

Soils are key to the delivery of a wide array of **ecosystem services**, including water and nutrient cycle regulation, food and fiber production, providing a physical basis for construction and habitat for various species. However, soils in the EU are exposed to numerous **threats**, which limit their ability to function and to deliver ecosystem services. These threats include **erosion, floods and landslides, loss of soil organic matter, salinisation, contamination, compaction, sealing, and loss of soil biodiversity**. Loss of soil functions and land degradation remain major concerns, and these will likely show continued deteriorating trends in the future (EEA, 2016).

The scale of **soil degradation** in the EU is significant with approximately 22% of European land affected by water and wind erosion. Around 45% of the mineral soils in Europe have low or very low organic carbon content, soil contamination is affecting up to three million sites, and an estimated 32-36% of European subsoils are having high or very high susceptibility to compaction. An increase in soil sealing results from construction and infrastructure. **Drivers** of soil degradation relate to increasing urbanisation, land abandonment, and intensification of agricultural production. Soil degradation reduces or

Figure 1. Schematic representation of relations between soil management, soil properties, processes and functions, and soil-supported services. Soil threats aggravated by soil management are negative impacts of practices on soil properties, processes or functions, that translate into diminished services. Soil management can also directly reduce services



eliminates **soil functioning** and their ability to support ecosystem services. It is therefore essential to maintain the services provided by soils by restoring soil functions. As such, soils should be sustainably managed, and degraded soils should be rehabilitated or restored.

For soils under agriculture, the five main **ecosystem services** include the provision of harvestable crops, clean freshwater and nutrients for plants and animals, conservation of suitable habitats for biodiversity, and maintenance of a benign climate. The corresponding **soil functions** are **primary production, water regulation, nutrient cycling, habitat support** and **climate regulation**. Relations between management and services are illustrated in the above figure.

Major threats to soils and soil services

Major soil threats at the whole-system level that call for policy action are:

- the **degradation** of **peatland** and similar **organic soils** by drainage under agriculture and forestry, mostly in Northern and North-Western Europe; and the potential **encroachment** of agriculture and forestry onto pristine peatlands as favoured by climate change in Northern Europe;
- the drastic alteration of traditional **wood-pasture systems** in various parts of the EU;
- the **grabbing of productive agricultural land** for non-agricultural use (sealing).

Any policies – new or existing – to address these threats merely need to aim at full conservation of the corresponding natural, semi-natural and agricultural status of these systems.

Both peatlands and extensive grassland and wood-pasture systems store **large amounts of soil carbon**. The protection of these soils and carbon stocks is vital in view of climate change mitigation, conservation of their rich biodiversity (in soil and aboveground), regulation of the water cycle, and (in the case of grasslands and wood-pastures) protection from water erosion. These systems need full protection at the whole-system level. **Their loss cannot be reversed** on the human time scale. In the case of wood-pastures, current legislation (EU level instruments and their local implementation) appears inadequate to offer protection to such cross-sectoral (agro-forestry) systems.

Growing global food demand in coming decades already calls for intensification of all suitable land in current use, and **loss of productive land by sealing is therefore simply unaffordable**. Such loss increases the need to further intensify production on remaining farmland, increasing pressure on extensive systems with high biodiversity, affecting soil services in both types of systems. Ironically, the best soils are often the first to disappear under urbanisation, as soils near original habitation have been historically better amended. As for the intention of 'no-net land take' by 2050 (COM 571/2011) – by the way, not legally binding – the projected time span seems too comfortably long, given the expected steep rise in global population and the associated increasing demands for food and biomass during this period. This is particularly worrying since other drivers such as **climate change will likely deteriorate conditions** for crop production in large parts of the EU and elsewhere.

Other major threats are:

- Soil **erosion** by water; and
- **compaction** of the subsoil by heavy machinery.

These, as opposed to the above threats, require policy measures that enforce the **focussed adaptation of farming practices**.

Water erosion occurs in many regions of the EU and the remedies must be chosen to match the local conditions. The formal delimitation of **vulnerable zones**, and the keeping of effective soil cover types, and/or tillage systems tailored to the local potential and needs of such zones should be considered, in our view, as necessary steps towards the **enforcement of practices** that reduce the loss of topsoil by erosion.

We regard subsoil **compaction as a serious and wide-spread menace**, and include it in our shortlist of major threats that call for policy action. Its main driver is the ongoing upscaling of farm mechanisation to increase labour productivity. This trend is likely to cause increasing damage to the subsoil and will threaten crop production under climate change. It would be a sensible precaution - comparable to speed limits on highways - to **introduce a maximum permissible limit to the wheel load carrying capacity** (WLCC; Schjønning et al., 2015) for traffic on all agricultural soils. This would trigger innovation towards lighter and smaller systems, autonomous transport systems for harvesting operations, and similar advances. (Because of its impact on farm economy – level playing field - such measure would need EU-level underpinning.)

Soil in the farm economy

Although the soil is the farmer's most precious economic resource, there may be several reasons **why farmers might not take specific actions to protect or improve their soil** in their own commercial interest. These reasons include:

- Soils generally **respond slowly** to changes in management practices. Even given sufficient time, soil properties can only be improved within certain limits set by climate and parent material.
- **Effects** of soil improvements on crop **yields remain** often **unnoticed**. Inter-annual (weather-related) variation in attainable yield, timing of operations and the use of inputs, and crop genotype all have relatively strong effects on actual yield and quality of the produce, and can easily mask benefits from improved soil conditions, even in the long run. The low responsiveness of yield to management-induced soil improvements is, we believe, a major reason for farmers to avoid practices that may improve/protect one or more aspects of the soil, but bring disadvantages relative to current management.
- Benefits or problems may show **only** under **extreme conditions** (e.g., drought, waterlogging) or go unnoticed as long as critical thresholds are not crossed (disease suppressiveness).
- Farmers are **unaware of existing solutions** to address the problem; or no ready-for-practice solutions really exist.
- Farmers can sometimes **correct** for soil-induced crop stresses, by **using extra inputs** (fertilisers; crop protection agents; irrigation) or alternative equipment (deep-plow; drainage) – thus largely **externalising** the cost of such 'repair measures' if their impacts (emissions; lowering groundwater table; loss of biodiversity) are mostly felt elsewhere.

- Threats are simply not present on the farm.

Public and European Union interests in soils

Various aspects justify the tackling of soil threats at the **European Union** rather than at the national level:

- Biophysical impacts of threats do not stop at member states' borders.
- Economic impacts of threats, or consequences of the policy measures to counter them, are not limited to the member state either ('spill-over').
- The European Union level allows the **spatial optimisation** of interventions and the **sharing** of their **costs**.
- The **cost efficiency** of the necessary research and innovation efforts to make soil management more sustainable increases with the geographic extent of the application domain.
- Local democracy may hamper the implementation of unpopular measures, if these lack **central underpinning** at higher level.

Biophysical principles in soil management

Soil management that aims to protect soils and promote their services can be usefully guided by the following **general principles**:

- Soil **cover protects** the soil.
- Fresh organic material, irrespective of its source, is the main **food for soil life**.
- Soil **disturbance** (tillage) has a **negative impact** on soil macro-fauna such as earthworms.
- Heavy **traffic load** and high passing frequency of farm machinery causes **damage** to soil structure.
- High **frequency of host crop** species promotes the build-up of **soil borne pathogen** populations which may cause crop damage.

Practices that aim to protect soils and promote their services are often referred to as '**soil improving practices**'. This qualification does not express a general truth: '**win-wins**' are **rare** and all such practices come with **trade-offs**: negative consequences (sometimes grave) for either soil condition itself or for soil-supported services, with impacts on crop yield, farm income, climate, air and water quality, or biodiversity. Soil management must therefore be **optimised within the wider frame of sustainability goals**, not just focussing on the preservation of the soil itself.

Practices denominated as 'soil improving practices' include crop rotation (as opposed to monoculture); forms of **reduced tillage** (versus mouldboard ploughing); the use of **organic manures** (versus mineral fertilisers); the retention of **crop residues** (versus removal); and the use of **catch** and **cover crops** and **green manures** (versus bare fallow in between main crop seasons). More radical deviations from conventional practices are, for example, the use of **ley phases** (one or more years) in arable rotations, and the use of **guided traffic** ('tram-lines') for wheels to pass always over established tracks. **Organic farming** and **conservation agriculture** strive at the 'whole-farming-system level' to protect soils and promote their services. As stated, all of these come with drawbacks that cannot generally be ignored.

Although many soil threats are expressions of a development that we tend to characterize as '**intensification**', its opposite '**extensification**' - including approaches known as organic agriculture - is an **unfocussed and likely inadequate** answer to counter these threats. Instead, **specific** and probably pedoclimatically **tuned measures are needed** with respect to organic matter management, machinery weight specifications, hydrological management, the use of agro-chemicals, and landscaping are needed.

Soil carbon and climate

Peatlands are the most efficient carbon stores of all terrestrial ecosystems, containing 455,000 Mt of carbon globally, or twice the amount found in the world's forest biomass. The majority of this carbon is stored in saturated peat soil. Pristine peatlands are still sequestering carbon at a rate of 96 Mt carbon per year. EU soils store more than 70,000 Mt of organic carbon, as compared to about 2,000 Mt of carbon altogether emitted by member states annually. **Releasing** just a small fraction of the **soil carbon** to the atmosphere could **wipe out emission savings** in other sectors of the economy.

Practices to promote the accumulation of SOM in agricultural soils, or to mitigate its decline, can definitely bring agronomic benefits and contribute to the protection of soils and various soil services. However, the **scope for accumulating SOC (a major constituent of SOM) as a climate mitigation measure is very limited**, given the following considerations:

- the **availability** of additional **carbon sources** is limited;
- **nitrous oxide** (N₂O) emissions are associated with most practices that enhance SOC accumulation;
- **farm economy**/market demand limit the cultivation of crops with high SOC contribution;
- **only** carbon that originates from **extra primary production** and carbon that is **saved from incineration** of residues contributes to climate change mitigation;
- the use of organic residues for **bioenergy to replace fossil fuel** is likely to contribute substantially more to climate mitigation than soil incorporation of the biomass;
- gains in SOC made by adjusting farming practices are reversible – i.e., gained SOC can be **lost rapidly** if the practice is **discontinued** – while fossil fuel savings forgone and N₂O emissions are irreversible.

Monitoring and enforcement

Soil differs markedly from other compartments of the biosphere (atmosphere, water bodies) in that it is hardly subject to mixing, while its key processes and governing properties are highly heterogeneous in space due to variation in parent material, exposure and topography, climate and management history. **Heterogeneity** and absence of mixing imply that the monitoring of soil condition requires high sampling density, is usually expensive, time consuming or difficult, and is prone to large sampling and measurement error. Interpretation of observed indicator values is complex and needs local calibration. For certain properties, the use of geophysical methods may resolve such constraints.

We recommend that soil protection **policies** allow for **local ranking of priorities**: soil services in need of promotion, and/or threats in need of mitigation. Subsequently, a **two-pronged approach** can combine (i) the promotion/enforcement at **farm level** of locally suitable practices, with (ii) monitoring and evaluation at the **landscape and catchment**

scale of services delivered and/or threats mitigated. The two-pronged approach requires a priori assessment of **practices most likely to deliver the desired outcomes**, given the local conditions and the services in focus, as well as **identification of solutions** to overcome local barriers that hamper adoption of the practices.

This approach accounts implicitly for **soil-mediated services** as well as **services/threats directly affected** by the management practice itself, i.e., without involving a change in soil condition or soil processes. Although (some) policy measures are already in place – both at EU and national level – to protect non-soil compartments (water, air, habitats), we argue that so many trade-offs, synergies and other links exist between the goals of those ‘other environmental policies’ and those of soil protection, that policies can be more effective and coherent if they **target soil protection as part of a well-balanced set of locally prioritised environmental stakes**. Whereas the goals would be pursued at the catchment and landscape scales, the compliance with policy measures should be pursued at farm level.

Soil protection policies would be most usefully implemented if combined with **collective action**, and extension programmes with stepped targets and communication on progress achieved. **Compliance** of farmers with proposed measures can be verified by remote sensing or other survey methods, which is **generally less complex than the monitoring and interpretation of soil status indicators directly**. The formulation of **integrated soil management plans** at farm level should be a **mandatory tool** in this approach. It should be geared towards protecting the soil as well as the services the soil supports. Implementing such integrated soil management plan requires guidance by **independent experts**. The planning of measures, and regular evaluation of progress based on monitoring as well as barriers encountered should be part of the **permanent cycle** that would constitute integrated soil management planning.

Examples of services (or threats mitigated) that can be monitored at **catchment/landscape** scale are the regulation of hydrology (response time in water channels), mitigation of erosion (sediment load in channels), provision of clean water (surface water pollutant load; nutrients; biocides), and aboveground biodiversity. Some services, in contrast, require **in-situ (per field) measurement** for their assessment. These include soil biodiversity, botanical biodiversity, and C-sequestration. However, we regard it unlikely that it is feasible to monitor these for enforcement or verification purposes. Finally, for threats which cannot be countered by the individual farmer due to the scale of required measures (e.g. salinization, desertification, peatland degradation, landslides) it is self-evident that the catchment or landscape scale is the only level permitting effective action.

Innovation, communication, learning and extension

Changes in soil management come with **barriers** that complicate proper **integration into the farming system**. It appears that combining technological innovation with exchange of experiences and insights by practitioners will be vital to promote practices that bring extra cost or other difficulties. “**Sustainable intensification means more knowledge input and more ‘services output’ per ha**” (Buckwell et al., 2014). This definitely applies also to the subdomain of soil management.

Some basic **biophysical principles** cannot be changed and the barriers defined by them are, in the long run, perhaps the most challenging. Addressing these may require drastic changes in production systems. Other barriers can be tackled by **focused research and innovation** efforts. Developments in aerial surveillance, computer vision and crop sensing,

autonomous transport, GPS guidance, crop breeding and genome editing, and many other fields of technology may bring solutions to long-felt bottlenecks. Turning technological development towards resolving conflicts and trade-offs in soil management is, in our view, the most promising track to progress. Improved methods for seeding, weed and pest control, harvesting and transport operations, and post-harvest processing of produce can all directly or indirectly contribute to more sustainable soil management. And so can targeted breeding of crops, cover crops and green manures.

Communication tools and **access** to existing **knowledge** for innovation are rapidly improving with the help of web semantics and linked open data infrastructure, decision support via hand-held devices, etc. All of these serve to support learning networks and advisory systems. For an overview of the roles ICT can play in agriculture and participatory innovation, we refer to EU SCAR (2015). We can only confirm that current R&D policy under **Horizon-2020** and the **EIP-AGRI** - with its focus groups setting agenda's, thematic networks surveying bottlenecks and possible solutions, and operational groups connecting the various actor-types per supply chain around shared problems - has set out an ambitious programme deserving of continued support.

Innovation towards more sustainable management can be **promoted by legislation** that sets requirements and so enforces the search for solutions. For such twinned approach (support innovation; legal enforcement) to result in desired outcomes, it must recognise the large **diversity of bottlenecks** felt by farmers, set by local biophysical and economic constraints. Also, it must likely be accompanied by **independent advice**. Parts of Europe have seen the collapse of such independent advisory systems over the past decades. This hampers the infusion of knowledge for improving ecosystem services, where focus now is largely restricted to commercial interests. Such **commercially driven advisory systems are not likely** to foster the innovation needed to drive the needed long-term local innovations in sustainable soil management.

Policy instruments and initiatives in EU and Member states

The **Thematic Strategy for Soil Protection** (COM(2006) 231) has been guiding soil protection efforts in the EU since its adoption in 2006. Its objective is to foster soil protection while encouraging sustainable use of soil, based on the prevention of further degradation, the preservation of soil functions, and the restoration of degraded soils. Also in 2006, the Commission submitted a proposal for a **Directive** establishing a framework for the protection of soil and amending Directive 2004/35/EC (COM(2006)232). It would have obliged member states to ensure that land users must take precautions to prevent or minimize adverse effects of their actions on soil functions. The proposal, however, was withdrawn by the Commission in May 2014, after being blocked by various States in the Council.

Various more recent initiatives, each in its own right, offer some opportunity for soil protection:

- The **Roadmap** to a Resource Efficient Europe established the non-binding goal of reducing the amount of land take to 'no-net land take' by 2050 (COM 571/2011).
- The EU Seventh **Environment Action Programme** (7th EAP; in force from 17 January 2014) recognizes that soil degradation is a serious challenge and that by 2020 land must be managed sustainably in the EU, and soil must be adequately protected and the remediation of contaminated sites well underway. Issues specifically mentioned are soil erosion, maintenance of soil organic matter, and remediation of contaminated sites. It also calls for revisiting 'a binding legal framework' as soon as possible.

- The EU and its member states have signed to global **Sustainable Development Goals** (SDGs), adopted in September 2015 by the UN Sustainable Development Summit. Among the ambitions of the SDGs is to achieve a more sustainable agriculture, in which soils play a central role; and soils are mentioned under many of the other SDGs as well.
- Another initiative targeting the protection of soils is the current “**People4Soil** European Citizens' Initiative”. This initiative invites the Commission to "recognise soil as a shared heritage that needs EU level protection and develop a dedicated legally binding framework covering the main soil threats”. The initiative is backed by more than 400 associations across Europe, and aims to collect the required 1 million signatures by 12 September 2017.

By itself, and potentially in interaction with other initiatives, the EU’s **Common Agricultural Policy** (CAP) is a key EU instrument to encourage the sustainable management of resources including soils, and the delivery of public goods related to the environment and climate. Past reforms of the CAP have sought to enhance the delivery of environmental benefits by various ways: 1) **Cross-compliance** links CAP payments to a set of Statutory Management Requirements (SMRs) based on EU legislation and of several standards for the Good Agricultural and Environmental Condition of land (GAEC). There are in particular GAEC concerns related to maintain soil carbon stocks and minimizing erosion; 2) A **greening** component that emphasizes cropping system diversity; and 3) A **rural development pillar** that co-finances a number of measures that can have an impact on soils, e.g. productive and non-productive investments, agri-environment-climate measures, organic farming, and areas facing natural and other specific constraints. (It is up to Member States to implement or not measures under this pillar; there is a European menu to choose measures from.)

The **instruments and measures** for **soil protection** currently implemented at EU and MS level, including those under the two CAP pillars, were subject to a **recent inventory** carried out in the frame of the Thematic Strategy for Soil Protection (assigned by the Commission, ENV. B.1/SER/2015/0022). The inventory (Frelih-Larsen et al., 2016) gives an updated and detailed picture of **policies at various levels** (national legislation, voluntary schemes, and implementation of EU policies). It covers measures in the frame of the above EU level initiatives, as well as national initiatives. Some of their **key findings** are summarized in Annex III. These include the following (shortened citations from Frelih-Larsen et al., 2016):

- **Many** different policy **instruments** at EU and Member State level exist that either explicitly reference soil threats or soil functions, or implicitly offer some form of protection for soils. 35 EU level and 671 Member State policy instruments were identified.
- When looking at the weaknesses of EU level policy instruments in protecting Europe’s soils, the **lack of a coherent, strategic policy framework** was highlighted across all policy clusters.
- The analysis of nationally initiated policy instruments (national initiatives) in the EU-28 Member States confirms that the **lack of strategic coordination** is an important theme.

1. INTRODUCTION

Soils constitute the skin of the terrestrial earth, and they are the place where many of the interactions between the atmosphere, hydrosphere, geosphere and biosphere happen. As such, land and soil are key components of the EU's natural resources that contribute to numerous goods and services (Schwilch et al., 2016). Soil, as defined in the EU Thematic Strategy for Soil Protection (COM(2006) 231), is 'the top layer of the earth's crust, formed by mineral particles, organic matter, water, air and living organisms. It is the interface between earth, air and water and hosts most of the biosphere'. Soil is differentiated into horizons of variable depth, which differ from the material below in morphology, physical make-up, chemical properties and composition, and biological characteristics. The most fertile portion of soil, generally lying between 0-30 cm belowground, is called 'topsoil'. Soil fertility results from soil characteristics (e.g. texture), nutrient inputs and other management practices (Bronick and Lal, 2005).

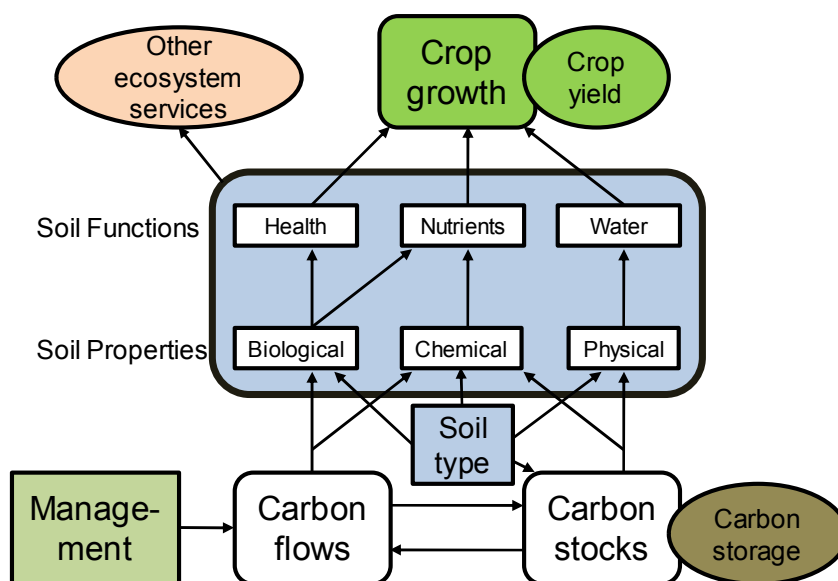
Soils are thus key to the delivery of a wide array of ecosystem services, including water and nutrient cycle regulation, food and fiber production, providing a physical basis for construction and habitat for various species (Schwilch et al., 2016). However, soils in the EU are exposed to numerous threats, which limit their ability to function and to deliver ecosystem services. These threats include erosion, floods and landslides, loss of soil organic matter, salinisation, contamination, compaction, sealing, and loss of soil biodiversity. A recent report on the state of the European Environment established that loss of soil functions and land degradation remain major concerns, and that these will likely show continued deteriorating trends in the future (EEA, 2016).

The scale of soil degradation in the EU is significant with approximately 22% of European land affected by water and wind erosion (Jones et al., 2012). Around 45% of the mineral soils in Europe have low or very low organic carbon content, soil contamination is affecting up to three million sites, and an estimated 32-36% of European subsoils are having high or very high susceptibility to compaction (Jones et al. 2012). An increase in soil sealing has also been identified due to construction and infrastructure development (EEA 2016). These soil threats moreover drive the loss of soil biodiversity (Smith et al., 2015).

Several underlying causes of soil degradation have been identified and linked to global and regional developments and trends (EEA, 2016). These drivers relate to increasing urbanisation, land abandonment, and intensification of agricultural production. Soil degradation reduces or eliminates soils' functions and their ability to support ecosystem services. It is therefore essential to maintain the services provided by soils by restoring soil functions. As such, soils should be sustainably managed, and degraded soils should be rehabilitated or restored. In particular, a balance should be achieved between food production and soil preservation/protection, and good soil governance is needed, at all levels (Frelih-Larsen et al., 2016).

There is a growing global awareness that soil functions and related services are under pressure. The functionality of soils can be judged and monitored by either measuring the pressures that are underlying the threats mentioned above (e.g. use of heavy machinery either or not in combination with more extreme weather), the state in which the threat finds itself (e.g. extent of compaction), or the impact (e.g. lower primary production, less water regulation, less biodiversity). These links between soil management, soil function and soil services are complex and often highly locally specific (as illustrated – with focus on just soil

Figure 2. Links between soil management of carbon flows and related soil properties, soil processes and soil functions and how this influences crop production as well as other ecosystem services



Sources: SmartSOIL project.

carbon management - in Fig. 2). Given the importance of management for soil threats and soil functions, much of the governance centers on how to achieve improved management (Freluh-Larsen et al., 2016).

A brief EU soils policies history is cited from Stolte et al. (2016): "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 (Stolte et al., 2015). It would have obliged member states to ensure that any land user whose actions affect the soil in a way that would be expected to hamper significantly the soil functions set out in the Directive, 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. The EU Seventh Environment Action Programme (7th EAP) entered into force on 17 January 2014 and might in some ways compensate for the blocking of the Soil Framework Directive. [...] The 7th EAP recognizes that soil degradation is a serious challenge. It provides that by 2020 land is managed sustainably in the EU, 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 and increase soil organic matter and to remediate contaminated sites." The 7th EAP also proposes that addressing issues facing EU soil resources within 'a binding legal framework' should be revisited as soon as possible. Furthermore, the Roadmap to a Resource Efficient Europe also established the goal (non-binding) of reducing the amount of land take to 'no-net land take' by 2050 (COM 571/2011).

The EU and its member states have signed to global Sustainable Development Goals (SDGs), which were adopted in September 2015 by the UN Sustainable Development Summit. This SDG agenda is universal and creates responsibilities for all countries, and as

part of this comes also the ambition to achieve a more sustainable agriculture to which soils play a central role, and soils are mentioned under many of the other SDGs as well.

The EU's Common Agricultural Policy (CAP) is potentially a key EU instrument to encourage sustainable resource management and the delivery of public goods related to the environment and climate. Past reforms of the CAP have sought to enhance the delivery of environmental benefits. This is implemented in various ways: 1) Cross-compliance links CAP payments to a set of Statutory Management Requirements (SMRs) based on EU legislation and of several standards for the Good Agricultural and Environmental Condition of land (GAEC). There are in particular GAEC concerns related to maintain soil carbon stocks and minimizing erosion; 2) A greening component that emphasizes cropping system diversity; and 3) A rural development pillar that co-finances a number of measures that can have an impact on soils, e.g. productive and non-productive investments, agri-environment-climate measures, organic farming, and areas facing natural and other specific constraints.

There are also a number of other initiatives at EU or member state level that target the protection of soils. An example is the People4Soil European Citizens' Initiative². This initiative invites the Commission to "recognise soil as a shared heritage that needs EU level protection and develop a dedicated legally binding framework covering the main soil threats". The initiative is backed by more than 400 associations across Europe.

This report provides an overview of soil threats in Europe, how these can be managed, and which policy incentives might be implemented to sustainably manage soils to minimize impacts of soil threats. Chapter 2 provides an introduction to the concept of soil quality and of the functions and services that soils perform. In Chapter 3 we give an overview of soils in Europe with particular emphasis on their suitability for various soil functions and services, and their vulnerability to threats. The threats to soils are discussed in Chapter 4, with a detailed description of each threat and its relevance in a European context. Chapter 5 presents a range of management practices that can mitigate one or more of these threats. Both individual and combined management practices are presented. Chapter 6 then presents and discusses how various policies can be put in place to support the protection of soils and their functions. Finally, conclusions and recommendations are given in Chapter 7.

² A European citizens' initiative is an **invitation** to the **European Commission to propose legislation** on matters where the EU has competence to legislate. A citizens' initiative has to be backed by at least one million EU citizens, coming from **at least 7 out of the 28 member states**. A **minimum number of signatories** is required in each of those 7 member states. This deadline for this ECI to collect 1 million signatures is 12 September 2017.

2. SOIL SERVICES AND SOIL QUALITY

2.1. Services and functions

Worldwide there is an increasing awareness that ecosystems, including soils, provide a wide suite of services to mankind (CICES, 2013; Keesstra et al., 2016; www.teebweb.org). As far as soils under agriculture are concerned, the five main ecosystem services include the provision of harvestable crops, clean fresh water and nutrients for plants and animals, conservation of suitable habitats for biodiversity and maintenance of a benign climate (Schulte et al., 2014). The corresponding soil functions are primary production, water regulation, nutrient cycling, habitat support and climate regulation. Primary production can be defined as the capacity of a soil to produce plant biomass for human use, providing food, feed, fibre and biofuel (Tilman et al., 2002). Water regulation pertains to the capacity of a soil to remove harmful compounds and to receive, store and conduct water for either use by agricultural crops or by use in other ecosystems or human activities whilst reducing the risks of droughts, floods and erosion (Coates et al., 2013). Nutrient cycling can be defined as the capacity of a soil to receive nutrients in the form of farm, industrial or urban 'wastes', to provide nutrients from intrinsic resources and to effectively carry over these nutrients into harvested crops (Schröder et al., 2016). Habitat support can be defined as the capacity of a soil to sustain conditions that are favourable to soil organisms and the aboveground organisms directly depending on them (Turbé et al., 2010). Climate regulation includes the capacity of a soil to minimize emissions of greenhouse gases or to reduce their negative impact, among which its capacity to store carbon in non-labile forms (Lal, 2004). Fig. 3 summarizes the relations between these various concepts.

The extent to which these functions are delivered in practice depends on the interactions between climatic conditions (temperature, rainfall, radiation), intrinsic 'fixed' soil properties (e.g. texture, parent material, depth to bedrock or – in some locations - groundwater table) and dynamic soil properties affected by soil management (e.g. bulk density, pH, organic matter, biological activity). This implies that the same level of service provision, if attainable at all, is likely to require substantial differences in management under different pedo-climatic conditions. Another consequence of the interplay of factors is that some environments are probably better suited to perform certain functions and deliver specific services than others, regardless of management efforts. From that point of view, optimized soil management results in mosaics rather than in a uniform appearance of all functions everywhere, the scale of the eventual mosaic probably having an arbitrary character. It is only fair to say that management decisions may mutually reinforce functions, but can at the same time favour one or more functions at the expense of one or more other functions (Power, 2010). Consequently, there is no such thing as a one size fits all soil strategy. Decisions must therefore be based on careful considerations taking due account of local demands and potentials for services, as well as synergies and trade-offs and weighting of alternative options for achieving these services.

2.2. Synergies and trade-offs

Minimum tillage, for instance, can protect soil life from surface-living predators, increase soil carbon (C) in surface layers, increase the water infiltration capacity of a soil (Rasmussen, 1999; Hobbs et al., 2008; Alvarez and Steinbach, 2009) and thus promote the decontamination of substances in that water by extending the residence time (Rivett et al., 2008). A better infiltration can be advantageous in arid regions where the availability of water is limiting crop yields (Holland, 2004). In moister regions, however, negative effects

of minimum tillage on crop yield will generally outweigh the benefits (Van den Putte et al., 2010; Hansen et al., 2010; Newton et al., 2012; Arvidsson et al., 2014; Brennan et al., 2014) and will thus in some cases even negatively affect primary production and nutrient cycling. Minimum tillage may save diesel fuel and thus reduce carbon dioxide emissions but can at the same time under wet conditions increase the emission of nitrous oxide from soils (Palma et al., 1997; Lehtinen et al., 2014) and of ammonia from applied manures (Huijsmans et al., 2015). Minimum tillage can also augment the need for chemical weed control which may have detrimental effects on water quality (Chauhan et al., 2012). Similarly, artificial drainage involves dilemmas. Drainage generally extends the length of the growing season in wet environments which helps to increase crop yields (Schulte et al., 2012). However, drainage also stimulates the oxidation of organic C and reduces the denitrification capacity of soils (Heinen, 2006). Less denitrification is positive from a nitrogen cycling perspective but may at the same time increase the risk of nitrate leaching (Coyle et al., 2016).

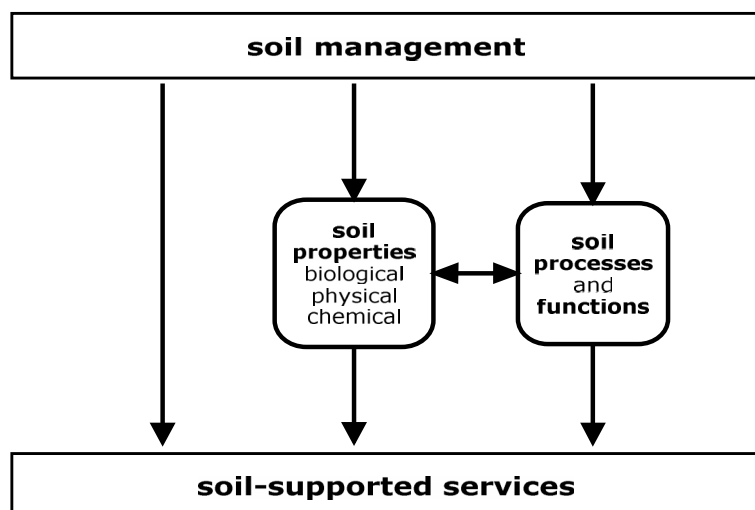
Straw management represents another example of competing interests. Leaving straw in a field, either directly after harvest or indirectly in the form of the bedding component of solid manures, provides food for soil life and can contribute to the long term productivity of a soil via improvement of the physical soil fertility (Hudson, 1994; Fraser and Piercy, 1998). Leaving straw may, however, initially cost yield due to hibernation of pests or diseases, or the fixation of nitrogen. Besides, the positive climate regulating effect of using straw as a substitute for fossil fuel outweighs the positive effect that retaining straw has, at least temporarily, in terms of C sequestration (Powlson et al., 2011).

Cover cropping is another example of how management may simultaneously have positive and negative effects on soil functions. It helps to reduce nutrient losses to water bodies and hence contributes to nutrient cycling and enhancing soil C content. Green covers undoubtedly supports the conservation of habitats relative to bare soils but their successful establishment may depend on the harvest date of preceding main crops and thus require concessions to the length of the growing season of those main crops. This can carry a price in terms of primary production of crop species such as potatoes and maize which both happen to be in need of those N scavenging cover crops in view of their soil mineral N residues (Schröder et al., 1996a; Schröder et al., 1996b).

The conversion of grassland into arable land represents a case showing how functions may compete. This conversion can be a sensible choice where the focus of primary production is to carbohydrates instead of proteins. There is convincing evidence, however, that land use change from grassland to arable land is associated with less biodiversity, greater water pollution risks, smaller recovery rates of applied nutrients, and a temporary increase of greenhouse gas emissions.

The application of industrial residues complements the above list of dilemmas. Acceptance of industrial residues (e.g. biosolids) as soil amendments undoubtedly contributes to the function of nutrient cycling by sustaining the upstream movement of nutrients (to farms, industries, households) via harvested crops, but it may simultaneously jeopardize the biodiversity of soils and deteriorate the quality of nearby water bodies due to the contaminants that these residues tend to contain (McGrath et al., 1994; Erhardt and Pruess, 2001; Motoyama et al., 2011; Peyton et al., 2016). Moreover, the application of residues (manures, sludges, compost) is often carried out with heavy equipment because of the bulky nature of those products. This can have a detrimental effect on the soil structure (Batey, 2009) with negative effects on the utilization of nitrogen (Douglas and Crawford, 1998) and phosphorus (Johnston and Dawson, 2010).

Figure 3. Schematic representation of relations between soil management, soil properties, processes and functions, and soil-supported services. Soil threats aggravated by management are negative impacts of practices on soil properties, processes or functions, and that translate into diminished services; soil management can also directly affect the services normally supported by soils, without impacting the soil (properties, processes, functions) itself. Each arrow can represent threats as well as benefits



2.3. Soil quality

The above examples show that what is good or bad for one function is not always necessarily good or bad for another function, at least not in all environments. For similar reasons it is questionable to endeavour a concrete, ubiquitously applicable definition of soil quality, expressed in terms of intrinsic or dynamic soil attributes. After all, attributes such as texture, organic matter, pH, air filled porosity, rooting depth, infiltration capacity, and biological activity, including mineralization and respiration, have little meaning when looked upon in isolation, disconnected from environmental aspects such as the amount and distribution of precipitation, ambient temperatures, and the kind of management to which the soils in question are exposed (Letey et al., 2003; Sojka et al., 2003; Schröder et al., 2016). This nuance should not be interpreted as if these attributes have no relevance at all, if alone because soils need to be resistant and resilient in view of the anticipated more extreme weather events (Olesen et al., 2011). What needs to be avoided, however, is to think that 'more' (e.g. organic matter) is always and everywhere 'better' for all functions at the same time (Loveland and Webb, 2003). This is also true for biodiversity (*sensu* diversity, abundance, activity), regardless of the fact that soil quality, soil health and soil life are often presented as a trinity (Doran and Zeiss, 2000; Brussaard et al., 2007; Kibblewhite et al., 2008). Mineralization of soil organic matter is, for instance, undoubtedly associated with soil life (Coleman, 2008), but soil biodiversity is generally not a reflection of the ability of agricultural soils to convert organically bound nutrients into plant-available nutrients, as opposed to what has been suggested by Sandhu et al. (2015). Mineralization rates appear to be most and for all a reflection of the amount and type of organic substrate that has been put into those soils in the first place (Langmeier et al., 2002; Boschard et al., 2009). Of course, mineralization is hampered when specific groups or organisms are withheld or fully destroyed (Griffiths et al., 2000; Wagg et al., 2014; Rashid et al., 2013). However, that does not imply that soil life must be spared at all costs, bearing in mind that the conservation of soil biodiversity carries a price in terms of other functions than habitat

support, as explained in earlier examples. A critical appraisal of the functionality or redundancy of biodiversity within agricultural systems is therefore imperative (Walker, 1992; Giller et al., 1997; Wellnitz and Poff, 2001; Setälä et al., 2005).

3. AGRICULTURAL SOILS OF EUROPE AND THEIR DISTRIBUTION

3.1. Soil types in EU agriculture

3.1.1. Soil Typology and pedogenetic processes

The Europe-wide spatial inventory and harmonisation of soil types - the Soil Geographical Database of Eurasia (SGDBE) - was compiled by the Scientific Committee of the European Soil Bureau (Toth et al., 2008). The naming of Soil Typological Units follows the classifications used in the World Reference Base for Soil Resources (WRB, FAO 1998) and in the FAO 1990 Soil Legend (FAO, 1990), while refinements and adaptations to specificities in Europe were included. The FAO legend itself is based on major pedogenetic processes that lead to soil differentiation, and which are dictated largely by climate and parent material. Examples of such processes are the accumulation of organic material (due to cold and wet climate retarding decomposition), podsolisation (leaching and precipitation of Fe and Al compounds and humic substances in wet climates, typically on quartz-rich acidic materials) and 'lessivage' (downward displacement and accumulation of clay particles in medium textured soils). The above version of the WRB recognises 30 Reference Soil Groups (RSGs), 23 of which are relevant to the EU. The distribution of soils in the EU at RSG level is given in Fig. 4, and their extent in Table 1. (Note: in the WRB revision of 2015, stagnosols are recognised as an additional RSG on its own; these are soils with strong mottling due to redox processes caused by stagnating surface water. No EU soil map based on WRB2015 has been made available, so far. We therefore follow Toth et al. (2008).)

Pedogenetic classification is relevant to the understanding of the spatial distribution of soils as determined by geology (parent material) and climate. To underline similarities between soils arising from such determinants or their associated processes, RSGs have been grouped into so-called sets. The first grouping (WRB 1998 (FAO, 2001)) into 10 sets was mainly based on similarities in parent material, topography, or climate. A revision (WRB 2006), which again distinguished 10 sets, rather emphasized specific soil formation processes or governing factors. Several of these reflect the combined influence of climate and parent rock. Examples are water (wetting/drying; hydrology; stagnation), salt accumulation, Fe/Al chemistry (displacement; concentration by weathering), and the accumulation of organic matter. Besides such natural processes, both groupings (1998; 2006) recognize 'limited age' (soils that do not or only weakly express effects of forming factors) and 'human influence' as keys to corresponding sets. For reasons explained by Toth et al. (2008), those authors followed the 1998 edition rather than the later update. The largest set, in spatial extent, is that of mineral soils conditioned by the climate of the humid and subhumid temperate regions (30.7% of land area in the EU). These soils generally show the signature of downward water movement associated with precipitation surplus. (Apart from the above sets, a more general distinction is made between zonal soils (defined by the prolonged typical influence of a given climate); intrazonal soils (zonal soils modified by unusual dominance of a non-climate factor), and azonal soils (immature)).

While primarily relevant for understanding the spatial distribution of soils, RSG names may also convey clues on the soil's suitability for agriculture; or about threats, with or without agriculture. Histosols (peat soils), for example, are organic soils too wet for agriculture in natural (undrained) state. Solonchaks and Solonchaks are too saline for productive agriculture, Cryosols too cold, and Leptosols too shallow (near-surface bedrock). Podzols

are poor and acidic in nature, and are productive only if well amended. In contrast, Anthrosols (enriched through past human effort) and Chernozems ('black soils') are highly fertile and versatile soils. In all these examples the diagnostic factor (the key to assigning a soil to a certain RSG) itself defines the soil's (un)suitability for agriculture, but the information conveyed in soil names at RSG level is generally limited. Moreover, for several RSGs the main diagnostic factor does not express the soil's suitability for specific uses or services at all. The WRB taxonomy includes a second (below RSG) level, which expresses a soil's features in more detail by so-called qualifiers. Some qualifiers express topsoil or subsoil characteristics of direct relevance to the soil's use potential. Examples are Arenic (sand texture); Dystric (low base saturation, i.e. low natural fertility); Histic, Mollic and Plaggic (humous topsoil); Gleyic (shallow groundwater table); and Leptic and Lithic (bedrock near surface).

3.1.2. Expression of spatial heterogeneity

Toth et al. (2008) explain that the Soil Geographical Database of Eurasia (SGDBE) is part of the European Soil Information System (Liedekerke et al., 2004; Panagos 2006) and results from collaboration between soil survey institutions and soil specialists in Europe and neighbouring countries. The SGDBE represents harmonized information on EU soils at the scale of 1:1,000,000 expressed on maps with a total of 24,000 polygons to cover the EU. With a total EU land area of about 4,15 million km², the average polygon size is about 170 km². A polygon is the smallest spatial unit (geometrically defined area) presented as homogeneous unit on a map. Higher resolution is usually required to express true spatial heterogeneity of soil types, that is, to delineate surface areas occupied by a single soil type (Soil Typological Unit, STU). For this reason - and because STUs may occur in specific associations - soil mapping units (SMUs) usually refer to associations of several STUs. The relative importance of STUs in each SMU is defined in the map legend. Finally, it is relevant to note that the SGDBE contains more information on each STU than just the soil's name (composed of RSG and qualifiers). Such additional ('semantic') information refers to properties ('attributes') not conveyed by the taxonomy, and may be relevant to the soil's use potential and associated threats.

In short, each colour on the 1:1,000,000 soil map refers to a specific association (SMU) of soil types (STUs). Spatially unconnected areas of the same SMU are called polygons.

3.2. Brief description of Reference Soil Groups relevant to agriculture

Abstracting from Toth et al. (2008), we mention below those RSGs that each occupy at least 3% of the EU land area, with their typical characteristics relevant to agriculture and/or soil threats. These RSGs are the Arenosols, Cambisols, Fluvisols, Gleysols, Histosols, Leptosols, Luvisols, Podzols and Regosols.

Arenosols are coarse-textured (sandy) soils, either developed in deposited sands (windblown, glacial, fluvial, lacustrine or marine deposits), or in situ developed over old quartz-rich rock. They are easily erodible with their slow weathering rate and/or erosion of the surface, factors that limit their pedogenetic development. Water and nutrient holding capacities are low, as is base saturation (the proportion of the adsorption complex occupied by bases, i.e. non-H⁺ cations). High permeability and easy workability make these soils favourable for agriculture (notably for horticulture) if water and nutrient are supplemented. Arenosols are typically susceptible to wind erosion, low fertility, acidification, and the leaching of agrichemicals (nutrients, biocides) to ground and surface water.

Table 1. Spatial extent of Reference soil Groups (WRB 1998) in the European Union

Reference Soil	Km ²	%
Acrisols	10626	0.3
Albeluvisols	76865	1.9
Andosols	8705	0.2
Anthrosols	3428	0.1
Arenosols	149776	3.6
Calcisols	9288	0.2
Cambisols	110759	26.7
Chernozems	78492	1.9
Fluvisols	221669	5.4
Gleysols	219781	5.3
Gypsisols	4110	0.1
Histosols	268741	6.5
Kastanozems	3532	0.1
Leptosols	435713	10.5
Luvisols	610941	14.7
Phaeozems	70439	1.7
Planosols	18981	0.5
Podzols	566874	13.7
Regosols	222322	5.4
Solonchaks	11728	0.3
Solonetz	9857	0.2
Umbrisols	329	<0.1
Vertisols	36447	0.9
Total soil cover	414624	100

Sources: Adapted (decimals reduced) from Toth et al., 2008.

Cambisols are the largest single RSG (26.7% of EU land area) and constitute a set (WRB 1998) on their own. They are found in nearly all regions of the EU and occur under a wide variety of environmental conditions and vegetation types. Cambisols are mineral soils that

show limited development due to limited age, related to geomorphological processes – e.g. glaciation, erosion, solifluction and sedimentation – that reset the clock of soil formation.

Nevertheless, their age is still estimated in units of 10^4 to 10^6 years, time lapses required for pedogenesis to imprint some (diagnostic) level of alteration relative to the parent material (to which change the Latin root *cambi-* refers). The type of alteration is expressed at the second taxonomic level (qualifier) and may give clues to suitability for agriculture or susceptibility to threats (see above). These soils are important to EU agriculture and occupy wide stretches of the most fertile lands in the EU.

Fluvisols are young soils characterised by signs of (past or present) frequent flooding and deposition of fresh sediments, such as stratification of the parent material still unaffected by soil formation processes (e.g. bioturbation). They occur throughout the EU and occupy specific positions in the landscape, near the riverbed in upstream areas and on floodplains in the lower reaches of river systems. As with Cambisols, relevant properties are expressed at the qualifier level of the taxonomy. The level topography and often high fertility (depending on the texture of the sediments) of Fluvisols render these soils attractive for agriculture, if flooding risk is limited; they may require drainage, too.

Gleysols are characterised by prolonged saturation of (parts of) the profile with groundwater. They are abundant north of the Paris-Bucharest line, with large extents in the UK, Netherlands, Germany, Poland and the Baltics. These soils generally require drainage to make them suitable for agriculture, but can then be used for arable cropping, dairy farming or horticulture (Toth et al., 2008). (*Stagnosols* are a recently defined additional RSG. These soils bear the marks of redox processes under influence of stagnant surface water, the movement of which is impeded by more or less impervious strata. They are included under Gleysols in the map shown in Fig. 4. An EU map based on WRB 2015 is not yet available.)

Histosols or peat soils are composed mainly of organic material accumulated due to retarded decomposition of plant material. Cold and anoxia due to wetness are the main factors controlling the accumulation of peat material, and for this reason Histosols occur mostly in northern (boreal, sub-arctic, low arctic) regions, as well as in regions where such conditions prevailed during the Pleistocene (the Netherlands, Germany, Poland). Where the climate favours agriculture, these soils are often drained and then highly prone to mineralisation which converts, in the long run, the soil's entire carbon stock into CO₂. Depending on their origin, the material is poor in nutrients (as in soils derived from raised bogs) or richer, allowing the annual release of substantial amounts of nitrogen supporting crop production.

Leptosols are present throughout Europe in any mountainous region. They occur frequently in the Mediterranean region. They are shallow due to the presence of bedrock near the surface, and are mostly under forestry.

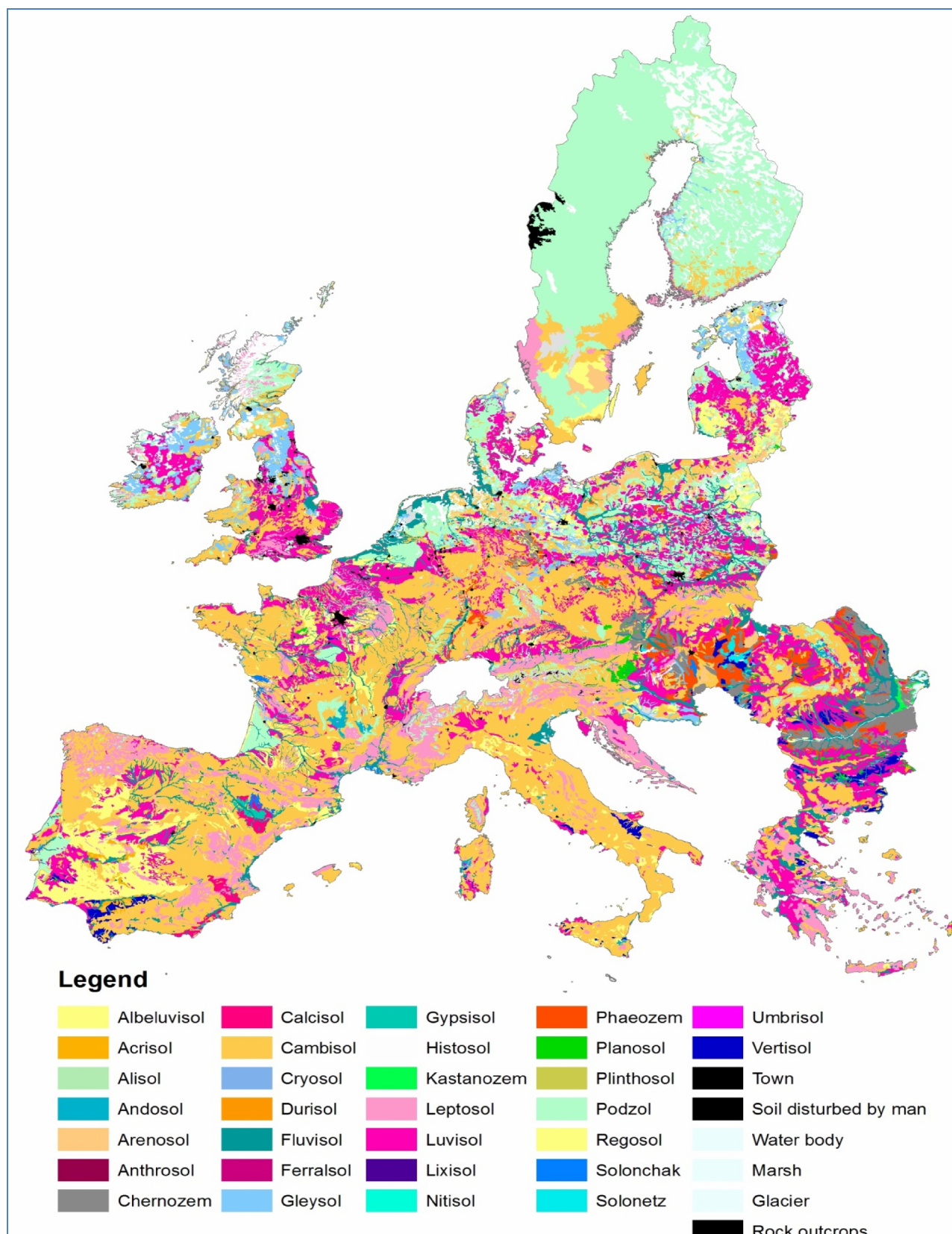
Luvissols are characterised by a textural contrast in the profile, due to accumulation of clay particles washed from shallower horizons (lessivage). After Cambisols, they represent the second largest RSG of the EU, and occur throughout the EU except in Scandinavia, Scotland and the Netherlands (with rare exceptions in the latter country). They can be very favourable soils for agriculture, for example when developed over loess deposits.

Podzols have the accumulation of organic-iron/aluminium complexes in the so-called spodic horizon. The washed-out material overlying this horizon is bleached and gives these soils their name (which refers to 'ash' in Russian). They are naturally poor. Zonal podzols

(i.e., as determined by climate) occur typically in the Boreal zone and do therefore not favour agriculture. Intrazonal podzols (determined by quartz-rich parent material) may occur in any climate with sufficient precipitation. These soils need improvement for agricultural use (deep ploughing to break the hardpan, fertilisers, and/or liming).

Regosols are a rest group of soils that do not meet taxonomic requirements for other RSGs. The expression of soil forming processes is mostly confined to a limited surface horizon. They are extensive in eroding lands. Agriculture can be limited by different factors (low temperature, prolonged dryness, topography (steep terrain), but some Regosols are suited to highly productive farming after amendment (e.g. drainage).

Figure 4. Distribution of soil types in the EU



Source: European Soil Data Centre (ESDAC), courtesy of Mr. Panos Panagos, May 2017.

4. SOIL THREATS AND ASSOCIATED PROCESSES

4.1. Brief overview of soil threats

We define soil threats as processes or agents that deteriorate (some of) the functions of soils and the services that soils provide, or that change the state of soils and – if prolonged – are expected to damage soil functions and services in the long run. While some of these processes (or pressures, drivers) occur naturally, emphasis in this report is on threats caused by human activity through agricultural soil management. Soil management (Chapter 5) constitutes the whole set of practices (including land use change itself, e.g. grassland to arable, extensive to intensive) that, purposely or not, affects the state of soils or their functioning or the services they provide. The latter extension implies that we include in our analysis practices that may have little or no impact on the soil (or even its functions), but that do affect wider goals (e.g. water and air quality; climate; aboveground biodiversity). The extension is relevant because care for soil (its quality, state, processes) may be at the expense of such other goals.

Major soil threats as recognised by respective comprehensive studies at European level are listed in Table 2. We added the loss of soil fertility (ability to provide nutrients to the crop), the loss of aboveground biodiversity, and the spreading of soil-borne diseases as explicit terms, though some studies may have included these in other threats. The table includes soil sealing, but this threat - though highly relevant and requiring firm action - is largely beyond agricultural soil management, and it is a matter of legislation and not science or management, and is therefore not extensively discussed.

The following sections aim to summarize for each threat its nature (processes, susceptibility of specific soil-climate combinations, aggravating factors), extent and actual or potential impacts on functions or on other threats. In these sections we gratefully sample from the excellent overview of soil threats by Stolte et al. (2016), compiled under the RECARE project (<http://www.recare-project.eu/>).

Some aspects of soil threats are highly relevant to the present study, yet are not covered here for the sake of brevity. Such aspects include comparison and analyses of existing frameworks for the classification of ecosystem services; monetary valuation of such services; and indicators, tools and methods for the assessment and monitoring of threats to the states, functions or services of soils.

In the following, we rank the soil threats in order of importance or urgency by our conception. Such ranking cannot be based on an absolute truth, as different threats are measured in different entities, to each of which weights may be assigned ultimately by society. The basis of our ranking is explained in Section 6.6.

4.2. Degradation of peat soils

4.2.1. Extent

Peatlands are the most efficient carbon stores of all terrestrial ecosystems, containing 455,000 Mt of carbon, or twice the amount found in the world's forest biomass. The majority of this carbon is stored in the saturated peat soil. Pristine peatlands are still sequestering carbon at a rate of 96 Mt carbon per year (Dunn and Freeman, 2014).

Table 2. Soil threats as listed by various European studies

	Threat	Louwagie et al., 2009	Jones et al., 2012	Stolte et al., 2016
1	Erosion by wind	X	X	X
2	Erosion by water	X	X	X
3	Floods and land slides	X	X	X
4	Degradation of peat soils			X
5	Carbon loss in mineral soils	X	X	X
6	Compaction	X		X
7	Salinisation and sodification	X	X	X
8	Contamination	X		X
9	Acidification		X	
10	Loss of soil fertility			
11	Desertification		X	X
12	Loss of aboveground biodiversity			
13	Loss of soil biodiversity	X	X	X
14	Spread of soil borne diseases			
15	Sealing (land take)		X	X

Note: Some threats recognised in this study were not marked by other studies. Among these, the loss of soil fertility can be repaired easily and may therefore have been omitted; the loss of aboveground biodiversity relates to the impact of soil management, rather than to the status of the soil; the spread of soil borne diseases is part of soil biodiversity, not necessarily of its decline. Moreover, tackling this threat calls for highly focussed measures, not generally applicable to combat the decline of soil biodiversity, and this threat (soil borne diseases) has a better defined bearing on primary production.

Louwagie et al. (2009) state that EU soils store more than 70,000 Mt of organic carbon, as compared to about 2,000 Mt of carbon altogether emitted by member states annually, and argue that “releasing just a small fraction of the carbon currently stored in European soils to the atmosphere could wipe out emission savings in other sectors of the economy”. Schils et al. (2008) concluded that “the largest emissions of CO₂ 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 high content of organic matter.” (Quoted from Van den Akker et al., 2016).

Organic carbon occurs in all soils. Peat soils are treated separately because threats to peat soils are quite unique, as are precautions to avoid them. Organic matter in peat and peaty soils has been typically formed during millennia, usually in thick layers and under natural conditions that inhibit decomposition; such layers may occur to great depth. Change – foremost by drainage – of natural conditions triggers peat soil degradation through aerobic decomposition and the loss of the accumulated C stock. In contrast, mineral soils typically hold organic matter in their top layer, where it reflects the balance between input and

decomposition of biomass during current and recent historical use (decades to centuries). Essentially, peat soils should remain in their natural state (un-drained and essentially under anaerobic conditions) to safeguard their C stocks. Indeed, recent work suggests that peat degradation, once triggered by drainage, cannot be reversed or halted by subsequent ditch blocking, due to the release of exo-enzymes (Fenner and Freeman, 2011; Peacock et al., 2015).

We follow the FAO definition of peat soils. For our purpose, organic soils (Histosols) are peat and peaty soils with at least 40 cm total thickness of layers that each hold at least 12% organic carbon within the top 100 cm of soil (12% C corresponds to about 20% organic matter). Further, the FAO definition includes also shallow organic rich soil overlying ice or rock. Van den Akker et al. (2016) point out that peat soil definitions vary with taxonomies used in different countries, and can be complex. While the FAO taxonomic criteria are the basis for numerical estimates (below), peatland degradation processes obviously extend to similar soils with strata rich in organic material that, however, do not meet the diagnostic requirements.

EU peat soils occupy about 32-34 Mha (Byrne et al., 2004; Schils et al., 2008) in the EU member states and Candidate Countries. Stolte et al. (2016) estimated the peat soils area at 22.9 Mha (for EU27), based on Joosten (2009). They calculated a conservative stock of 18,700 Mt of C in these soils, just over the 17,000 Mton estimated by Byrne et al. (2004). These values correspond to about 25% of the above (Louwagie et al., 2009) stock of total organic C in all EU soils. More than half the European peat soils area is located in Norway, Finland, Sweden and the UK.

About 5.8 Mha of EU peat soils is drained, of which 3.6 Mha is under agriculture: 0.95 Mha as cropland and 2.65 Mha as grassland (Schils et al., 2008). Germany, Poland, the Netherlands together hold about 2.4 Mha of drained peatland under agriculture (Schils et al. 2008), which is 70-85% of their total peatland area (Van den Akker et al., 2016). Agricultural use succeeding initial drainage (cultivation and conversion to arable land, liming, use of nitrogen fertilisers) causes rapid mineralization of organic matter and degradation of peat soils (Kechavarzi, 2010). Other degrading factors (Van den Akker et al., 2016) are erosion, wildfires and climate change (increased temperature; drought periods). Drainage itself sets off a process of subsidence which involves consolidation and compaction, oxidation of organic matter mediated by microbiota, and shrinkage by drying. As a consequence, further drainage is then required to keep groundwater below the falling topsoil level. While annual subsidence of drained peatland can locally be in the order of several cm per year, a long term mean of about 2 mm/y was found by (Erkens et al., 2016) who estimated a total subsidence of 1.9 m for the Dutch coastal plain over the past millennium.

4.2.2. Impacts on functions and services

Mineralization of peat soil substrate upon drainage may set free substantial amounts of nutrients which support crop growth, depending on the nature of the peat. Other services are negatively impacted: water buffering, habitat for biodiversity, and carbon storage. Peatlands under arable farming have much lower biodiversity capacity than grasslands. The transformation of peat grassland into arable land after the peat layer has been oxidized is generally associated with a strong decrease of biodiversity (Van den Akker et al., 2016). Substantial literature is available on peatland restoration. The bottom line is that the restoration of peatland may restore some of its CO₂ sequestration capacity, but not (in the short term) to the net carbon sink rate found in natural bogs (Waddington and Price, 2000). While peat soils under agriculture occupy less than 1% of the EU area under

agriculture, it is doubtful that the remaining 99% of EU mineral soils will not have the capacity to absorb all C potentially released from the drained agricultural peatland (Van den Akker et al., 2016).

Besides the loss of habitat for biodiversity, peatland degradation causes massive emission of CO₂, as well as associated N₂O. After Indonesia, the EU is the second largest hotspot for peatland CO₂ emission, with a total of 173 Mt CO₂ per year from drained peat soils (agriculture 100.5, forestry 67.6, and extraction 5.6 Mt CO₂ per year; Van den Akker et al., 2016), or about 4% of current annual EU28 GHG emission of 4,420 Mt CO₂ from combustion (<http://www.eea.europa.eu/data-and-maps/data/data-viewers/greenhouse-gases-viewer>, year 2014). The intensity of CO₂ emission (t/ha/y) from peatland under agriculture is about 50 times higher than under forestry. EU agricultural peatland emits 25 t CO₂/ha/y (or 27 t/ha/y CO₂-eq including N₂O associated with peat oxidation), based on reference rates (Oleszczuk et al., 2008) of 20 (grassland) and 40 (arable land) t CO₂/ha/y (Schils et al., 2008; App. 1). The intensity for forested peatland is about 0.5 t CO₂/ha/y (weighted average Finland and Sweden; Joosten, 2009). [Conversion coefficients: 1 kg C corresponds to 3.67 kg CO₂ and to 1.82 kg SOM.]

Transformation of peat soils under agriculture is widespread and rapid. Van den Akker et al. (2016) suggested, based on Schils et al. (2008), that more than 18,000 km² (1,8 Mha) of peat soils have already been lost (no longer classified as peat soil) by oxidation. The transformation rate is illustrated by a case in the Netherlands, where in few decades between 1965 and 2005, over 50% of the area in the northern Netherlands previously classified as peat soils was no longer classified as such (Van Kekem et al., 2005). Similarly, over 50% of peaty soils (soils with substantial strata of peat material that fail to meet diagnostic criteria for Histosols) had been measurably degraded (thickness of peat layers and/or organic matter content in peaty layers decreased). In 12% of the area, peat layers had vanished completely (De Vries et al., 2014).

4.3. Erosion by water

Erosion by water is a more important problem for the European soils than the erosion by wind. Contrary to the wind case water erosion occurs everywhere, in particular when either the rain falls with great intensity, the soil is erodible, i.e. its aggregates are more susceptible to be dislodged or destroyed, the slope enhances the gain of kinetic energy by the water running downhill, or there is inadequate vegetation cover on the surface. There are several types of water erosion, sheet and rill erosion, what in geomorphologic terms is known as diffuse erosion, and linear or concentrated erosion, when the concentration of water generates gullies with wall landslides and abundant mass of sediments.

Water erosion has been a great problem throughout human civilization. Erosive processes form part of the rock cycle in geology, acting at different time scales from the geological time scale to centennial to decadal scale, where human intervention is more appreciated. Estimations made by Wilkinson and McElroy (2007) indicate that human activity in agriculture, construction, and mining have outstripped the rate of geomorphic change by natural processes. The estimated erosion rate of 0.016 mm/year of Phanerozoic time has increased to 0.6 mm/year in agricultural areas. In a similar way Montgomery (2007) describes the role of erosion in the decline of the great civilizations of history, as a consequence of the behavior of man as a parasite on soil in the apposite words of Edward Hyams (1952).

Water erosion damages are in-site loss of soil fertility, and off-site contaminant dispersal as well. A peculiar problem inherent to water erosion is the silting of reservoirs. Syvitski and Kettner (2011) observed that in spite of the acceleration in the production of sediments, in the last thousand years their load contribution to the oceans decreased due to the proliferation of dams. Reservoir silting is a menace for the supply of freshwater in the future. Graf et al. (2010) inspected the state of several North-American reservoirs, pointing out that their respective capacities were diminishing, albeit at different rates. Kondolf et al. (2014) and Schleiss et al. (2016), present some options for the prevention and mitigation of the silting problem.

The assessment of the magnitude of both soil losses and sediment dispersal at European scale is not an easy task. Several attempts have been made, e.g. Van Rompaey et al. 2003 and Van Oost et al. (2009), but, as Govers et al. (2016) recently explained, the quality of soil loss rates estimates at a great scale is rather low either for extrapolation of limited number of data, or based on model predictions with neither a proper calibration nor a correct parameterization at the relevant spatial scales. At any case the estimates of Cerdan et al. (2010), their Table 9, for water erosion due to sheet and rill erosion, and those of Van Oost et al. (2009) for the sediment fluxes transported by water, their Table I, give a reasonable perspective of the importance of this problem.

As in the case of wind erosion, the role of man in accelerating soil loss and sediment yield by agronomic and forestry practices, in particular by tillage erosion - also known as mechanical erosion: the clod breaking and subsequent displacement and redistribution of soil particles by agricultural and forest implements - is evident. In the estimations of van Oost et al. (2009), their Figs. 5 and 6 and Table I, the intensity of tillage erosion can be more important than that of water erosion in several regions of Europe, especially in the Mediterranean countries. This, with the reservations indicated by Govers et al. (2016) about the accuracy of these estimates, is a relevant factor to consider for the future of agriculture in the continent.

Govers et al. (2016) make very opportune reflections on the required conservation works for this century, based on (i) clear observations on the information we have on soil erosion processes, (ii) the available technology for soil conservation, (iii) the consideration of the different concurrent processes, (iv) the motivation of the farmers, and (v) the permanent revision of the conservation strategies. Techniques applied are not always the most adequate, since a certain philosophy prevails in many conservation agencies to choose one single solution for all cases.

4.4. Compaction and related physical problems

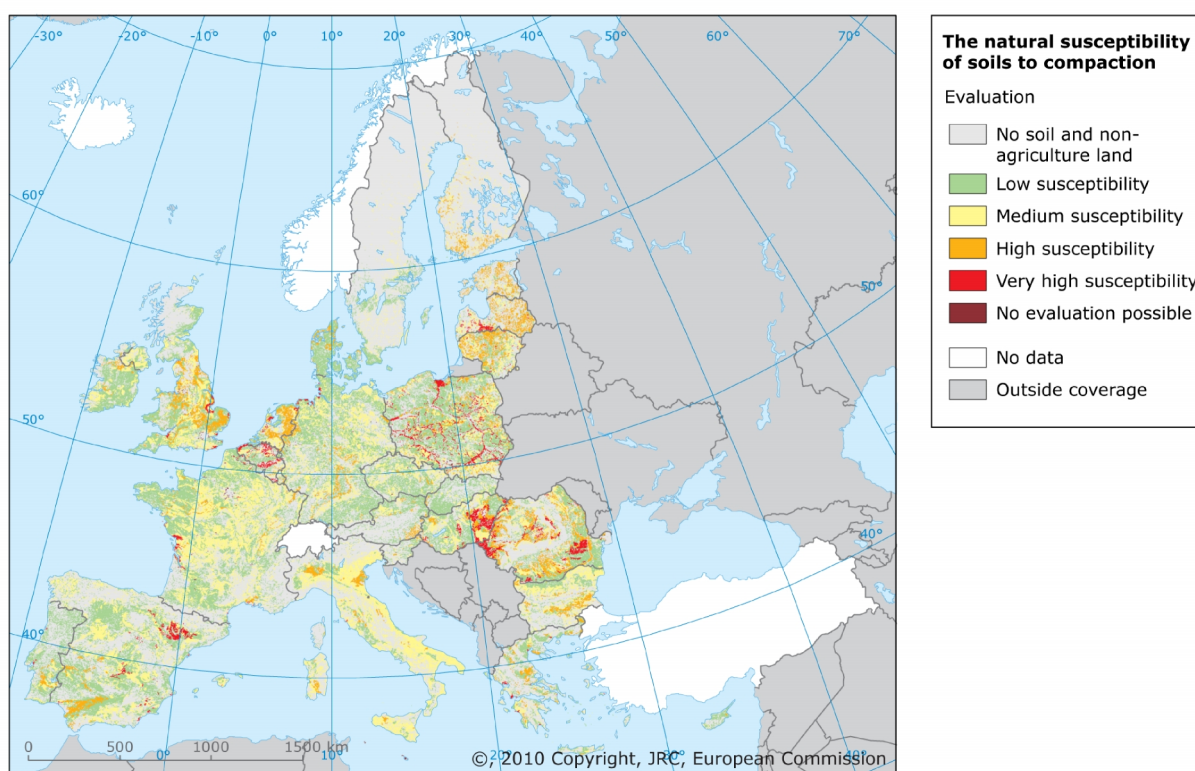
Soil as a porous medium located at the Earth's surface is very susceptible to compaction forces. The immediate effect of a load applied on the soil surface is the compression and deformation of the aggregates reducing the volume of the larger pores, often called structural pores with effective diameters between 10^{-4} to 10^{-2} m (e.g. Tuller and Or, 2002), through which most of the fluid transfers take place, a place for the roots to grow in search of nutrients, and where water is retained.

As Hamza and Anderson (2005) acknowledge, soil compaction, one of the most important processes of soil degradation, is very difficult to detect requiring a close inspection to evaluate the structural changes in the soil and their potential effects on crop growth and development. The compaction of the soil can be estimated by measuring the changes of bulk density, soil water retention, air permeability, or the resistance to penetrometer

probes. The agricultural and forest use of the soil induce compaction. Håkansson et al. (1988) show the total wheel-track area covered by several vehicles for different crops in Sweden. This study indicates great compaction risks considering that, for instance, a tractor passes the same spot at least three times during a cereal growing season. The compaction induced in soil and subsoil by the agricultural machinery wheels, especially under wet soil conditions, is a matter of great concern. Berli et al. (2001) recommended the pre-compression stress developed in Soil Mechanics, as a threshold not to be surpassed if the soil deformation must be kept within the elastic range, and, consequently, reversible, avoiding any plastic deformation. Compaction effects are persistent. Sharratt et al. (1998) detected the effects on the bulk density of the soil of the Wadsworth Trail in Minnesota active between 1864 and 1871, and Brevik and Fenton (2012) reported similar effects - on the thickness of the A horizons and in the saturated hydraulic conductivity - in the Mormon Trail in Iowa active between 1846 and 1853.

Soil compaction can be caused in the successive wetting and drying cycles during a normal year as has been explained by Ghezzehei and Or (2000) and Or et al. (2000). When the soil drains the matric component of the soil water potential generates shear stresses which can exceed the shear strength of their aggregates. Then the aggregates deform coalescing into cluster through welding at the contact areas. The consequences of these processes are crust formation at the surface, and change of the pore size distribution. Additional deformation is induced by tillage implements. These effects are more pronounced in loamy soils without the cohesive forces of the clay soils or the resistance properties of the sand particles in coarse soils. The use of heavy tillage implements like the disk harrow, causes a plow sole which hinders the pass of roots, water and nutrients to deeper soil layers and induces subsurface runoff and erosion particularly in tilled fields (Verbist et al. 2007). Surface crusts can be separated by the genesis (Valentin and Bresson, 1992) in structural or erosion crusts formed by the impact of raindrops, consisting of a fine clay layer covered by sorted sand layers and depositional or runoff crusts, formed in the depressions. Biolders et al. (1996) detected both mechanisms and the distribution of the two main crust types and their intergrades in the field. Fig. 5 – Fig. 10 in Jones et al. (2012) - indicates the susceptibility of European soils to compaction.

Several methods have been suggested to alleviate soil compaction like the addition of organic amendments to increase the organic carbon content of the soil to develop a better soil structure with reinforced bonds between aggregates, the reduction of machinery, or even cattle trampling on the soil surface, the controlled traffic along fixed 'tram lines', the modification of agricultural machinery tyres to reduce the applied pressure on the soil, and the use of crops with vigorous deep rooting (Spoor, 1996). However, it is primarily the wheel load carrying capacity that determines subsoil compaction, e.g. at depths of 0.5-1.0 m, and that should be the parameter to be addressed by imposing legal limits to soil loading in agriculture (Schjønning et al., 2015).

Figure 5. Soil compaction susceptibility in Europe

Source: Jones et al. (2012).

4.5. Floods and Landslides

Floods originate when the incoming water cannot infiltrate into the soil at the same rate as it arrives. The soil profile in this condition may be fully saturated with water, producing what is known as the saturation excess. The infiltration rate at its surface is smaller than the supply rate, causing an infiltration excess. The runoff flow on the surface can coalesce inducing a discharge rate that cannot be conveyed by the fluvial network moving outside of the river channels. Locally the water flow can create, or enlarge a gully, through which water, sediments and other objects, such as vegetation residues, move. The flow can be accompanied by landslides in the walls that increase the sediment load, which may be fully displaced or left partially blocking the gully bed.

For the initiation of a flood the rain may appear suddenly in the form of intense showers causing flash floods, or it can fall with not too great intensity but in a persistent way, as described by Gioia et al. (2008) in their flood probability model. These types of rainfall were identified in the long term study of a gully network in southern Spain by Hayas et al. (2017).

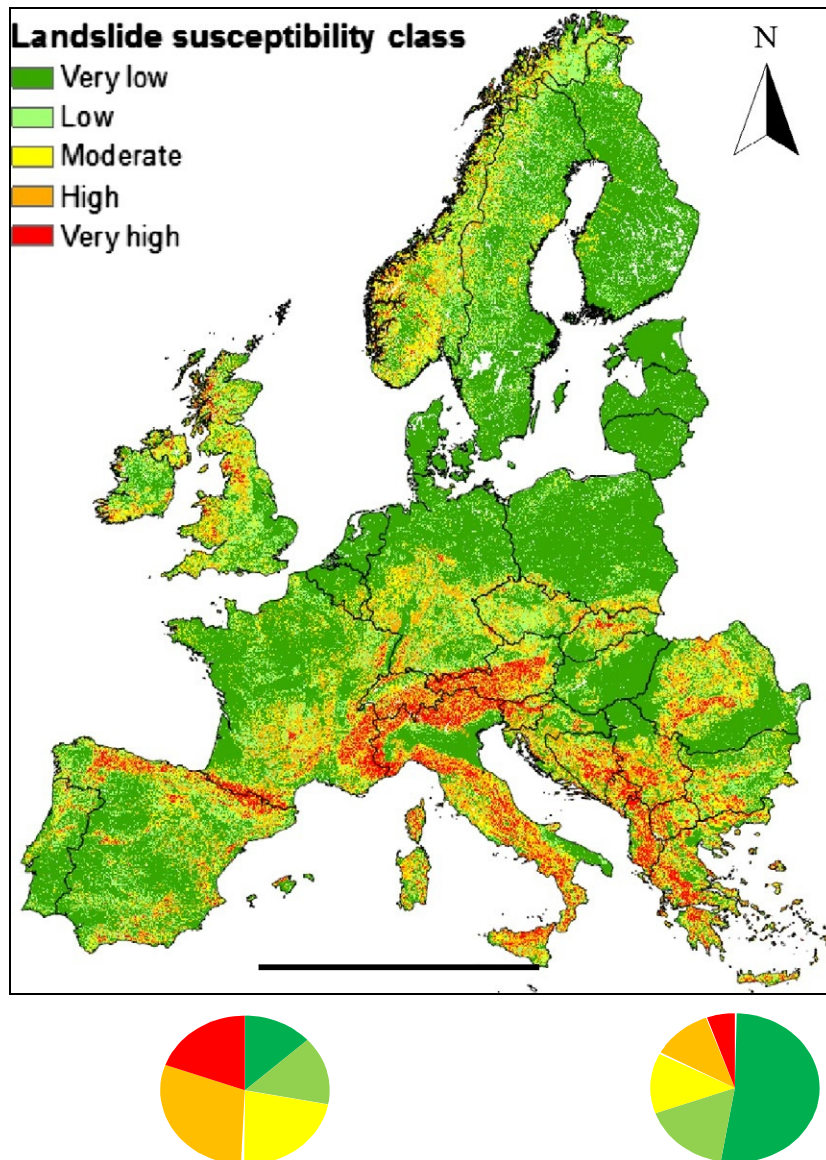
River floods can be produced by dam breaks. These are usually associated with landslides or debris flows. As Blöschl et al. (2015) explained, the main processes controlling river floods are i) atmospheric conditions (intense rains or rapid snow melting); ii) catchment conditions through changing land uses or soil states which induce a fast and effective runoff generation; and iii) the characteristics of the rivers themselves which enhance or retard the flow conveyance. Merz et al. (2014) provide in their Fig. 3 a clear scheme of the interaction of factors of the flood formation. The recent examples of floods in Europe in Germany, the UK and France indicate the threat not only for the soil but also for urban

environments. The occupation of potential flood areas and the absence of riverbed cleaning measures enhance the risk of flooding. Paoli (2016) discusses the practices of integrated flood management.

The landslide process starts when the component of the land mass weight parallel to the surface is no longer resisted by the friction and cohesion forces, and some trigger, such as an intensive rain or some earthquake, initiates the movement. The reduction of the vertical force due to the water pressure in the soil enhances the landslide risk. The vegetation reduces the risk due to the increase of the cohesive forces, although the increment of weight can act in the opposite way. The landslide triggering effect of rain was described with a simple but effective model by Iverson (2000). A more complete explanation has been given by Lehman and Or (2012) who represent the landslide as the culmination of local failures in a cascade of successive events traveling along the hillslope. The hillslope in the model of Lehmann and Or (2012) is a set of soil hexagonal prisms held by the roots of a tree, for example an olive tree, which are interconnected by frictional and tensile mechanical bonds. The connection of the forces follows the fiber bundle model (e.g. Hansen, 2005). When a soil-rock interface fails, the load is redistributed to neighbouring columns, propagating either downhill or uphill, producing a catastrophic event. This phenomenon is often observed in sensitive areas, where entire prisms or soil blocks move as isolated units towards the bottom of the gullies.

In regions with swelling and shrinking soils, in addition to the landslides during the rainy season, there are some landslides produced by the release of cohesive forces after large dessication cracks have formed during the dry summer season, although with smaller consequences for the soil.

Using a landslide database of the different countries of the European Union (van der Eeckaut and Hervás, 2012), the Joint Research Centre (JRC) developed a model based on an ordinary logistic regression to produce a map of the landslide susceptibility for the whole EU (van der Eeckaut et al., 2012). Another landslide susceptibility model has been elaborated by the International Centre for Geohazards (ICG). A comparison between the results of both models (Jaedicke et al., 2012) shows some differences, but both identify the mountainous regions of the EU, especially in the Southern countries, as the more susceptible for landslides. They distinguish rainfall and earthquakes as the main triggers. These types of estimates are not always precise. Corominas et al. (2014) have prepared a complete set of recommendations to perform a quantitative analysis of landslide risk. Günther et al. (2013) introduced a Tier-based approach for the assessment of landslide susceptibility. In the Günther et al. (2013) approach, Tier 1 - given the slope of the terrain, the land cover and the lithology - identifies the priority areas for landslide susceptibility. Based on these identified priority areas, Tier 2 evaluates the susceptibility at finer scale but requires more complete information of the area. Fig. 6 (Fig. 7 in Günther et al., 2014) indicates the landslide susceptibility across Europe.

Figure 6. Classified Pan-European landslide susceptibility map

Source: Günther et al. (2014).

4.6. Decline of soil organic matter content in mineral soils

4.6.1. Dynamics of SOM pools

Soil organic matter (SOM) is the residue of primarily plant biomass that may have passed or not through various trophic levels (of soil organisms), changed its composition, and that was prevented, one way or another, from conversion back to CO₂. Where conditions are very restrictive for biotic decomposition (cold/wet/acidic), the accumulation of plant biomass may lead to the formation of purely organic strata (organic soils, Section 4.2). In mineral soils under agriculture, however, input and decomposition of organic material are of roughly the same order of magnitude, and their balance defines the SOM equilibrium level approached in the long run (decades to centuries). (We use SOC – which refers to only the carbon fraction of soil organic matter – and SOM side by side here, as the cited literature uses either of these, and one cannot always replace the other, depending on the context.)

A considerable fraction of SOM in agricultural soils may date back to pre-cultivation times, when the soil was formed under natural vegetation. After converting forest or grassland soils to arable farming, it is virtually impossible to sustain their usually large initial SOM stocks. This is because annual decomposition in mineral soils is roughly proportional to the size of the SOM stock (all SOM being substrate for heterotrophic organisms), whereas new SOM is formed only at a fixed rate from annual inputs (crop residues, possibly manures), which are limited in arable systems. The combination of a proportional breakdown rate with a fixed input rate by itself - i.e. irrespective of stabilisation or saturation mechanisms discussed below - dictates that the SOM stock seeks an equilibrium. Freibauer et al. (2004) gives estimates of the losses associated with the above transitions in land use.

Humification is the transformation of dead biomass into decomposed material where the original biological tissue structure is no longer recognised (Whitehead and Tinsley, 1963), or generally into a more stable organic fraction that is increasingly resistant to further breakdown. The humification coefficient (hc) is the fraction of an input cohort (of organic material) that remains present after approximately one year of exposure in soil. This coefficient is among the key parameters regulating the SOM balance (Saffih-Hdadi and Mary, 2008; Salazar et al., 2011). So higher hc values express higher recalcitrance, which decreases in the order of lignified material > ruminant manure > monogastric manures > cereal straw > green plant material. Values of hc for root material can be about twice those for residues of shoot material (Rasse et al., 2005; Katterer et al., 2011; Berti et al., 2016). Values of hc decrease with higher temperature regime (microbial activity) and increase with clay content, as protective associations are formed between SOM and micro-aggregates and clay particles (Katterer and Andren, 1999; Falloon and Smith, 2009; Yoo et al., 2011; Poeplau et al., 2015). This is why it is difficult to accumulate SOM in light soils, and in warmer climates. According to some studies (e.g., Poeplau et al., 2015) nitrogen supply can increase the hc of materials with high C/N ratio such as straw, due to N supporting the microbial efficiency (more microbial tissue formed per unit SOM respired). This would present a dilemma when C sequestration for climate is among the aims of SOM accumulation, because N supply also implies losses of the GHG N_2O , especially in presence of C). Initial decomposition relates to easily degradable components such as sugars and proteins but also hemi-cellulose and cellulose, while lignin materials are more recalcitrant (Lashermes et al., 2012). Removal of easily degradable components also occurs in digestion by ruminants or during composting, which is why cattle manure and composts have higher hc values than fresh plant material (Thomsen et al., 2013).

The SOM equilibrium level approached over time depends on the amount of annual inputs, their hc -value under local conditions, and the relative decomposition rate of the accumulated SOM pool. The latter two coefficients depend on the initial composition of the organic material (notably C/N ratio and lignin content), environmental conditions (notably temperature, soil moisture), and stabilisation or protection mechanisms that are largely related to soil texture (Six, Conant et al. 2002). Such mechanisms define a maximum ('saturation') level for SOM stocks, depending on soil properties. Also (Tan et al., 2014) reported evidence for feedback of SOC stock on decomposition rate. The fact that storage is limited has obvious consequences for C-sequestration strategies.

In spite of hc differing for different input materials, some studies suggest that it is foremost the total rate of annual fresh C application that governs SOC accumulation over longer periods. (Bhoganal et al., 2009) reported that the same fractions of C input were retained - as SOC - from crop residues (22% over 23y) as from cattle manures (23% over 9y), as mean across several long term experiments (LTEs). Also Maillard and Angers

(2014) found little effect of initial composition of manure types on SOC accumulation, in a meta-analysis over 49 trials. Such apparently conflicting evidence merits closer inspection.

SOM accumulates mostly in the topsoil and is mixed, in tillage systems, throughout the plough layer. Nevertheless, a considerable fraction of treatment induced changes in SOM stock can be found in the upper subsoil, too, and the SOM in the subsoil is typically more recalcitrant than in the topsoil (Chirinda et al., 2014). (Katterer et al., 2014) found this fraction to be 27% (in the 25-40 cm depth interval), showing that SOM balances (and SOM monitoring as part of climate change mitigation schemes) should not be restricted to the plough layer. For Danish agricultural soils the amount of carbon in the subsoil (25-100 cm depth) is equivalent to that of the topsoil (0-25 cm) (Taghizadeh-Toosi et al., 2014).

SOM plays a central role in many soil processes and is at the base of the food chain for virtually all soil life, which in turn builds soil structure (aggregation by hyphens, glues etc.; burrowing) and may help suppress plant pathogens, while liberating plant nutrients in the turnover process. A good soil structure, in turn, regulates gas exchange and the infiltration and retention of water, and allows soil exploration by plant roots. SOM also governs the retention of water and nutrients directly by adsorption due to its physico-chemical properties. Its role in supporting crop water supply is thought to be primarily due to allowing better/deeper root growth and infiltration, than to the increase of soil water retention properties (which is in the range of 5-10 mm per % SOM in the plough layer).

Aggregate stability, too, is favored by SOM and is, among other conditions, vital in protection against erosion by water and wind. Morari et al. (2016) calculate a reduction of soil loss by water erosion by 50%, due to a doubling of SOM from 2% to 4%, based on the Universal Soil Loss Equation. All these processes and properties are closely related to SOM, but different functions are likely performed by different SOM fractions. The separate assessment of such relations is complex and requires comparison between interrupted and continued contrasting treatments in LTEs (as in Bhogal et al., 2011). While diverse source materials may affect soil properties differently, (Bhogal et al., 2009) concluded that effects on soil properties – similar to those on SOC stock itself, see above – are largely determined by just total C input as overall factor. They did report, however, larger benefits for soil physical properties from manure-derived SOC than from crop residue-derived SOC.

Kotschi (2013) recently stated – based on an experiment by Mulvaney et al. (2009) – that the use of mineral fertiliser N reduces SOM contents via enhanced decomposition. However, Powlson et al. (2010) had earlier shown that this trial was inadequate for such a conclusion, as N input levels were confounded with other factors. Russell et al. (2009) showed that N fertiliser use rather has a positive effect on the amount of C returning to the soil via residues, although the overall effect may be countered by an increased decay rate resulting from reduced C:N ratios. Apparently, too little N has a negative effect on the amounts of crop residues returning to soils (and thus on SOM), whereas excess N does not help SOM accumulation. (Apart from, possibly, a positive effect of additional N on microbial efficiency which might promote SOM accumulation as stated above (Poeplau et al., 2015)).

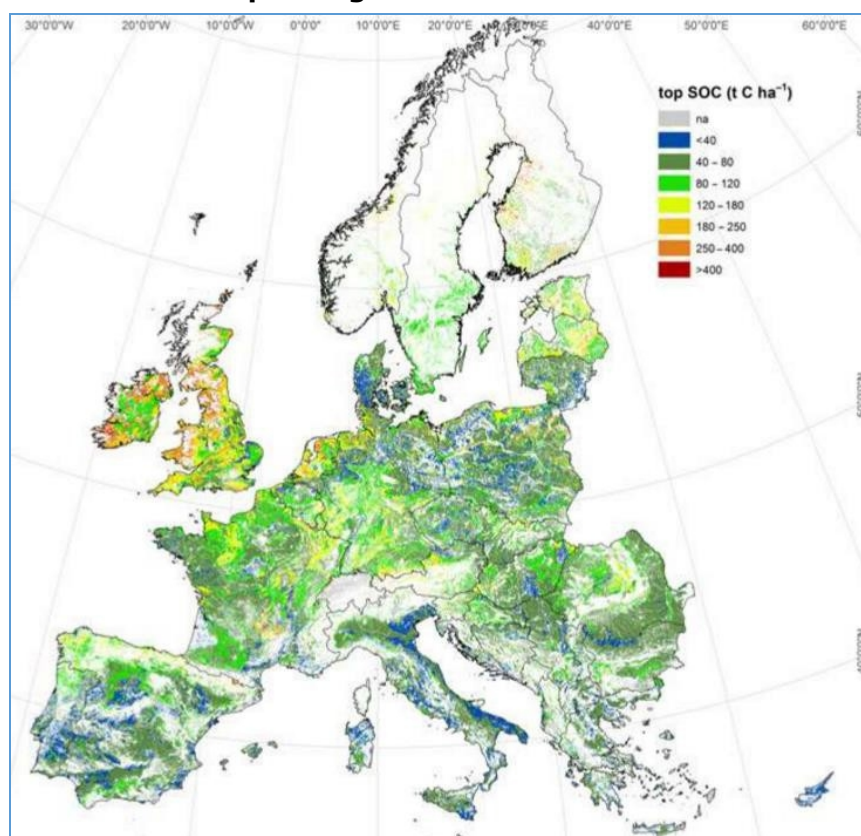
4.6.2. Threats to SOM

Based on soil descriptors in the OCTOP database (Jones et al., 2005)), Morari et al. (2016) give SOC stocks of 13,000 Mt in arable and 8,500 Mt in grassland soil of the EU28, for the 0-30 cm topsoil. They stress that SOC estimates remain uncertain and present other estimates based on, respectively, an EU25 survey (LUCAS Topsoil survey, Toth et al., 2013), and on modelling (CAPRESE). The CAPRESE effort applied the CENTURY model (Parton et al., 1987) to a grid expressing soil-climate-land use combinations (Lugato et al., 2014). According to CAPRESE, SOC is estimated at 17,600 Mt in 0-30 cm in agricultural soils at pan-European scale (i.e. an area expanded beyond the EU28), or some 20% below the above estimate from OCTOP. This contrast illustrates the difficulty and uncertainty involved in the estimation of SOC stocks, and more so of their changes over time.

There are only few estimates of changes in SOC stocks based on monitoring. Morari et al. (2016) cited results from the UK and Scotland over 1978-2007 (Reynolds et al., 2013); (Chapman et al., 2013), respectively), and the Netherlands over 1984-2004 (Reijneveld et al., 2009), all of which reported nil or positive changes. In contrast, decreases were reported for Finnish cropland over 1974-2009 (Heikkinen et al., 2013) where mineral soils lost 0.4% (220 kg C/ha) of their SOC stock annually; for Wallonia in Belgium over 1955-2005 where cropland soils lost 0.25% (116 kg C/ha) of their stock annually (Goidts and van Wesemael 2007); and for Flanders where mean annual loss was 480 kg C/ha over 1990-2002 (Sleutel et al., 2007). The Flanders case, however, also included conversion from grassland to arable cropping, where SOC stocks commonly drop by 30% to 60% in succeeding few decades (Guo and Gifford, 2002; Lettens et al., 2005). For Italy (national scale), Fantappie et al. (2011) estimated that SOC stocks in 0-50 cm during the 1991-2009 period had decreased by about 20% as compared to the 1961-1990 window. While meadows were more affected than forest or cropland soils, declines were found for all three land use types. This study combined extensive monitoring (> 17,000 observations) with multivariate spatial regression. Mixed results were found in Bavaria (Germany) over 1986-2007 (Capriel, 2013), with SOC in 54% of cropland fields unchanged, in 25% of fields declined (on average 1.35% of their stock per year), and in 21% of fields increased (on average 0.95% of stock per year). An extensive analysis for Denmark (590 sites under agriculture) first showed no decline over 1986-1997 (Heidmann et al., 2002) but did show a decrease over longer period (1986-2009) (Taghizadeh-Toosi et al., 2014), of 200 kg C/ha/y or 0.14% of the SOC stock (0-100 cm) annually, which loss was attributed in part to higher temperature in the second decade (Taghizadeh-Toosi and Olesen, 2016).

Modelling (CAPRESE) allows to explore effects of drivers such as land use change and climate change on SOC, and to account for the effects of soil properties and other factors on a uniform basis. Fig. 7 (Lugato et al., 2014)) shows the calculated SOC distribution in European agricultural soils. Morari et al. (2016) summarized from that study that the smallest current stocks were simulated for the Mediterranean region (often below 40 t C/ha), largest in NE Europe (80 to 250 t C/ha), and small stocks (<40 t/ha) in coarse glacial deposits (parts of Denmark, northern Germany, Poland, Lithuania, in spite of their latitudes above 50 °N). Over various climate-emission scenarios up to 2100, the model predicted C losses in southern and eastern Europe (on 30% of European area), and gains in central and northern Europe which outweighed the above losses. The net gain over the pan-European area was explained from increased C input (higher atmospheric CO₂; and more favorable crop growth conditions at higher latitudes), that would offset increased soil respiration.

Figure 7. Soil organic carbon (SOC) stock in the top soil layer (0-30 cm) of European agricultural soils



Source: Lugato et al. (2014) as cited in Morari et al. (2016).

Factors that contribute to the threat of SOM decline (and therefore SOC decline) specifically in the Mediterranean were summarized by a European expert team (EIP-AGRI Focus Group, EFG; Anonymous, 2015): high soil temperatures, erosion as promoted by often rugged terrain with steep slopes, the occurrence of torrential storms, low soil cover due to drought (summer) and land use (vineyards, olives; Meersmans et al., 2012), calcareous soils, limited extent of grassland, and occasional wildfires. Based on field-measured plot data, Maetens et al. (2012) reported mean annual soil losses of 10-20 t/ha from orchards (versus <1 t/ha for (semi)natural vegetation). (On the other hand, the frequent occurrence of stony soils and clay soil in the pan-Mediterranean are factors that mitigate runoff and soil loss, according to Maetens et al. (2012)). Regarding irrigation, a practice widespread in the Mediterranean, there is conflicting evidence as for its effect on SOM (Costantini and Lorenzetti, 2013). While moisture-induced microbial activity is expected to enhance decomposition, biomass inputs (crop residues) may also increase under irrigation, depending on anterior land use.

The CAPRESE results seem to be contradicted by recent extrapolations (statistical modelling) from 49 long term experiments in Europe, North America and Asia (Crowther et al., 2016). That team suggests that SOC stocks would drop under all their climate scenarios, especially at high latitudes. They calculate net emissions from SOC equivalent to 12-17% of total anthropogenic GHG emissions over the next 35 years, and warn that this could 'drive a positive land carbon - climate feedback that could accelerate climate change'. As these predictions are not process-based, and their statistical frame covers a wide range of biomes including the permafrost zone, the relevance of these outcomes to EU agricultural soils remains unclear in our view.

In short, there is no consensus of current trends in the evolution of SOM stocks in EU soils, and the picture is definitely complex. Nevertheless, it is clear that SOM stocks are low and/or decreasing in specific, extensive regions of the EU.

4.6.3. SOM-related soil services

Below we document possible effects of declining SOM stocks on two main soil services commonly associated with SOM: primary production and climate change mitigation. For the many interactions with soil threats we refer to Annex II.

Effects of SOM decline on services – primary production

Farmers generally recognize the importance of maintaining SOM levels in their soils, as found in a survey among 2500 respondent farmers in over 20 different farm types in seven European MS (Pronk et al., 2015). Yet, direct relations of crop yields with SOM are not commonly found in research, possibly because SOM benefits for crops may appear under extreme conditions only (drought, flooding, disease outbreaks), or below a certain threshold in SOM content not commonly crossed in trials. Ever since the arrival of mineral fertilisers, attempts were made to quantify the importance of SOM for crop production (e.g. at Rothamsted, UK, starting 1850. Johnston et al., 2009). The assessment is complex and outcomes are highly variable across the many long term experiments (LTEs) in Europe. A major difficulty is that SOM levels cannot be varied without simultaneously affecting other factors, most obviously nutrient supply. In factorial trials, contrasts in SOM level are rather an outcome of contrasting practices - similar to yield contrasts - than an imposed factor contrast. So while the practices themselves (green manuring; use of animal manures and composts; retention of crop residues, etc.; see Chapter 5) can be tested, SOM benefits can only be estimated by correlation. One way to exclude nutrient effects from such trials is to assure nutrient sufficiency, using mineral fertilisers to compensate for differences in nutrient supply. Following this approach, Hijbeek et al. (2017) assessed the 'additional yield effect' (i.e. an effect beyond that caused by the associated nutrient supply) of practices that contribute organic inputs, in a meta-analysis across 20 LTEs in Europe with different crops. They found no overall significant additional yield effect. They did report, however, significant benefits for potato (+7%), maize (+4%), and for spring sown cereals (+3.4%). These results largely confirm (Johnston et al., 2009) who emphasized benefits of SOM for spring sown and tuber crops because of their limited root systems or shorter time windows to exploit the soil for water and nutrients. Hijbeek et al. (2017) also found significant correlations between increase in attainable yield and increase in SOM content (root and tuber crops; spring cereals), and generally larger 'additional yield effects' of organic inputs on light soils and, surprisingly, in wet climates.

The above studies refer to ranges of N supply high enough not to limit crop production. Alternatively, at lower levels of N supply it is –by definition- this very factor which largely determines yield. Under such constraint, organic inputs support lower yields than do mineral N fertilisers, at equal total-N input rates (Schröder et al., 2005; Schröder et al., 2007; Schröder, 2014; Zavattaro et al., 2014). This implies that total N losses per unit output are larger under organic manuring, contributing to emissions of ammonia, nitrate and N₂O, with impacts on acidification, eutrophication and climate, respectively.

C-sequestration for climate change mitigation

Soil organic carbon (SOC) constitutes 48%-58% of SOM (Morari et al., 2016) and is an important sink in the C cycle. (In discussing the link with climate, as below, we prefer using the term SOC over SOM, as it is the C-component of SOM that is relevant to C-

cycling. However, where citing original sources we use the term – either SOM or SOC – as used in the source.) The need to protect and augment SOC stocks as a measure in climate change mitigation strategies has long been recognized (Freibauer et al., 2004). Ignoring the immense amount of C locked in geological carbonate deposits governed by the long-term C cycle (Berner, 1998; Levi, 2000; Beerling, 2007), the global SOC pool is second to only the oceanic carbon pool, and is over twice the size of the atmospheric, or more than three times the size of the biotic C pool (Janzen, 2004).

The enhancement of SOC stocks in soils has also been called for in the recent “4-per-mille initiative” at COP21 (UNFCCC 2015). It derives its name from the estimate that an annual increase of 4 per mille of current global SOC stocks would make up for the current gap between anthropogenic CO₂ emission, and the absorption of CO₂ in oceans and forests. To assess whether this is a viable option and how it should be approached, requires careful accounting. Net C-sequestration from the atmosphere requires either (1) reducing the rate of organic C conversion into CO₂; or (2) increasing the rate of C sequestration from the atmosphere. Ignoring CO₂ trapping by mineral carbonation (C sequestration by rock weathering), option (2) simply implies increasing net primary production (NPP). Option (1), within the context of agriculture, involves foremost the protection of current SOC stocks. Additionally, practices could be considered that deliberately aim to accumulate SOC, such as retaining crop residues. There are serious doubts, however, whether soil incorporation of crop residues (and other organic waste streams) would reduce CO₂ emissions, as compared to alternative uses. To the contrary, bioenergy produced from this biomass could replace fossil fuels, possibly a more effective mitigation measure than storage in soil (Poeplau et al., 2015). The amount of CO₂ emission thus avoided was estimated to be seven (!) times larger than the equivalent SOC increment, in the case of cereal straw over a period of 100 y (Powlson et al., 2008). Of course, soil incorporation remains a valid mitigation option if compared to plain incineration/landfilling of organic waste without energy recovery. Further, reduced tillage has been suggested as a promising measure to increase the residence time of SOC (i.e., Option 1), but it appears with recent estimates to have very limited effect (Powlson et al., 2014). As for increasing NPP (Option 2), the inclusion of grassland in rotations, and the use of cover crops are among the most obvious options (see Chapter 5).

Several organic waste streams increase N₂O emission upon application to soil (Baral et al., 2017) which can offset their contribution to C sequestration (Bos et al., 2015). For Denmark, Taghizadeh and Olesen (2016) calculated that the main options for increasing SOC in Danish soils (returning straw; cover crops; conversion to grassland) if applied to realistic extent, would amount to 0.5 Mt CO₂ per year, offsetting only 5% of agricultural methane and N₂O emissions. Finally, measures to enhance soil C input by shifting the rotation towards crops with more residues (such as cereals), or larger root-C input (cereals, grassland) can be very expensive to the farmer, with cost estimates exceeding current EU ETS carbon credit price by 100-fold (Bos et al., 2016).

At the EU level, a valid question is: where in the EU would SOC accumulation be most efficient from a mitigation perspective? Given the smaller decomposition coefficient in cooler climates and finer textured soils, regions with those conditions could present the better option. In addition, if accumulated SOC indeed increases its relative decomposition rate (Tan et al., 2014), then soils with smaller SOC stocks should also be favored – unless their small stocks reflect poor SOC retention characteristics of the environment in the first place. Perhaps more relevant is how much organic waste could be mobilized, in excess of current use in agriculture. Based on Meyer-Kohlstock et al. (2015) it can be calculated that all compost that could potentially be produced in EU28 from the organic fraction of all municipal solid waste (OFMSW or bio-waste) (44 Mt compost) and industrial food processing (15-25 Mt compost)

adds up to a total of maximum 69 Mt, or about four times current production of 17.5 Mt compost. That potential would suffice for about 6.4% of EU28 arable land area (108 M ha), at an annual application rate of 10 t/ha. The amount of C in 69 Mt of compost (16.5% C, Meyer-Kohlstock pers. comm.) corresponds to 0.09% of the current SOC stock in EU28 arable soils (13 Gt in 0-30 cm, Jones et al., 2005). With a hc of 0.9 (Katterer et al., 2014), application to soil of the extra 51.5 Mt (beyond 17.5 Mt current use) annually would constitute an annual increase by 0.06% of SOC stocks – that is, if SOC stocks were not currently decreasing under actual management.

We conclude that the scope for increasing SOC stocks as a climate mitigation measure is very limited due to (i) constraints on the availability of C-sources, (ii) nitrous oxide emissions associated with most practices that enhance SOC accumulation, and (iii) in view of farm economy. Moreover, (iv) only carbon that originates from extra primary production and carbon that is saved from incineration contributes to climate mitigation. Further, (v) the use of organic biomass for bioenergy - replacing fossil fuel - is likely to contribute more to climate mitigation than soil incorporation. Finally, (vi) gains in SOC made by adjusting farming practices are reversible – i.e., that SOC can be lost rapidly if the practice is discontinued – while fossil fuel savings forgone and N₂O emitted are irreversible. However, practices to promote the accumulation of SOM in agricultural soils, or mitigate its decline, can definitely bring agronomic benefits and contribute to the protection of soils and various soil services. (See also Chapter 5).

4.7. Decline of biodiversity

4.7.1. Soil biodiversity

Tibbet (2016) distinguishes (after Jensen et al., 1990) ecosystem diversity (variety of habitats in the soil); species (i.e., taxonomic) diversity; genetic diversity of a species (combination of different genes within a population of one species, and variation between populations of the species) and across a community; as well as phenotypic and functional diversity. Globally, soils are believed to contain a quarter to one third of all living organisms on the planet (Jefferey et al., 2010, as cited by Tibbett, 2016). This enormous diversity is attributed largely to the heterogeneity of microhabitats in soil. In short, biodiversity comprises all biological variation from genes to species, up to communities, ecosystems and landscapes (MEA 2005). It is the variation in soil life from genes to communities, as well as the variation in soil habitats, from micro-aggregates to landscapes (Turbé et al., 2010).

Very little is known about the taxonomic diversity of soil biota. The organisms are grouped (Swift et al., 1979) into macro-fauna (larger than 2 mm; e.g. earthworms, up to badgers), meso-fauna (0.1 to 2.0 mm; e.g., mites, springtails, enchytraeids), micro-fauna (smaller than 100 microns; the single-celled protozoa, the multicellular nematodes and rotifers), and microorganisms (bacteria, archaea, fungi; some authors include viruses, too). For each of these, the number of species known, and estimated fraction that known species represent of the total number in each group, is given in Table 3.

There is less functional than taxonomic diversity, as species share overlapping functions (Wellnitz and Poff, 2001), although Tibbet (2016) warns that such similarity may not necessarily relate to species survival success in case of extreme events. Turbé et al. (2010) distinguish three 'all-encompassing ecosystem functions' by soil biota: (1) transformation and decomposition; (2) biological regulation; (3) soil engineering. A characteristic assemblage of organisms that can perform such function is called a functional group. The

Table 3. Estimated global number of aboveground and belowground organisms

Group	Organisms	Known	% Known
Plants	Vascular plants	270000	84%
Macro-fauna	Earthworms	3500	50%
Meso-fauna	Mites	45231	4%
	Springtails	7617	15%
Micro-fauna	Protozoa	1500	7.5%
	Nematodes	25000	1.3%
Microorganisms	Bacteria	10000	1%
	Fungi	72000	1%
Marine species	All marine organisms	230000	30% ⁵

Source: Turbé et al. (2010). Adapted by them from De Deyn and Van der Putten (2005) and Wall et al. (2001).

corresponding groups, respectively, are 'chemical engineers' (bacteria and fungi), biological regulators (these regulate microbial activity and include protists, nematodes, microarthropods), and the 'soil ecosystem engineers' (after Jones et al., 1996) that alter soil structure by bioturbation (earthworms, termites, ants, isopods, moles).

Plant roots are also seen as members of the latter category. Services provided by the above respective groups are mineralisation of organic material (chemical engineers), contributing to element cycling and plant nutrition; suppression of plant pathogens (regulators), and contributing to the hydrological cycle (ecosystem engineers promoting soil structure, water infiltration and retention).

Drivers/Threats

Tibbet (2016), following Huber et al. (2008) (ENVASSO project), defines the threat of decline in soil biodiversity as the 'reduction of forms of life living in soils (both in terms of quantity and variety) and of related functions'. Such decline is thought (Tibbet, 2016) to cause a reduced resistance to change (of diversity and functions), as well as reduced capacity to recover after perturbation (resilience).

Turbé et al. (2010) distinguish land use change, climate change, chemical pollution, GMO's and invasive species as major drivers of decline in soil biodiversity. The impact of these on biodiversity is often indirect, that is, via other degradation issues (our 'threats'): erosion, SOM decline, salinization, sealing, compaction, and contamination (Montanarella, 2007). As separate sections in this report are dedicated to these specific threats, we focus in this section on direct effects of the drivers listed by Turbé et al. (2010). Admittedly, there are overlaps between the direct and indirect mechanisms.

Different pressures are associated with the different 'levels' at which biodiversity is expressed: at the soil ecosystem level, these are changes in land use, climate, exploitation intensity, hydrology and soil chemical properties. At species diversity level, changes in chemical properties, competition with invasive species, and ecotoxins. For the genetic level, 'genetic pollution' is regarded an additional pressure (Jeffery et al., 2010).

Soil biodiversity adapts to changes in *land use*, but it may take decades to reach new equilibria, (Van Veen et al., 2006) and for ecosystem services to adjust accordingly (Turbé et al., 2010). As this study is confined to agricultural land use, shifts involving forestry or urban use are ignored. Changes then relate to shifts between arable cropping and grassland systems, and to shifts between practices within either of these systems (see Chapter 5). Grassland soils generally present richer biodiversity than cropped soils, and inclusion of

grassland in an arable rotation has been recommended to restore biodiversity, SOM and disease suppressiveness (Garbeva, 2004). Cropland soils, due to regular mechanical disturbance, tend to host fewer arbuscular mycorrhizal fungi as well as fewer earthworms (Turbé et al., 2010).

Whereas it is widely recognised that land use shifts cause long-lasting changes in community structure and species abundances of soil biota (Korthals et al., 2001; Buckley and Schmidt 2001; van der Wal et al., 2006; Smith, 2008), we found little evidence that such shifts – within agricultural land use – would involve irreversible changes in biodiversity, or that the potential of soils to deliver ESS would be affected primarily via changes in biodiversity.

Intensity. The widespread use of fertilisers, pesticides and herbicides, monoculture cropping and the removal of residues in intensive farming are considered important causes of soil biodiversity loss (Tibbet, 2016; Wachira, 2014). Reduced tillage systems, and more generally conservation agriculture (Louwagie et al. 2009), aim to address this aspect (see Chapter 5). Organic farming, with its often high inputs of plant residues, may promote only some specific taxa, such as earthworms (Birkhofer et al., 2008, cited in Turbé et al., 2010).

Climate change may involve changes in mean temperature, precipitation, cycles of freezing/thawing and wetting/drying (that may affect aggregation), photosynthesis, and – through feedbacks – atmospheric CO₂ level. All of these may affect community structure and species abundance, as well as services provided by the soil via soil biodiversity. Direct threats can also arise from extremes such as flooding (followed by anoxia and compaction) and prolonged drought (Tibbet, 2016).

The likely effect of rising temperatures on microbial activity, and thereby on soil respiration, soil carbon dynamics and atmospheric CO₂ has been debated, with conflicting evidence from laboratory (increased respiration) versus open field studies (microbial acclimatisation) (Turbé et al., 2010). However, a recent meta-analysis (49 field experiments in three continents) suggests that a rise in average global soil surface temperature by 2 degrees C up to 2050 ('business as usual'-scenario), will increase soil respiration by an amount equivalent to 12-17% of expected anthropogenic emissions over this period (Crowther et al., 2016). The microbially mediated decline of soil carbon stocks, however, should be regarded as a major general soil threat rather than a direct threat by climate change to soil biodiversity.

Climate change can affect soil organisms differentially, promoting some and reducing others. Insect pests are believed to generally become more abundant with rising temperatures (Cannon, 1998, in Turbé et al., 2010). Turbé et al. (2010) distinguish direct effects (migration/range expansion; seasonal shifts; length of reproduction cycle; overwintering) from indirect effects that affect interactions of pest species within the system. They stress that effects are highly context dependent, and call for a case-by-case approach in formulating precautionary measures. They conclude that all mitigation measures taken to limit global climate change are expected to have a beneficial impact on soil biodiversity preservation, soil functioning and associated services.

Soil contamination. See separate section on soil contamination.

Genetically modified organisms (GMOs), too, can be viewed as a potential source of pollution by various pathways (Turbé et al., 2010): altering the quality and quantity of growth substrates, microbial community structure, efficiency of microbially mediated

processes, or the genetic transfer between GM crops and bacteria. Those authors suspect that the normal operating range (NOR) of organisms in agricultural soils is very wide, due to drastic impacts that standard practices (tillage, manuring, use of pesticides) already exert on the community. They call for future studies with an emphasis on functions, notably decomposer and enzymatic functions, rather than 'biodiversity as a whole'. Kolseth et al. (2015) judge the risk of GMOs upsetting the soil ecosystem as being small, relative to substantial changes induced by the different crops themselves that pass over the soil in rotation systems.

Invasive species arrive through transport (soil, plant material), tourism, and migration, the latter enhanced by climate change. Many possible interactions causing community disturbances have been documented (Turbé et al., 2010). Biologists recognize that invasive species can often be controlled by introduction of their natural enemies, but are generally weary of this option as numerous examples worldwide have shown that this may create new problems. Limitation on imports of plant materials is a general precaution which is, however, complicated by the low predictability about which species will become invaders (Turbé et al., 2010).

Relation to other soil threats and soil functions

As stated, soil biodiversity decline is closely related to other soil threats (Stolte et al., 2016), which by themselves affect biodiversity. Those are treated in separate sections of this chapter. Conversely, biodiversity decline might also aggravate other threats, such as soil pollution where biodiversity plays a role in remediation/detoxification (Tibbet, 2016). Rather than defining one-way cause effect relations, however, biodiversity decline is likely to be part of negative feedback loops, such as lower activity (causing reduced porosity, bioturbation, aggregate stability) leading to physical degradation (and reduced infiltration and moisture retention) in turn reducing vegetative cover, energy (organic matter) input, and increasing runoff and erosion. Similar feedbacks can be supposed for other threats, functions, and services. As this study focuses on soils under agriculture, the occurrence and control of soil-borne diseases are among the most crucial aspects of biodiversity that affect crop production, the major service of agricultural soils. We therefore include a dedicated section to soil-borne diseases.

4.7.2. Soil health: soil-borne diseases and soil suppressiveness

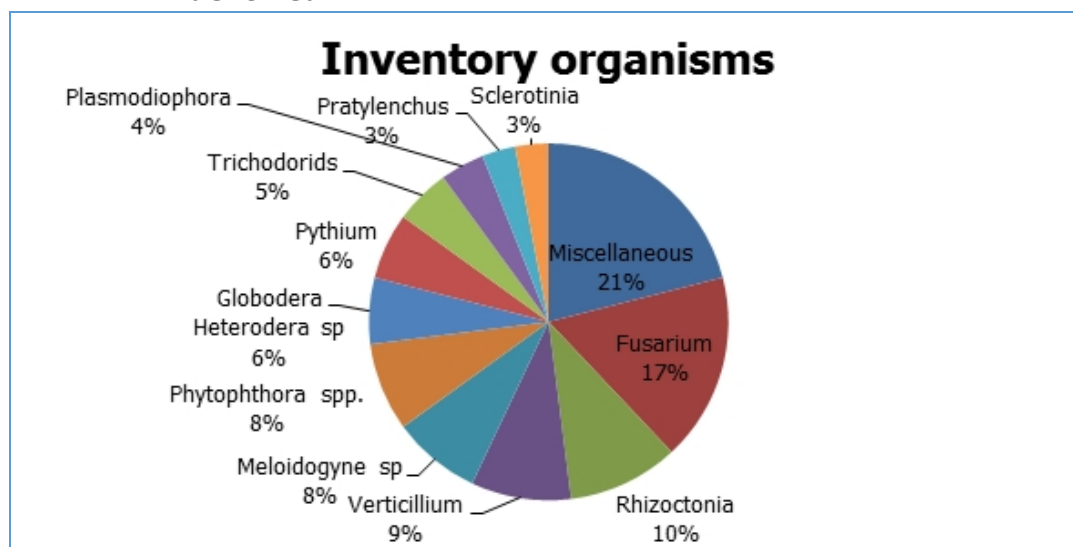
Soil borne plant pathogens are among the more studied components of soil biodiversity. Soil borne diseases are caused by fungi, nematodes, bacteria and viruses. Viruses are often transmitted by nematodes and fungi. Plant parasitic nematodes alone reduce world global yields by some 10%, representing a value loss of \$125 billion annually (Chitwood, 2003). Especially since the UN Montreal protocol, which phased out the use of methyl bromide as a common control measure – and with additional bans on disinfectants such as dichloropropene and methylisothiocyanate – the prevention and control of soil borne diseases has become a matter of simultaneously optimising soil and crop management practices, towards an integrated strategy (IPM) for soil health. A European expert team (EIP-AGRI Focus Group, EFG; Molendijk, 2015) reviewed these challenges. They focussed on fungi and nematodes as these seem to have the largest incidence and impact on crops. We sample here from the EFG team's findings.

- No statistics on soil borne diseases are available from EU MS on incidence, acreage and yield loss, or data are treated confidentially by authorities (apart from quarantine organisms). Exceptions are Austria, Scotland and Spain. The EFG, therefore, resorted to literature and questionnaires. They found *Fusarium* spp, *Verticillium dahliae*, and *Rhizoctonia solani* (fungi), and the nematode genera of *Meloidogyne* and *Globodera* to be

the most important organisms causing wide-spread diseases in Europe. Other important pathogens are listed in Table 4. Most affected crops include olive, tomato, potato, cucumber, carrots, lettuce, sugar beet, and brassicas. Fig. 8 shows the relative importance (crop species affected) of different organisms, within the set of 128 disease x crop combinations selected in the EFG study.

- Soil health is the biological condition of the soil which determines yield attainable in absence of water or nutrient stresses. More than just the absence of disease, it constitutes the ability of the soil itself to cope with new incoming diseases and to keep population levels of pathogens - a small minority of species - sufficiently low to prevent crop damage (Janvier et al., 2007). The capacity of soil to achieve this through antagonistic activity of microbiota is called 'soil suppressiveness'. Disease potential is linked to inoculum density (pathogen population level), and to the inoculum capacity to provoke a disease. Both density and capacity/activity of the inoculum are largely controlled by soil suppressiveness. Other factors that control actual crop damage include the genotypic tolerance of the crop, climatic conditions, and soil properties such as pH, aeration, humidity and nutrients, all of which affect crop vigour. The EFG states that 'everything that improves the vigour of the crop increases its tolerance to damage'. This, they explain, should be seen as a sound foundation only. Next should come a dedicated soil health strategy, which goes beyond a focus on specific pathogens.
- While mechanisms are poorly understood, the EFG regards high infestation levels arising from poor biodiversity and poor soil structure as the main causes of soil borne diseases. Indeed, soil health is believed to be promoted by practices that aim to improve physical, biological and chemical aspects of soil quality. These include reduced tillage, smart rotations, the use of compost and green manures, avoidance of machinery that damages soil structure, increasing SOM, and good fertilisation practices.
- The EFG stress the importance of, besides preventive measures, awareness and knowledge among players in the entire production chain, aided by knowledge-based planning (including DSS tools), and monitoring. Top research priorities, as ranked by EFG, are (1) identification of best protocols for the application of biocontrol agents (BCA), these are among the most promising innovations; (2) developing science-based sampling strategies and high-throughput diagnostics; (3) finding indicators to predict the suppressing quality of composts and similar products, based on knowledge of underlying mechanisms. The development of a soil health strategy is discussed in Chapter 6.

Figure 8. Inventory of major soil-borne pathogens in EU agriculture, as identified through questionnaires among members of EIP Focus Group 'IPM Practices for soil-borne diseases'. Pie size reflects the importance of the pathogen species or genus in Europe. The class 'Miscellaneous' refers to diseases of only local importance. In total (100%) 128 combinations of disease-by-crop (or sector) were identified



Source: Molendijk (2015).

Table 4. List of the most common soil-borne pathogens in the EU

Fungi	Nematodes
Verticillium dahliae	Meloidogyne sp
Gaeumannomyces graminis	Pratylenchus penetrans
Rhizoctonia solani	Xiphinema index
Fusarium spp	Globodera sp.
Pythium spp.	Heterodera spp.
Phytophthora spp.	Ditylenchus dipsaci
Sclerotinia sclerotiorum	Trichodorids
Sclerotinia cepivorum	Paratrichodorids
Plasmodiophora brassicae	
Chalara elegans	
Synchytrium endobioticum	

Source: based on literature and interviews by Molendijk (2015).

4.7.3. Aboveground macro-fauna

The attention given to the ecosystem service of *habitat provision* tends to focus on the subterranean micro-, meso- and macro-fauna. This probably results from the demonstrable or surmised effects of those groups on the delivery of other services. In a broader sense, however, *habitat provision* by soils also pertains to other important groups of macro-fauna, as these too rely on what soils offer them. Examples can be found among mammals (e.g. badgers, moles) and birds (e.g. meadow birds, Turdidae). The main feed components of these animals include preys that live in the upper soil layers or close to the soil surface. In this review we have not extended the concept of *habitat provision* to the fauna that is more indirectly depending on soils, i.e. via the vegetations, herbs, crops and weeds thriving on soils. Note that it is exactly this form of biodiversity that may interact with the ecosystem service of *primary production* via competition (e.g. crop-seed eating birds, grazing geese and deer) or pest control (e.g. insectivorous and weed-seed eating species). In contrast, it is not very likely that the soil surface-related macro fauna interacts significantly with any of

the other soil services. However, soil management directed at those other services certainly does affect the surface-related macro fauna. Its intrinsic and cultural value therefore justifies that due attention is given to such management impacts.

It seems fair to say that any management that is detrimental to earthworms, is so to soil-related aboveground macro-fauna. Earthworm densities are highest where there is something to eat for them. Consequently, the presence of soil-related macro fauna is likely to decrease in the following order: fertilized grassland producing a vast amount of (overturning) roots and crop residues > unfertilized grassland > arable land amended with manures and retained crop residues > arable land from which crop residues are removed. Within arable soils a negative relationship can be found between earthworm densities and the extent to which soils are perturbed, as tillage destroys the pores they are living in and exposes them to opportunistic scavengers such as Corvidae and Laridae. Drainage also has significant effects on the macro-fauna because it increases the depth at which earthworms withdraw. Drainage can therefore reduce the availability of earthworms to species feeding on them. In the case of meadowbirds there is also an indirect negative effect of drainage, as drainage stimulates early cuts in spring that drastically reduce the survival rate of clutches and chickens (Schekkerman, 2008; Kentie, 2015). See also Section 4.8 on soil contamination.

4.8. Contamination

Soil pollution in agriculture relates to the use of pesticides; the presence of pharmaceuticals, PCBs, PAHs, and heavy metals in composts, manures and sewage sludge; as well as heavy metals in fertilisers, notably cadmium in phosphate fertilisers. More recently, biochars, too, constitute a potential source of toxic compounds (Lyu et al., 2016; Smith et al., 2016). All of these pollutants have been shown to affect soil biodiversity or species abundance. Effects of pollutants depend strongly on bioavailability of the pollutant, which in turn depends on environmental conditions (pH, moisture content, adsorption complex), and detoxification pathways in organisms. Also, urease and nitrification inhibitors in fertilisers and manures have been suspect for affecting microbial communities. Such fears were, so far, not confirmed in a recent review (CISL, 2016). The amendments themselves often cause larger changes in community structure than the agents added. Also (Suleiman et al., 2016) and (Soares et al., 2016) found no effect of nitrification inhibitors on microbial composition or diversity, or on the nitrifier community specifically.

Given the (known) impacts of chemical pollution on selected organisms, Turbé et al. (2010) expect significant impacts of pollution on soil biodiversity functioning. A function expected to be the most affected is nutrient cycling (as mediated by organic matter decomposition) and the water control service (via the regulation of soil structure by 'ecosystem engineers' as earthworms). However, to date methods capable of quantitative assessment of biodiversity impairment are still lacking, although progress was made in qualitative approaches (Semenzin et al. (2009), cited by Turbé et al. (2010)). Nevertheless, thus far there is no evidence that lack of decomposition is impeding the release of nutrients from organic compounds.

Contamination includes many aspects. It may pertain to the accumulation of heavy metals or polycyclic aromatic compounds applied in the form of sludges originating from industries or municipalities. Chemicals used within agriculture itself may, however, also contribute to a gradual accumulation of these compounds as they can be part of fungicides, fertilizers (Cd) or feed additives (Cu). There is a growing awareness of possible long-term

implications of the use of human and livestock medicines, including contraceptives, on soil services. The eventual impact depends on the capacity of a soil to bind and/or degrade these compounds. In a wider sense contaminations also encompass harmful organisms, such a newly introduced resistant weed species.

Earthworms take up organic contaminants and heavy metals from the soil and accumulate these in their body tissues (Stephens et al., 1995; Usmani and Kumar, 2015). These compounds can be present in agro-chemicals and wastes, and management therefore impacts on the soil-related macro fauna via this transfer mechanism. The application rate and composition of mineral fertilizers, organic fertilizers and pesticides is, in combination with soil properties, decisive for the eventual bio-availability of these harmful compounds and, consequently, the macro fauna. The widespread use of helminthicides in livestock production has also received attention in view of this transfer mechanism. However, the impact of these pesticides on earthworm populations appears to be limited (Yeates et al., 2007; Schon et al., 2011; Scheffczyk et al., 2016), and so are probably the consequences for the macro fauna. Similar findings were reported by Steel and Wardhaugh (2002). They pointed out that these helminthicides may yet have a negative effect on insects involved in the preprocessing that makes dung pats fit for further degradation by earthworms.

4.9. Erosion by wind

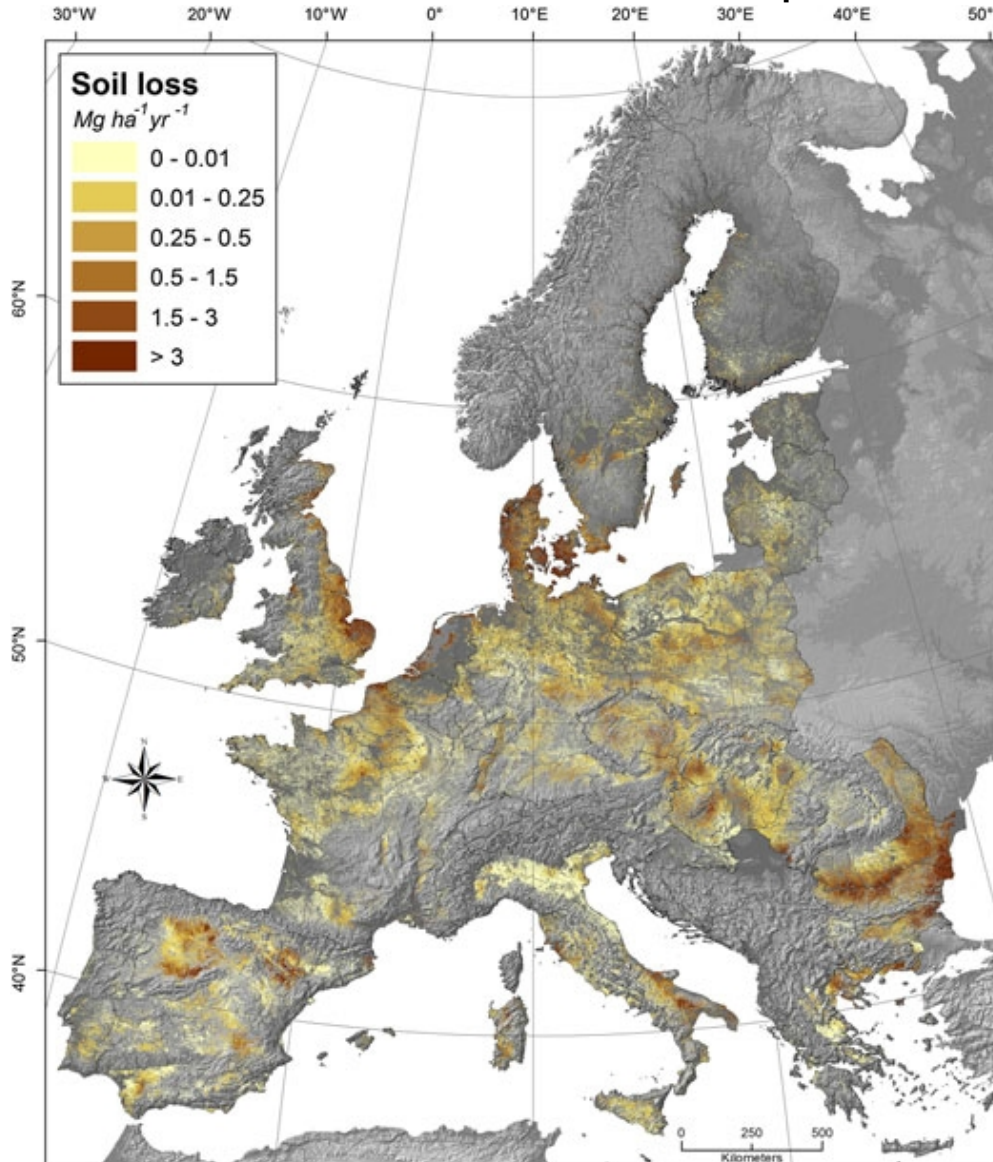
Wind erosion represents a menace for soils in the places where there is enough fetch for the flowing air to reach a threshold of speed, or shear stress, to initiate the movement of soil particles. This condition usually happens in open plains or valleys, in the shore, or in vast areas with very low relief. Usually wind erosion is restricted to desert-like environments but its effects can appear on a larger area. Soil erosion starts when certain thresholds, either shear stress or velocity, are exceeded by erosive agents like wind. Once the solid particles, sediments, begin to move they can travel appreciable distances depending on the energy of the carrier, the erosive agent, being later deposited as this energy decays. Therefore, the problems caused by erosion are basically due to the loss in the original soil, productivity decrease, and to the effects of the moving particles, abrasion in the exposed surfaces in the case of wind, and of their accumulation on soil and other surfaces. Both types are usually recognized as in-site and off-site effects, respectively. The dispersal of contaminants by the wind erosion can be a matter of concern like the 'valley fever', or coccidioidomycosis, reported in the American Southwest, caused by the displacement of a saprophytic fungus from the soil in the desert to populated areas (Malo et al., 2014).

Wind erosion moves mainly two types of particles: sand, the greatest one, with effective diameter above 0.07 mm, and dust, the finest one, with smaller effective diameters, although the boundary between them depends on the shear stress of the wind (Shao, 2000). While the sand is displaced over short distances, dust travels as a suspension between continents. Gusts from the dust storms originating in the great Dust Bowl drought of North America in the past century, arrived at Europe (Ingram and Malamud-Roam, 2013, Chap. 3). The Saharan dust appears almost regularly in the Mediterranean Basin as indicated by the optical density measurements of Moulin et al. (1998), in their plate 1, showing the solid particles density in the air during spring and summer seasons. More recently the dust emission from the tephra deposits produced in the 2010 eruption of the Eyjafjallajökull volcano in Iceland reached great part of the European continent (Arnalds et al., 2013). Some countries like the United Kingdom, Denmark, Netherlands, Romania, or Hungary, or regions like Scania in Sweden, Les Landes in France, or the Depression of the

Ebro River in Spain are affected by wind erosion generating sediments, but the aeolian deposits can fall in more countries.

As Warren and Barring (2002) indicated, this type of erosion is not a major problem in Europe but it can provoke damage in some areas. After the first topsoil survey of the European Union was made by Tóth et al. (2013) in the Land Use / Land Cover Area Frame Survey (LUCAS), Borrelli et al. (2014) evaluated the susceptibility of European soils to wind erosion, adopting an equation proposed by Fryrear et al. (1994) in their primitive wind erosion equation in what was denominated the erodible fraction (EF). Although such an equation could be questioned as being a linear combination of different particle size range contents, ratios, calcium carbonate and organic matter contents, neglecting the evaluation of intervening physical processes, the results of Borrelli et al. (2014), as reflected in their Table 2, seem reasonable. Later Borrelli et al. (2016) presented the index of land susceptibility to wind erosion (ILSWE), which includes in addition to the erodible fraction, the wind force, basically the square of the wind velocity, the vegetation cover, based on the data base and the land roughness. Their results summarized in their Table II, and Figs. 5 and 6, reduce the land susceptibility of the different countries of the European Union to a few countries like Spain and Denmark. A recent application of a wind erosion model to European data (Borrelli et al., 2017), yielded some estimates of potential soil losses in different countries. Fig. 9 gives some indication of the wind erosion threat to the soil.

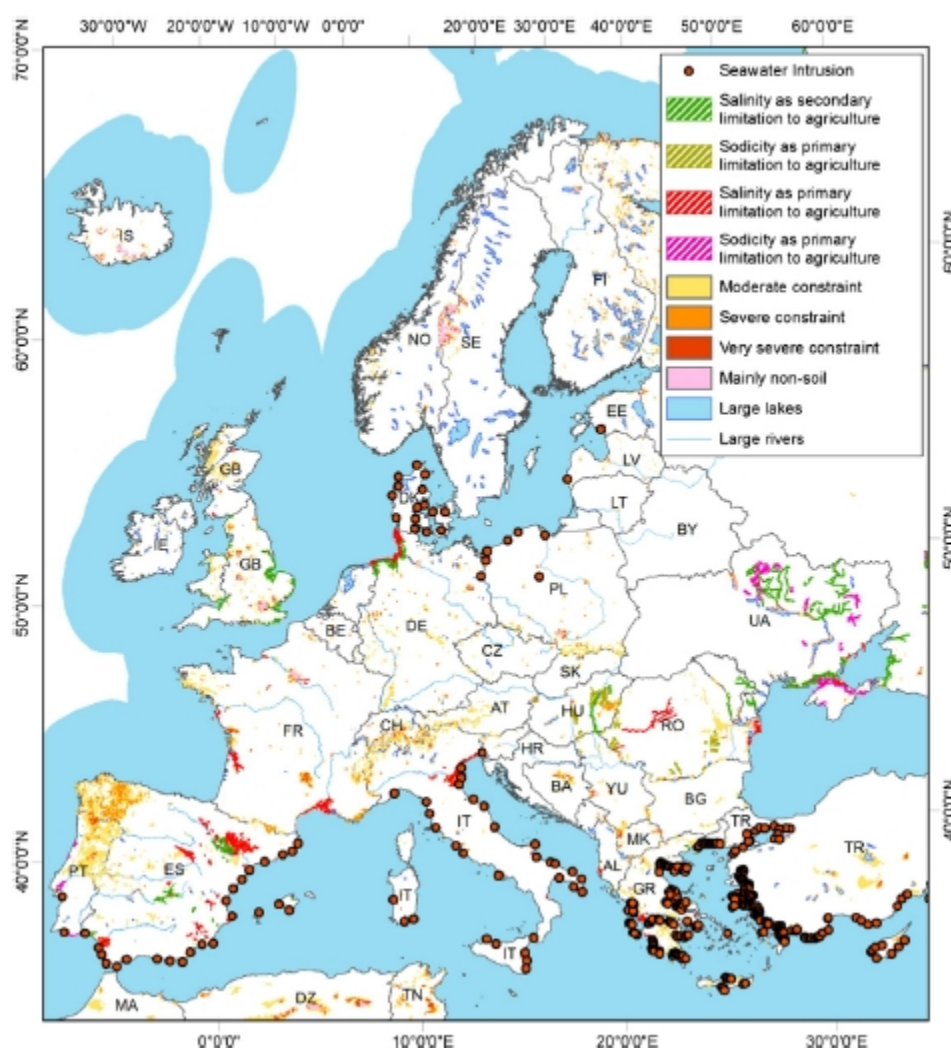
One aspect to take into account in the evaluation of wind erosion is the introduction of blowers in the agricultural management of fruit tree orchards. In a trial of macadamia orchards in New South Wales, Dalby et al. (2010) detected a soil loss due to the blowers as much as twice the loss under other harvest implements. The use of blowers in the harvest of many fruit orchards in the Mediterranean countries must be evaluated to assess their effect on soil degradation.

Figure 9. Potential wind soil loss modelled for the European arable land

Source: Figure 2 in Borrelli et al. (2017).

4.10. Salinization and sodification

Many soils in the world are affected by salts, due to its formation on saline sediments, or to the acquisition of salts either by atmospheric or by management causes. Over the oceans there is a solid suspension of salt crystals formed by the drying of sea water droplets. When the winds carry this solid suspension inland and rains leach them towards the soil, the salt particles - called cyclic salt - can salinize the soil. The use of fresh water for irrigation is another cause of secondary, or man-made salinization. It occurs when there is insufficient leaching of solutes out of the soil. In a classic article Garrels and Mackenzie (1967) showed how the concentration of natural waters increases the concentration of monovalent cations, such as sodium, and decreases the concentration of divalent ones, with the final result of salinization, or more properly, sodification risk. Salinization implies a concentration of solutes in the soil solution which can (i) reduce the osmotic component of water potential, reducing its difference with respect that of the plant solution, thus negatively affecting water absorption by plant roots; (ii) disturb the chemical

Figure 10. Saline and sodic soils, and seawater intrusion areas in Europe

Source: Daliakopoulos et al. (2016).

composition of the soil solution, affecting the normal nutrient uptake with the risk of creating toxic levels of nutrients that in nonsaline conditions could be even deficient for the plant; and (iii) produce toxicity due to the high concentration of some elements such as chlorine or boron. When the total solute concentration of the soil solution is not too high, but the relative concentration of sodium with respect the divalent cations calcium and magnesium is elevated, sodium ions can occupy a large fraction of the exchange complex provoking a risk of dispersed soil particles in the solution which - when moving through narrow pores - can be trapped, clogging pores and so producing an impervious seal. Therefore, the main problem of sodic soils is their imperviousness which prevents the water infiltration and the access of roots and many microorganisms to their interior. Sposito (2008, Chap. 12) presents a clear explanation of the physical chemistry of soil and water salinity.

Quite recently, irrigation with residual or poor quality water has been intensified (Oster, 1994). It gives a good opportunity to better utilise available water. However, the change of its chemical characteristics poses new risks that call for control towards its safe application. Smith et al. (2015) warn about the risk of concentrations of potassium and magnesium in the water, proposing a change in the sodium adsorption ratio to include all four mono- and divalent cations, the cation ratio of structural stability (CROSS).

The risk of soil salinization and sodification is elevated in Europe, since there are dry and hot periods in their climates which may increase the solute content in the solution. Daliakopoulos et al. (2016) produced a map of saline and sodified soils (their Fig. 3): soils whose electrical conductivity - a property closely related to the total concentration of solutes - is greater than 4 dS m^{-1} (for saline soils), and whose sodium to total salt concentration ratio is greater than 0.06 (for sodification risk). That map is shown here as Fig. 10.

The alleviation of the salinization problem is based on the removal of the solute addition, and the leaching of excess solute by the natural rainfall, or by irrigation with fresh water, and the installation of drainage systems. The more recommendable amendment for sodification problems is the addition of calcium ions to replace sodium ions in the exchange complex. A complementary practice is the addition of organic amendments allowing tolerant plants to establish and colonize the soil, in preparation of subsequent introduction of less tolerant species.

4.11. Soil acidification

Soil acidity is the lowering of soil pH through a number of processes to which also human activities contribute (Goulding, 2016). For soils in natural areas, forests and extensive grassland this mostly occurs through deposition from the atmosphere of acidifying gases or particles, such as sulphur dioxide, ammonia and nitric acid. The deposition of these compounds has declined over time in Europe, because of environmental policies that have emphasized the cleaning of emissions from industry and power production (sulphate) as well as from traffic (NO_x) and reducing ammonia emissions from agricultural activities.

The most important causes of soil acidification in agricultural soils are the application of ammonium-based fertilizers and urea, elemental sulphur fertilizer and the growing of legumes (Bolan and Hedley, 2003). This happens because these sources of nitrogen and sulphur convert to nitrate and sulphate, which then subsequently through leaching can acidify the soil. The fixation of atmospheric N₂ by legumes results in the formation of NH₄ within the root nodules by nitrogenase, the uptake of an excess of cations, especially K⁺, and therefore a net release of protons to balance the charge. Over time this can substantially reduce soil pH (Bolan and Hedley, 2003). Plant growth and nutrient uptake result in localized acidification through the exudation of acids from plant roots (Hinsinger et al., 2003). Also, decomposition of soil organic matter by microorganisms lead to acidification in neutral soils through the production of CO₂ that forms H₂CO₃, and the leaching of carbonate contributes to the acidification.

A change in land use from agriculture to forest generally increases soil acidity (de Schrijver et al., 2012). This is partly due to the contributions from acidifying atmospheric deposition, but the rate of soil acidification is also determined by the tree species-specific leaf litter quality and litter decomposition rates. Indeed, differences in leaf litter quality among tree species create fundamentally different nutrient cycles within the ecosystem, both directly through the chemical composition of the litter and indirectly through its effects on the size and composition of earthworm communities. Poor leaf litter quality leads to forest-floor build-up and soil acidification (de Schrijver et al., 2012).

The soil pH decreases by a series of chemical processes, including the dissolution of carbonates, the replacement of exchangeable base cations by H⁺, the dissolution of aluminium-bearing and manganese minerals, and the dissolution of iron-bearing minerals. Acidification causes the loss of base cations, an increase in aluminium saturation and a

decline in plant growth and crop yields (Goulding, 2016). Severe acidification can cause irreversible clay mineral dissolution and a reduction in cation exchange capacity, accompanied by structural deterioration.

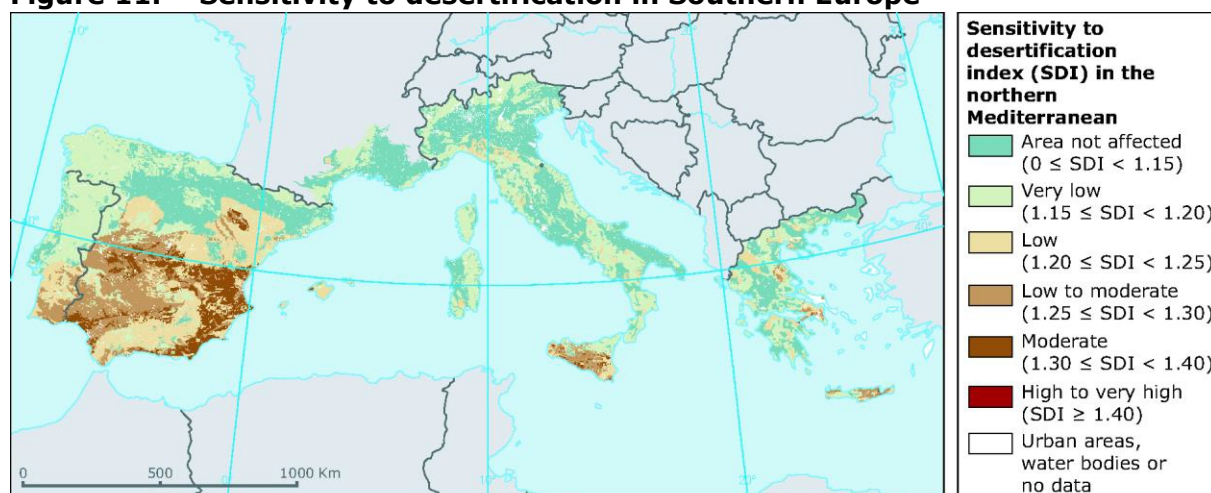
Evidence from an international survey in the Atlantic biogeographic region of Europe indicates that chronic nitrogen deposition is reducing plant species richness in acid grasslands (Stevens et al., 2010). In addition to nitrogen deposition, soil pH was found to be an important driver of species richness indicating that the acidifying effect of nitrogen deposition may reduce species richness.

In agriculture, soil acidification is easily countered by regular liming with calcium carbonate (lime) or related products (Goulding, 2016), and is important to keep soils productive which contributes to higher input use efficiencies and farm profitability. Nevertheless, farmers in certain parts of the EU seem to pay insufficient attention to soil pH. Where liming products are based on carbonates, their acid-neutralising effect goes hand-in-hand with net CO₂ emission. Basic slags (as CaSiO₃) and sulphates do not share this drawback. Similarly, flours of rapidly weathering mafic silicate rocks, such as olivine, can help neutralise soil acidity without net CO₂ emission, but they may bring pollutants (heavy metals) (Ten Berge et al., 2012).

4.12. Desertification

Desertification is a broad concept, with different connotations. Warren and Agnew (1983) distinguished between desertification as the process leading an environment to lose much of its vegetation, leaving no more than the 35% of the area covered, and land degradation, as an almost continuous damage to the sustainability of biomass production, remarking that the latter is much worse than the first process for its persistency. The management of this threat is very difficult both by the complexity of the causes and the variety of solutions. Kirby et al. (2016) detected three main processes inducing desertification: (i) soil erosion; (ii) loss of soil fertility; and (ii) long-term loss of natural vegetation, although they did not care about whether the vegetation loss could be natural or desirable, what is an important point discussed in some specific cases such as the revegetation of the Ascension Island (Catlin and Stroud, 2012). The extent of desertification is evident in the sensitivity, or susceptibility, to desertification mapped by the DISMED project, Fig. 10.1 of Kirkby et al. (2016), shown here as Fig. 11.

Figure 11. Sensitivity to desertification in Southern Europe



Source: Kirkby et al. (2016).

5. MANAGEMENT PRACTICES

Soils are primarily managed through soil cultivation (tillage), inputs of fertilisers, manures and composts (soil fertility management), management of water either by drainage or irrigation, and by the inputs of above- and belowground plant inputs through crop residue management and growing of cover crops. These measures have different functions depending on farming system and will affect soil threats differently depending on local soil and climatic conditions. There are also farming concepts such as organic farming and conservation agriculture that emphasize specific combinations of these measures.

The management measures impact soil threats primarily through

- Providing soil surface cover.
- Providing organic matter inputs for sustaining biodiversity and soil organic matter levels.
- Maintaining suitable soil chemical status (salinity, pH, nutrients).
- Maintaining suitable soil structure.
- Maintaining suitable soil water contents.

5.1. Tillage

Soil tillage involves various degrees of soil disturbance directed to the loosening of soils and to incorporation and even distribution of crop residues or applied fertilizers and organic amendments. The traditional tillage is ploughing that buries the crop residues and brings fresh soil to the surface. This procedure has the advantage that it contributes to controlling weeds and some residue borne diseases, and it generally also allows a good seedbed to be established thus ensuring good crop establishment under many different environmental conditions. However, intensive tillage also has drawbacks on soil quality as well as involving greater costs for the farmers compared with reduced or no-tillage practices. Reduced tillage involves cultivating the soil to a shallow depth to partly incorporate crop residues, whereas no-tillage involves directly seeding the crops into the previous stand of stubble and crop residues.

Crop productivity is affected in various ways by changes in tillage intensity. The main effects are through crop establishment, root growth, and water and nutrient uptake (Munkholm et al., 2013). However, there may also be indirect effects through weeds, pests and diseases. A global comparison of ploughed and no tillage systems showed an average reduction of crop yields by 9.9% with no tillage and no other changes in the cropping systems, although under dry conditions it produces yields equivalent to or even greater than conventional tillage systems (Holland, 2004; Van den Putte et al., 2010; Pittelkow et al., 2015). Decreased soil physical quality, in terms of excessive compaction of the untilled topsoil, is regarded as one of the primary reasons for yield reductions (Ball et al., 1994). This is especially problematic on weakly structured soils in humid temperate climate (Munkholm et al., 2003). Such compaction problems may partly be alleviated through use of controlled traffic where traffic is restricted to certain parts of the field (Hamza and Anderson, 2005). Alternatively, soil structure can be stimulated through biological activity as with the Conservation Agriculture approach (section 5.6).

The soil compaction that is associated with the use of heavy traffic, and which compacts the subsoil, is much more difficult to remedy than compaction in the topsoil (Hamza and Anderson, 2005). To avoid detrimental subsoil compaction Schjønning et al. (2012)

suggest that: "At water contents around field capacity, traffic on agricultural soil should not exert vertical stresses in excess of 50 kPa at depths >50 cm". Whereas some subsoil compaction may be alleviated through biological activity and natural weathering, this process is normally too slow, and therefore mechanical measures can be used (Spoor et al., 2003). These measures typically involve use of subsoiling techniques that break up a plough pan, and the subsequent use of vertical mulch to keep the fissures open. However, even though this may restore rooting and drainage, such treated soils are very susceptible to repacking. The most economical option would be to prevent soil compaction (Chamen et al., 2015).

Wind erosion occurs predominantly on the North European Plain (northern Germany, eastern Netherlands and eastern England) and in parts of Mediterranean Europe (Verheijen et al., 2009). Much of this is related to tillage that exposes the soil surface to the erosive forces of wind (Goosens et al., 2001). Therefore, reducing tillage intensity can greatly reduce wind erosion, in particular if this is coupled with stubble retention.

Water erosion is generally estimated to be the most extensive form of erosion occurring in Europe (Verheijen et al., 2009). In no-tillage systems there will be more continuous pores allowing increased rainfall infiltration. There are also typically more stable aggregates in the soil surface, and both properties reduce the risk of water erosion (Raclot et al., 2009), although the erosion risk will be further reduced with Conservation Agriculture approaches that recommend a surface cover of residues or plants (section 5.6).

Tillage may itself lead to erosion by moving soil downslope (Van Oost et al., 2006). This tillage erosion moves soil from convex slope positions, such as crests and shoulder slopes, and the soil is accumulated at downslope positions. This leads to loss of topsoil in parts of the landscape that then may severely affect crop productivity through less soil organic matter and less well developed roots in these degraded parts of the field (Chirinda et al., 2014). It has been estimated that the increase in soil tillage intensity over time in Europe has made tillage erosion more important than water erosion for the redistribution of soil. Changing from ploughing to no-tillage is a very effective measure to stop tillage erosion.

Change in soil tillage from ploughing to reduced or no-tillage will affect the vertical distribution of soil organic carbon in the soil profile, and these effects depend on both the depth distribution of organic matter inputs and vertical transport of organic matter down the profile. This may in many cases lead to a smaller effect of reduced tillage intensity on soil carbon stocks for the entire profile than if only the topsoil (0-20 cm) is considered (Baker et al., 2007; Govaerts et al., 2009). Luo et al. (2010) and Powlson et al. (2014) analysed data on soil carbon content from a series of long-term experiments with varying soil tillage intensity. Compared with the ploughed treatment, no-tillage resulted in a higher soil carbon content in the topsoil, but a similar reduction in the subsoil. For the entire soil profile, there was no statistically significant effect of tillage on soil carbon stocks. This contradicts previous assessments where reduced tillage intensity (in particular no tillage) was found to enhance topsoil carbon contents (Ogle et al., 2005; Conant et al., 2007). This resulted in estimated effects of no-tillage to enhance soil carbon content with about 0.3 Mg C/ha/year. Reducing tillage intensity was therefore considered as one of the most important measures to enhance soil carbon stocks (Smith et al., 2008), and the effect is still included in the IPCC guidelines for national accounting of soil carbon change in agricultural soils (IPCC, 2006). The recent assessments cast considerable doubt on the applicability of no-tillage for enhancing total soil carbon stocks, whereas it may still enhance carbon in the topsoil.

The net climate effect of no-tillage or reduced tillage is unclear. Whereas total SOC stocks remain unaffected – apart from SOC being concentrated in a shallower top layer - or increase only slightly, nitrous oxide emissions may increase under no-tillage, at least during the first decade after conversion (Six et al., 2004; Van Kessel et al., 2013). Both these review studies found evidence for emission to decrease again in the second decade. The effect of no-tillage on N₂O emissions, however, depends on the soil and climatic conditions, and increases are often found to be associated with heavy soils and wet climatic conditions (Rochette et al., 2008). Spiegel et al. (2014) reported an overall three-fold increase of N₂O emissions under no-tillage across LTEs in Europe, and could not confirm the decrease found by above authors over longer time periods.

Reduced or no tillage systems will have much less disturbance of the soil profile and this means that soil living organisms that are susceptible to disturbance will be favored, including mycorrhiza, earthworms and microarthropods (Bender et al., 2016). These organisms interact in a food web (Roger-Estrade et al., 2010), and they can have several beneficial effects on soil structure and contribute to crop nutrient supply (Mbutia et al., 2015). However, the eventual utilization and cycling of nutrients by crops is determined by many more factors than just the nutrient supply. The overall effect of reduced and no tillage therefore remains uncertain (Schröder et al., 2016). No tillage systems will favor the accumulation of soil organic matter near the surface, and this will act as a food source for surface living microarthropods. These will then act as a food source for surface living predators that can contribute to controlling insect pest species (Shearin et al., 2007) and reducing weed seeds (Nichols et al., 2015). However, the effectiveness of this control measure increases greatly with higher inputs of surface organic matter such as practiced in Conservation Agriculture (section 5.6). Despite the potentially increased control of weeds by weed seed eating organisms, Chauhan et al. (2012) pointed at a more extensive use of herbicides in reduced tillage systems.

On the other hand, reduced tillage intensity may also favor pest development. Indeed tillage, through its direct action on slug populations and indirect effects on habitat, is an efficient way of controlling slugs. Ploughing buries slug eggs, whereas coarse seedbeds favor slug populations (Roger-Estrade et al., 2010). Therefore, an increase in slugs is often observed after introduction of reduced or no tillage systems, and slugs may have detrimental effects on crop establishment.

Ploughing and other tillage methods is an efficient way of controlling pathogenic fungi that live or survive on crop residues. Phoma stemcanker, or blackleg, is one of the most damaging fungal diseases of rapeseed, and the fungus can survive for several years on rapeseed stubble (Schneider et al., 2006). Another example is Fusarium head blight (FHB) in wheat, which is a serious disease that not only reduces yield but also contaminates the grain with mycotoxins (Lori et al., 2009). FHB survives on surface residues of host plants (e.g. maize and wheat) and subsequently affects the wheat plants. Tillage is effective for controlling these diseases, and control of these diseases in systems with reduced or no-tillage requires higher use of fungicides or use of crop rotations that provide disconnection between host species. In some cases, on the other hand, as with Verticillium wilt of olive trees, or with the broomrape (*Orobanch*) parasitic weed that affects broadbeans and sunflower, tillage promotes the spreading of these pests.

5.2. Soil fertility management

Soil fertility management concerns the building up of suitable physical and chemical status of the soil. Soil pH is typically managed through the application of lime (calcium carbonate), which increases pH, whereas application of ammonium or sulphate based nitrogen (N) fertilisers lower pH. Nutrients can be managed through the application of mineral or organic fertilizers. Some of the nutrients are required in greater quantities (typically N and S), and in particular N will often need to be applied in sufficient quantities targeted for the specific crops, whereas other nutrients can be made available from the soil reserves, provided the nutrients are available in sufficient quantities in the soil. Organic fertilisers, including cattle manure, pig slurry, poultry manure, biogas digestate and compost, provide valuable nutrients (in the form of N, P, K and S, as well as micronutrients) to enrich the soil organic matter content and enhance soil fertility (Diacono and Montemurro, 2010). Applying organic fertilizers to the soil can reduce the need for mineral fertilizers while also stimulating crop growth and improving crop performance. However, nutrient dynamics in compost-amended soils could be affected by different site-specific factors, e.g. compost matrices, composting conditions, climate, soil properties and management practices (Diacono and Montemurro, 2010). The fertilizer efficiency of various types of organic amendmends, including manures and composts, depend on many factors among which the composition (C/N, nature of C and N) and the timing and positioning of applications (Grignani et al., 2007; Schröder et al., 2016).

Producing bioenergy from biomass pyrolysis offers an opportunity to genuinely sequester a portion of C in agricultural by-products as the recalcitrant char material, now termed biochar, which is very recalcitrant to degradation in the soil (Powlson et al., 2011). Biochar may improve crop growth through increased retention of nutrients and possibly of water. But caution is required as the land use implications of widespread production and use of biochar have yet to be fully evaluated (Sutherland et al., 2010). Also, biochar can be a source of pollutants, especially persistent organic pollutants produced during pyrolysis (Shrestha et al., 2010), although when added to polluted soils it can also decrease the bioavailability of some pollutants through adsorption (Beesley et al., 2010).

Soil pH is important for availability of many nutrients. For example, a more acidic soil fosters the availability of manganese. However, strongly acid soils may restrict soil microbial activity affecting availability of other nutrients. Adding Ca and Mg in lime or gypsum may also in some soils high in clay content improve soil structure by enhancing aggregate stability (Bronick and Lal, 2005).

Soil organic matter and soil carbon is built through the addition of organic matter. The use of manure and compost is particularly effective in building soil organic matter, since these sources add more recalcitrant than fresh organic matter (Maillard and Angers, 2014). The application of nutrients in mineral fertilisers can also enhance soil organic matter, since in particular surplus nitrogen decreases fine root biomass and, thus, reduces belowground litter production, but increases aboveground plant production and litter fall (Velthof et al., 2011). However, building soil organic matter also needs addition of other macronutrients (P and S) that also are major constituents of soil organic matter (Kirkby et al., 2011). The addition of organic matter to soils, for instance with manure and compost, leads to higher soil carbon contents that improves soil structure and lower bulk density, provided that suitable equipment and weather conditions are chosen for the application. Composting

materials can increase macroaggregation and rhizospheric aggregate stability (Caravaca et al., 2002).

Organic manures and composts benefit earthworms by providing additional food, by their mulching effects and by stimulating plant growth and litter return. Farmyard manure is a particularly beneficial form of organic amendment (Whalen et al., 1998). Furthermore, the nutritional value for earthworms of farmyard manure is considered higher than that of compost in which the applied organic matter is more decomposed and stabilized (Leroy et al., 2008).

In general, microbial populations are enhanced after the application of organic amendments. Several long-lasting experiments have demonstrated that soil biological properties such as microbial biomass is significantly improved by application of organic amendments, including farmyard manure and all types of compost (Diacono and Montemurro, 2010). The stimulation of bacterial soil life from organic amendments facilitates the development of bacterial-feeding nematodes that make N and P available to plants. Next to the nutritional effect of amendments there can be direct nematicidal effects. Application of organic amendments to the soil is also expected to increase the fungal population and hence the amount of fungivorous nematodes. However, this will mostly be the case 1) when, following bacterial decomposition, the remaining organic matter is more recalcitrant, 2) when organic amendments that are less easily decomposed are added to the soil, or 3) when soil conditions are not favorable for bacteria mediated decomposition, e.g. low pH or low nutrient concentrations (Chen and Ferris, 2000).

Soil pH influences plant disease infection and development directly by effects on the soil-borne pathogen and populations of microorganisms and indirectly through availability of soil nutrients to the plant host (Ghorbani et al., 2008). For example, potato common scab (*Streptomyces scabies*) can be severe for soils with pH above 5.2; however, increasing soil pH or calcium levels may be beneficial for disease management in many other crops. Plant diseases also respond to the level of macronutrients (N, P and K), most diseases being favored by higher nutrient availability (Ghorbani et al., 2008). Soil fertility management that enhances soil microbial activity may suppress soil borne plant pests and diseases by favoring antagonists. Such effects may be particularly favored through the application of manure and compost (Mehta et al., 2014). Unfortunately, it still remains difficult to predict which type of organic amendments is best suited for a specific disease under specific soil and weather conditions.

The use of fertilizers and manures can lead to accumulation of heavy metals (e.g. cadmium, lead and zinc) in soils, but the risk is likely small when current fertilizer standards and the concepts of critical loads are complied with (De Vries et al., 2008). The use of sewage sludge provides a particular problematic source of organic contaminants and heavy metals that needs to be treated with care in food production systems, but risks may be small unless high application rates are used (Laturnus et al., 2007).

5.3. Water management

Water management involves maintaining suitable soil water contents for soil management and for plant growth. Soils that are either too wet or too dry cannot be cultivated to establish crops, and too wet soils do not allow for mechanized traffic because this would cause substantial damage to soil structure. Soil that have very high groundwater table will not allow many plant species to be cultivated, whereas dry conditions will severely limit plant growth. To manage soil water both drainage and irrigation techniques are employed.

5.3.1. Irrigation

Irrigation will increase crop productivity and this in turn will increase the addition of crop residues to the soil. However, longer periods with suitable soil water contents will also increase soil organic matter turnover rates that deplete soil carbon stocks. Dersch and Böhm (2001) thus observed losses of 2.4 t carbon per ha caused by additional irrigation compared to a rain fed system for sugar beet and maize in a rotation with cereals in a 21 years period in the Pannonian climate. These effects are likely dependent on climatic conditions, both temperature and rainfall distribution.

Irrigation is the main factor affecting salinization of soils. Depending on irrigation practices, irrigation can either enhance or reduce soil salinity. If insufficient irrigation water is applied then salts (in particular sodium) will build up in the soil profile, negatively affecting plant production. In the Mediterranean region, irrigation has been shown to lead to increase in exchangeable sodium percentage and electrical conductivity (salinization), and decrease in organic matter (Nunes et al., 2007). The use of fertilizers and other inputs in association with irrigation and insufficient drainage cause soil salinisation, markedly in cases of intensive agriculture in compacted soils where leaching is limited (Eckelmann et al., 2006).

Soil salinization increases with time passed since start of irrigation, and may eventually compromise agricultural activity (Stoate et al., 2009). Sodification (the accumulation of sodium) results in soil degradation that may be a precursor for desertification, if these effects are coupled to climate change resulting in lower rainfall and higher evapotranspiration (Daliakopoulos et al., 2016).

5.3.2. Drainage

Drainage improves aeration of the soil and for drained wetlands, this is often associated with degradation of the large stocks of soil organic matter in these soils that would have had a permanent or temporarily high water table. These soils are often very fertile when drained, but over time the decline in soil organic matter may reduce fertility.

The largest emissions of greenhouse gases (both CO₂ and nitrous oxide) comes from drainage of peatlands, and some of these have been drained for decades or centuries. These emissions may be avoided by rewetting of these peatlands, primarily by stopping drainage. Peatland restoration is most commonly taken in two steps (Holden et al., 2004): first by the re-establishment of high water tables, and then by recolonization by important peat-forming species such as Sphagnum.

An alternative to drainage-based peatland agri- and silviculture is called 'paludiculture', where biomass is produced on rewetted peatlands. Ideally, the peatlands should be so wet that steady (long-term) peat accumulation is maintained or re-installed. In climate zones, where high production is possible, most mires by nature hold a vegetation of which the aboveground parts can be harvested without harming the peat sequestering capability. In those areas, natural peatlands are largely dominated by cyperaceae, grasses, and trees, i.e. growth forms that realize peat accumulation belowground by rootlets, roots, and rhizomes (Prager et al., 2006).

5.4. Cover crops

Adding catch or cover crops to crop rotations helps improve soil quality, reduce soil erosion, enhance nutrient cycling and water holding capacity, and as a result, potentially increase crop yields, at least in the longer term. Cover crops are grown to provide vegetative cover between rows of main crops in orchards and vineyards or between periods of arable crops to prevent erosion. They may also function as catch crops, which scavenge the remaining nitrogen after the main crop is harvested, thereby reducing losses from leaching.

Legume cover crops have been reported to increase the yield of many crops (Sainju et al., 2003). Non-legume cover crops, however, produced crop yields similar to or lower than those did without a cover crop (Li et al., 2016). Cover crops may have multiple effects on productivity. In fact, several studies found that sometimes cover crop cause an initial decrease in yield of the subsequent crop, but a positive effect in later years because of increased soil fertility (Torstensson and Aronsson, 2000). These effects are all dependent on a timely and good establishment of the cover crops (Doltra and Olesen, 2013) as well as their management in the course of the season. When destroyed too late, cover crops may compete for water and nitrogen with succeeding crops (Thorup-Kristensen, 1993).

Timely planting of cover/catch crops, such as clover, rye, or legumes, to otherwise bare soil helps to increase carbon and/or nitrogen levels within the soil, critical to soil quality. Planting cover crops increases soil organic matter (SOM) and thus soil organic carbon (SOC). SOM promotes nutrient cycling, which may result in more nitrogen available to plants and less lost through leaching. Overall, soil structure is improved, increasing workability, and reducing soil loss (erosion) and nutrient run-off. Depending on the cover crop-soil combination, water retention and infiltration may increase. On the other hand, dense cover crops may promote runoff over the leaf canopy, reducing erosion, but reducing infiltration at the same time (also due to occupation of vertical soil pores by roots). Also, cover crops compete for soil water, and may so reduce the soil water stock that remains for the main crop (in climates with winter rainfall only).

Wet soil conditions in spring, when the cover crops are to be destroyed, bring the risk of damage to soil structure. Depending on climate and species, cover crops are often killed by herbicides. If their biomass is massive, substantial soil disturbance may be involved in its incorporation. Last but not least, matured cover crops (when incorporated late, e.g. to avoid soil damage under wet early spring conditions) may cause microbial competition for nitrogen due to their high C/N ratio, leaving the main crop starved for nitrogen in its initial stage when cover crop residues are still decomposing (Thorup-Kristensen, 1993). In contrast incorporation of cover crops with low C/N ratio may increase the risk of N₂O emissions (Li et al., 2015). This illustrates the intertwining of carbon and nitrogen cycles that needs careful management to optimize climate benefits from cover crops (Pugesgaard et al., 2017).

Cover crops are particular relevant for reducing water erosion in perennial cropping systems such as orchards and vineyards (Prosdociimi et al., 2016a,b). However, they also reduce wind and water erosion substantially in arable farming systems.

There is no unambiguous conclusion possible on the effect of cover crops and green manure crops on plant-parasitic nematodes (Thoden et al., 2011). The most important aspect of cover crops is their host status for the plant-parasitic species present. This also

applies for choice of cover crops in terms of maintaining parasitic fungal diseases. However, applying a diversity of cover crop species may contribute to controlling both parasitic fungi and soil borne diseases (Hajjdar et al., 2008). This is particularly so, when plant species are grown that are non-hosts or which offer specific resistance genes to nematodes or diseases.

5.5. Residue management

Crop residues are materials usually not taken away but rather left in a field or orchard after the crop has been harvested. They include stalks, stubble, leaves, roots and seed pods. Some crop residues are removed from the land to be used as straw in stables, as animal feed or as a source of energy and may or may not be returned to the land later (e.g. with manure). In some cases, straw and stubble are burned in the field, although this practice is largely banned in Europe to prevent air pollution.

Crop residues act as a source of nutrients or as a source of stable organic matter, which improves soil physical, chemical, biological and hydraulic characteristics. These nutrient effects may be both positive and negative depending on the timing of nutrient availability and may only develop after several years (Børresen, 1999). Crop residue coverage has been observed to decrease yields because of poor weed control, excessively wet and cold soils, and poor seed placement and plant stand (Swan et al., 1994).

Crop residues remaining on the land supply additional organic matter to the soil, improving soil structure, root system development and plant growth. The addition of crop residues will enhance soil organic carbon with rates that correspond to about 10 % of carbon inputs (Kong et al., 2005), depending on climatic conditions with higher accumulation rates in cooler climates. Mulches of crop residues improve structure, reduce evaporative water losses, protect against raindrop impact and increase aggregate stability (Kong et al., 2005), modify thermal and moisture regimes and stimulate the biodiversity of the soil. The return of plant residues to soil benefits soil structure (Martens, 2000), depending on the amount and quality of the residue. Retaining crop residues on the soil surface is also an effective measure for reducing wind and water erosion (Blavet et al., 2009), by protecting the surface soil mineral particles from the erosive forces.

In general, residue removal can result in detrimental changes in many biological soil quality indicators indicating loss of soil function, physical stability, and biodiversity. Nutrient cycling can be impaired, too, except when nutrients removed in residues return to the soil at some other point in space and time (as occurs most often). Soil faunal groups may increase or decrease depending on residue quantity and quality (Abbott and Murphy, 2007). Karlen et al. (1994) found that 10 years of residue removal under no-till continuous corn, resulted in deleterious changes in many biological indicators of soil quality including lower soil carbon, microbial activity, fungal biomass and earthworm populations compared with normal or double rates of residue return. In addition, some disease-producing organisms are enhanced by residue removal, others by residue retention. Residue effect on pests and disease would depend on cropping practice, climate, and local pest or disease incidence.

5.6. Farming systems

There are many different farming systems in Europe, and they can be categorized in many different ways, e.g. depending on use of annual versus perennial crops, or the integration of livestock in the systems. Here we specifically consider the use of crop rotation, grassland management measures and two specific farming systems that are usually

compared with conventional management methods: organic farming and conservation agriculture.

5.6.1. Crop rotations

A crop rotation refers to the sequential planting of different crops on the same parcel over the course of several growing seasons. Improved crop rotations refer to specially tailored crop rotation regimes, such as alternating deep-rooted and shallow-rooted plants or alternating a series of crops to build soil fertility or to reduce disease, pest or weed pressures. Crop rotations can also include the use of intercrops where several plant species are grown at the same time.

There are many reports that crops in rotation produced more than in monoculture. Thus, Nevens and Reheul (2003) demonstrated the positive effects of 3-year grazed grassland breaks in an arable forage crop rotation: the crops following the grasslands showed high yields and, compared with permanent arable crop the N fertilization could be reduced substantially due to the release of nitrogen from the soil fertility built up under the leys. Conversely, Deike et al. (2008) reported that the total yield of the entire crop rotation was higher in continuous winter wheat cropping than in winter oilseed rape–winter wheat–winter wheat–winter barley rotation. However, these crops also have different economic values, and it may therefore still be financially beneficial to grow them in rotations, even if total yields are lower. In a 33-year experiment on sandy soil in Poland, a 1.5 years grass-clover ley once per four-year in an arable rotation cycle was equivalent to between 40 and 60 t/ha of farmyard manure per cycle, in terms of SOC accumulation (Pikula and Rutkowska, 2014). The same trial showed that leys enable nitrogen fertiliser savings of 50% to 90% in the arable phase (potato, winter wheat, spring barley) (Ten Berge et al., 2016).

Soil aggregate dynamics varies among different crops, crop rotations and cover crops (Jarecki and Lal, 2003). Rotation effects on aggregation reflect the crop chemical composition (Martens, 2000), rooting structure and ability to alter the chemical and biological properties of the soil (Chan and Heenan, 1996). These effects depend on soil texture and tend to be short-lived under conventional tillage regimes, but may be longer lasting under no tillage.

Earthworms are favored by crops that leave behind substantial amounts of carbon rich residues in the field. Therefore root crops, for which most of the crop is removed, discourage the buildup of earthworm populations (Edwards and Bohlen, 1996). In contrast legume-cereal intercrops (Schmidt et al., 2001) and grasslands (Lamande et al., 2003) provide degradable carbon sources suitable for earthworms.

Plant parasitic nematodes differ widely in their specialization on crops. Some species have only one plant family on which they can complete their life cycle, e.g. potato cyst nematode (*Globodera* spp.). Other species can propagate on a wide host range and belong to different plant families. The first group can be controlled by growing the host in wide rotations in a low cropping frequency. The second group can only be controlled by choosing the order of crops based on knowledge of the expected population dynamics of the present nematode species and the tolerance of the crops to nematode damage. Often farmer knowledge on the last group of nematode species is limited, and this may therefore lead to less optimal rotations (Nichol et al., 2011).

5.6.2. Grassland management

There are many different types of grassland from intensively to very extensively managed, and grasslands may be managed through cutting or grazing. This results in large differences in productivity and in how much organic matter is returned to the soil. Some of the primary threats to the grassland productivity are depletion of the vegetation through overgrazing or depletion of soil nutrients, which can be managed through controlling grazing pressure and adding fertilizers. There may also be changes in the vegetation structure over time.

Grasslands have a permanent crop cover that prevents wind and water erosion, as opposed to arable systems, orchards and vineyards. In addition, grasslands may also contribute to preventing landslides, although there are many contributing factors to landslide development (Metternicht et al., 2005).

Grasslands have considerably higher soil carbons stocks than arable soils (Soussana et al., 2004). This is primarily a consequence of higher belowground carbon inputs. Enhanced soil carbon contents can be achieved by converting arable land to grassland or through grassland improvement (reseeding or avoiding overgrazing) that enhance grassland productivity.

The greater inputs of root derived organic matter in grasslands fosters a higher microbial activity compared to arable soils. Earthworms are particularly favored under permanent (and temporary) grassland, where disturbance is limited and food is more abundant (Curry, 2004).

5.6.3. Organic farming

Organic farming prohibits the use of chemical fertilisers and pesticides and emphasizes the use of fertility building measures (Mäder et al., 2002). Crop rotation is the central tool that integrates the maintenance and development of soil fertility in organic systems (Watson et al., 2002). Nutrient supply to crops depends on the use of legumes to add nitrogen through biological nitrogen fixation to the system and limited inputs of supplementary nutrients, added in acceptable forms. Manures and crop residues should be managed to recycle nutrients around the farm. In comparison to conventional farming, organic farming relies on a long-term integrated approach rather than the more short-term targeted solutions common in conventional agriculture.

The effects of organic farming on soil threats depend on the actual implementation of the many different management options. A meta-analysis of long-term paired experiments of organic and conventional systems showed that organically farmed soils had higher soil organic carbon contents of 3.5 Mg C/ha. This seemed related to greater use of manures and composts in organic farming as well as crop rotations with a higher proportion of fertility building crops, such as grasslands. The greater carbon inputs and soil carbon stocks contributes to greater soil biodiversity in organic farming (Mäder et al., 2002). These and other ecosystem service claims of organic farming were recently scrutinized by Seufert and Romankutty (2017). They showed that the validity of claims is very context dependent.

Organic farming often claims to find itself in a better position to deliver ecosystem services than conventional farming (e.g. Mäder et al., 2002; Sandhu et al., 2010; Reganold and Wachter, 2016). Organic farming takes for instance a responsibility, at least intentionally, for the recycling of 'wastes', in spite of their more complicated management. Instead,

conventional farming may take the road of the least resistance by resorting to a much greater extent to mineral fertilizers and feed concentrates, leaving the processing of 'wastes' to other societal sectors whilst externalising the emissions associated with the production of mineral fertilizers and feed concentrates. In their review Hole et al. (2005) also confirm that on-farm biodiversity is generally better served by organic farming. This is, however, partly resulting from measures that have little if anything to do with refraining from pesticides or mineral fertilizers. Nature conservation areas, for instance, are often located on marginal land and their managers may be inclined to promote organic farming within and along the borders of these areas to protect them against negative effects from outside. The outcomes will in that case be the combined result of the organic farming practices themselves and surrounding areas (Winqvist et al., 2012).

Numerous dilemmas, many of them also pertaining to measures that are advocated or refrained from in organic farming, have been addressed in preceding sections of this report. In any case, it is evident that crop productivity of organic farming is lower, affecting at least that specific ecosystem service. De Ponti et al. (2012) estimate that yields are on average 20% less. This number would even drop below values of 30% if the required additional hectares for biological nitrogen fixation ('green manuring') would also be factored in (Schröder, 2014). This illustrates that there is at least a trade-off between in-farm biodiversity and the remaining land for off-farm biodiversity if the same volume of crops is aspired (Fischer et al., 2014). These lower yields must also be considered when emissions with impacts beyond just the local scale are considered such as greenhouse gases, ammonia and nitrate. These may be lower in organic farms on a per unit area basis, but not necessarily on a per unit produce basis (Kirchmann and Bergström, 2001; Stopes et al., 2002; Knudsen et al., 2014; Reganold and Wachter, 2016; Seufert and Ramankutty, 2017).

5.6.4. Conservation agriculture

Several of the cropping systems may make use of combinations of the management practices mentioned above. One such option is the use of conservation agriculture (Pittelkow et al., 2015), which is characterized by three principles: 1) Continuous minimum soil disturbance (minimum tillage), 2) Permanent organic soil cover (crop residues, mulches and cover crops); and 3) Diversification of crops grown (crop rotations, intercrops). This practice can save time, labor and fuel inputs while enhancing soil fertility compared to conventional farming. Compared to use of no-tillage applied without other measures, conservation agriculture provides higher crop yields, and they are on average only 2.5% lower than yields in conventional agriculture (Pittelkow et al., 2015).

Conservation agriculture is less adopted in Europe compared to other comparable world regions (Lahmar, 2010). These practices have been less promoted and researched in Europe. Conservation agriculture is not equally suitable for all the European agroecosystems. There is therefore need for better information on how conservation agriculture can help to improve soil and water conservation in the local conditions and which cultivation practices need to be sequentially adopted to improve ecological and socio-economic sustainability of the cropping systems. Priority would therefore be to define which regions in Europe are the most suitable for conservation agriculture taking into account climate and soil constraints, length of growing period, water availability and quality, erosion hazards and farming conditions.

Soil erosion by wind and water is strongly related to land use (Bakker et al., 2008). Farming systems that have a less intensive use of the land (fewer arable crops) and less bare soil will thus result in lower erosion and less sediment transport. Conservation

agriculture emphasized all of these aspects and therefore provides effective protection against both wind and water erosion. The measures applied in conservation agriculture that promote soil cover and accumulation of surface organic matter is also effective in enhancing water infiltration with positive effects on water buffering.

A layer of mulch on the soil surface favors larger surface-living degrader species. Thus, in conservation agriculture fields, large populations of predators are present when the pests arrive in spring, and biological control will therefore start earlier and be more efficient (Tamburini et al., 2016). Reduced tillage and mulching also have positive effects on large, anecic earthworms, which burrow very deep and improve soil infiltration and structure (van Schaik et al., 2014).

6. ELEMENTS RELEVANT TO THE DESIGN OF SOIL POLICIES

6.1. Introduction

We set out to describe soils and their distribution in the EU; their processes/functions and the role of these in providing services and the threats jeopardizing these services. Subsequently, we discussed how soil management options (practices) can help preserve soils and safeguard/promote their services. This last chapter summarizes elements for designing effective policy approaches towards such preservation of soils and promotion of services. This means that we will not restrict our analysis to just the preservation of the state of soil, but shall also cover the somewhat broader domain of soil management with its impacts on soil-supported services.

The limited scope of our study does not allow presenting a comprehensive overview or typology of existing policy instruments or measures. The recent thorough inventory and analysis by Frelih-Larsen et al. (2016) should suffice for that purpose, and we will include some conclusions from that and related work. Here, we will rather list a number of principles and features that we believe emerge from the body of literature covered in previous chapters, with a relevance to policy design. In structuring this information, our driving questions are:

- (1) (why) can't we expect farmers to take proper care of their soils, such fundamental resource in their private business? (Section 6.2);
- (2) if public intervention is warranted, what makes the EU level (the most) appropriate, in view of the Subsidiarity Principle? (Section 6.3);
- (3) which elements/components do we regard as essential for policies to be effective in the protection of soils and their services? (Section 6.4).

6.2. Soil as production factor in farming

Foremost among soil services, in a context of agriculture, is the provision of crop biomass for food and other purposes. This service has, obviously, both private and public dimensions (farmer livelihood; food security). For the moment, we focus here on the private (business) dimension. Among the many soil services, crop production is perhaps the single easiest-to-quantify service. Yield and yield quality are readily observed and annually accounted for in farming practice. Farming as a business optimises resource use for profit with a view on both short term survival and long term sustenance. Its production potential is largely defined by biophysical factors on the one hand (climate, soil, landscape/topography, hydrology, biotic pressures, etc.), and the rule of economy on the other. Land, with its important dimension of soil quality, is a private good valued in financial terms as rent and sales prices (Kilian et al., 2008; Feichtinger and Salhofer, 2013). Farmers seem well aware of the vital importance of soil condition to their business, as Pronk et al. (2015) found in a survey among 2500 respondent farmers from across 24 farm types in selected EU MS. If land/soil is a private asset and the farmer an entrepreneur, isn't proper soil care in the farmers' own interest, that is, without needing intervention? There are several reasons why the answer is not necessarily confirmative. (As stated and for the sake of argument, we narrow down the concept of 'proper care' to just 'what is good for yield or profit', be it from short or long term perspective; and will deal separately with other services and the public interest.)

Soils – their state and ability to support crop production – generally respond slowly to changes in management practices. Even given sufficient time, soil properties can only be improved within certain limits set by climate and parent material. Effects of soil improvements on crop yields remain often unclear or small. In farmers' fields, inter-annual (weather-related) variation in attainable yield, the timing of operations and the use of inputs all have relatively strong effects on actual yield and produce quality, and can easily mask benefits from improved soil conditions, even in the long run. The gradual improvement of crop genotypic performance, too, may mask slow soil degradation. Responses (to management) of soil properties that are considered relevant for crop growth, indeed, are more readily reflected in indicator values (e.g. penetrometer resistance; infiltration capacity; moisture retention; disease suppressiveness; earthworm abundance) than in crop yield itself. The low responsiveness of yield to management-induced soil improvements, we believe, is a major reason for farmers to avoid practices that may improve/protect soil but bring some disadvantage (see below) relative to current management. Research, too, has a difficulty generalising the yield benefits from soil improvement by specific practices. Outcomes vary with local factors whose impacts are hard to quantify. While a practice can have significant effect within a trial, meta-analyses aiming to generalise effects or to explain differentiated responses across different environments (soil, climate) meet with only limited success (Hijbeek et al., 2017; Zavattaro et al., 2014). Low responsiveness of yield to soil condition may also be a matter of perception: if benefits or problems show only under extreme conditions (of drought; waterlogging) or go unnoticed as long as critical thresholds are not crossed (disease suppressiveness), economic reasoning may be impaired by partial understanding. To some extent, this holds also for evidence from controlled trials where soils are rarely selected for their poor condition, and the occurrence of weather extremes (rare by definition) relies on natural variation, as in farmers' fields. (Quite contrary to the above, crop yields do respond strongly to one particular aspect of soil quality: chemical soil fertility. We ignored this aspect as – in most parts of Europe – it is of little relevance with regard to soil preservation, and low fertility is easily corrected where needed. Farmers know its importance only too well, to the extent that legislation was to be implemented throughout EU MS to constrain nutrient inputs.)

Other reasons why farmers do not necessarily share public concern (including that expressed by policy makers and researchers) about certain threats may include, in our view, that (i) such threats are simply not present or not readily noticeable on their farm; (ii) they are unaware of existing solutions to address the problem; (iii) no ready-for-practice solutions really exist; (iv) farmers can to some extent correct for soil-induced crop stresses, by using extra inputs (fertilisers; crop protection agents) or alternative equipment (deep-plow; irrigation) – thus largely externalising the cost of such 'repair measures' if their impacts (emissions; lowering groundwater table; loss of biodiversity) are mostly felt elsewhere.

In spite of all these difficulties in recognising benefits of practice-induced soil condition for crop yield, and to balance the above statements, it is only fair to say that substantial evidence exists – in history as in current times, in Europe and elsewhere – that massive degradation of soils may, in the end, render land totally unsuited to farming. The question – in this section – still is whether we can rely on market mechanisms to prevent such degradation. Economic success is measured in net revenue, not crop yield. Where yield benefits from soil improving practices are uncertain or small, their absence under 'conventional' management can easily be outweighed by gains in the productivity of other factors – e.g. labour – or by cost saving, at least in the short run. But is it also true in the long run? If the soil's condition for crop production was properly expressed in land rent or

sale prices, private interest would limit the decline of the soil's production capacity. Limits would be set by balancing land depreciation against gains in productivity of non-land factors. However, we have no evidence for that prerequisite (practice-induced soil condition expressed in price) to apply in practice, and we doubt that current knowledge could provide a sound basis for such pricing. So, if soil degradation is not prevented or contained by market forces, its cost (land productivity lost) is externalised and moved to future generations of farmers. A 'race to the bottom' (Cumberland, 1981) allowed by market failure is, obviously, to the detriment of public interest (food security), but also to the collective interest of the farming community itself. On shorter time scales, the transmission of soil borne diseases (if left unchecked due to data privacy) and wind erosion (damaging neighboring cropped fields) are other examples where the immediate interest of the individual farmer can be at conflict with of the farming sector at large. Other threats to food production cannot be countered by the individual farmer (e.g. salinization; desertification; land slides and flooding) and therefore call for collective action.

Finally, a decline in productivity of agricultural land would also imply that additional land is needed to meet food demand. The consequent reclamation of marginal and pristine lands goes at the expense of their (non-food) ecosystem services, again a public good. We conclude that the protection of farmland productivity is of collective farmer interest and wider public interest for all of the above reasons.

6.3. Justification for EU level action

The above goes to show that, even if soils provided no other services than crop production for farm income and public food security, their fate cannot likely be left to market forces. Soils are our focus here, but this conclusion obviously extends to safeguarding the wider concept of food security itself, which has driven CAP from its inception in 1962, in spite of its various reforms.

In reality soils provide many other public services besides food security. As long as the cost of their protection is not expressed in consumer prices, safeguarding these services is a matter of public concern. We list them again, ignoring details and without being exhaustive: the regulation of water, carbon and nutrient cycles; and the provision of biological habitat above and below ground. These services are difficult to quantify, more so in economic terms. Nevertheless, their significance to the wider society is evident where mismanagement causes floods, siltation of water ways and infrastructures, widespread decline of biodiversity, pollution of water and air, and emission of GHG driving climate change.

The Subsidiarity Principle stipulates that actions at EU level are only allowed in situations where policy objectives cannot be sufficiently achieved through MS actions (Revesz, 1997). We see various justifications for tackling soil threats at the EU rather than the national level. (i) Biophysical impacts of threats do not stop at MS borders. (ii) Economic impacts don't either. (iii) The EU level allows the spatial optimisation of interventions and the sharing of their costs; (iv) Efficiency of research and innovation efforts; and (v) Local democracy may hamper the implementation of unpopular measures, if these lack underpinning at higher level.

Ad.(i). River systems carry pollutants (nutrients, biocides, pharmaceuticals) and sediments over long distances across national borders and into coastal waters, affecting ecosystems along their way. Siltation of riverbeds, as well as deficient water retention on slopes cause

downstream flooding. Fine soil particles are carried by winds across northern plains. Ammonia lost from manured and fertilised fields causes eutrophication and acidifies soils upon deposition (NEC Directive 2001/81/EC). Biodiversity in soils may be a local stake, but farmland and meadow birds and insects migrate over long distances and depend on habitats for feeding and breeding, including on land under agriculture. Plant pathogens are transported in traded produce and equipment, some are carried by wind. Causes of desertification are complex but the process is not likely just local and extends across the Mediterranean basin. Declining soil carbon stocks contribute to climate change. (Food security was already listed as an obvious public issue of EU or global dimension.)

Ad. (ii). A common market calls for uniform rules to prevent displacement of polluting or other activities with externalised cost. This can lead to a “race to the bottom” (Cumberland, 1981), where local resources are sacrificed for short term outcomes, in a context of competition among regions or MS.

Ad.(iii). Semi-natural habitats or lands under low-intensity agriculture can be rich in biodiversity. These are a collective heritage deserving of collective protection. Public funds for biodiversity conservation are likely better spent in those areas than in low-diversity areas. Another example is the conservation of the large carbon stocks contained in peatlands and wetlands. Protecting these is a far more effective climate mitigation option than attempts to increase the SOC stocks elsewhere. Cost for peatland protection would therefore be better spent than for C-sequestration elsewhere. This would call for redistribution of funds within the EU as organic soils are predominantly found at higher latitudes.

Ad.(iv). Current practices have evolved in a context set by economy and the local biophysical potential, and are highly tuned to fit these local conditions. Changing these involves addressing barriers, often of ecological / agro-technical nature (Pronk et al., 2015). This requires research and innovation and (independent) extension services. Public resources for these activities are better spent if widely felt bottlenecks are addressed (shared among MS), and the acquired knowledge is applied across wider geographic areas. This is the central concept in the EIP-AGRI Thematic Networks approach under the EC Horizon 2020 Program.

Ad. (v). Protection of soils and their services, where it implies the introduction of obligation or constraints, is likely to meet locally with skepticism and resistance as illustrated by the rejection in 2012 of the proposed GAEC for the protection of organic and wetland soils in various MS, and the withdrawal of the Soil Framework Directive by the Commission. Whereas these examples illustrate the difficulty of setting a frame at EU level, it also underlines the strong force of lobby groups in the MS themselves. Without a centralised underpinning, MS will face difficulties pursuing effective protective measures where they are at conflict with local short term interests. This was exactly the reason why the European Commission recently developed a detailed reference document to which individual MS Action Plans required by the EU Nitrates Directive were exposed before their approval (EC, 2011).

6.4. Elements for policies

6.4.1. Introduction

Below we will return to some aspects treated in previous chapters, now with a more explicit view on implications for policies. The first question is whether soils-oriented

policies should address the preservation of the soil itself, or rather promote the concept of sustainable soil management which targets soil services and the wider goals they support. Before entering into the more complex balancing of pros and cons of practices, we then propose a few 'no-regret' priorities. Next, we present a brief overview of practices and processes and how these can mitigate – or aggravate – the major soil threats. This section also highlights factors that limit the adoption of certain practices often regarded as 'soil-improving practices'. Also, we aim to derive implications – of the material collected in this study – for monitoring to assess effectiveness of practices and compliance of actors in soil preservation programs. Finally, we will discuss the need for agricultural knowledge and innovation systems that address adoption barriers.

6.4.2. Should policies target soils or soil functions?

For policies to be effective in safeguarding soil services, they must recognise that (i) some services more or less coincide with the soil's condition itself (C-sequestration; soil biodiversity); (ii) other services rely on the soil's condition to support processes underlying the service (food security; regulation of water cycle); and still others (iii) are promoted or hampered directly by soil management rather than via the soil's condition (provision of clean water; aboveground biodiversity)³. Obviously, this separation cannot be very strict, if only because states and processes are intricately linked. Moreover, some services can be placed in all three categories (e.g. climate regulation). Nevertheless, our point is that a comprehensive soils policy should not just focus on preserving/improving the (physical, biological, chemical) condition of soils, but rather on enhancing the services they support. This is not only so because of possible conflicts (practices promoting soil condition but hampering a service; or promoting one service while hampering another) but also because the monitoring of the services themselves is often easier than the monitoring of soil properties that support them. A focus on services also facilitates the debate among stakeholders. While expressing the soil's state requires jargon, the necessity to prevent flooding, landslides, climate change and decline of biodiversity is immediately understood by all parties. (See Section 6.4.6. for a discussion on scales with respect to enforcement and monitoring, respectively.)

6.4.3. 'No regret' actions

The simple truth is that productive / intensive systems are not generally rich in biodiversity. Among the other services, crop production prevails. The improvement of soil management in such systems is usually associated with economical and technical challenges and with uncertain responses to the proposed practices (Chapter 5; and Section 6.4.4). In contrast, extensive systems are valued for the services they already provide. All that these areas need is full protection, rather than improved practices. Their services include habitat for biodiversity, regulation of the water cycle, and C sequestration/retention. Peatlands, wetlands, highlands with soils rich in organic material in northern countries, but also the extensively grazed agro-forestry systems of southern Europe ('dehesa') and similar systems in Eastern Europe all host rich biodiversity as well as large carbon stocks. Their conservation should be of primary concern, both from a climate mitigation and biodiversity conservation perspective. The same holds for upland areas in mountain regions in various parts of Europe. Where such systems support

³ Examples of such direct impacts on services are the emission of herbicides, nutrients and other agri-chemicals to groundwater and surface water bodies; ammonia emission to the atmosphere and the associated deposition elsewhere, generating acidification and eutrophication; the emission of the GHG nitrous oxide as affected by tillage, organic inputs, and mineral fertilisers; aboveground biodiversity as affected by crop rotation, weed control and crop residue management; net farm revenue as affected by input savings e.g. fuel. While this is a mixed list, they share the characteristic that the impact on the service is more obvious than the impact on the soil, and that the impact does not rely on the changing of soil properties.

extensive agriculture, their biodiversity value is associated with the agricultural activity and the challenge here is to sustain a level of exploitation that is no longer currently profitable.

The amount of carbon stored in drained peat soils under agriculture (3.6 M ha) can be roughly estimated at 2,000 to 2,800 Mt C. Losing this C-stock to oxidation and re-capturing it in the (99%) mineral agricultural soils in EU-28 would imply an increment of 17 to 23 t C/ha on average across the EU-28 (based on Chapter 4; mean depth of organic strata in peat soils under agriculture assumed equal to peat soils elsewhere; grassland and rough grazing excluded from calculation). This seems highly unrealistic. Even in cool regions (favorable for grassland as well as for SOC accumulation) annual C-sequestration attained by converting arable land to grassland was estimated at only 1 t C/ha/y (Taghizadeh-Toosi and Olesen, 2016 for Denmark). Moreover, such 'C-translocation' would involve nitrous oxide emissions: first from peat degradation, and then again from sequestration elsewhere (see Bos et al. (2016) for examples of tradeoffs in C-sequestration on arable land). In conclusion: pristine peatlands must remain untouched and degraded areas restored where possible.

One potential tragedy for the future of the soil in the EU is the risk of the decline of the wood-pasture ecosystem of some European regions. As defined by Garrido et al. (2017), wood-pastures are natural areas that combine scattered trees with pasture grazed by animals like pigs, sheep or cows, and that are locally known as *montado* (Portugal) or *dehesa* (Spain), although they occur in many other countries (e.g. Beaufoy 2014). The wood-pastures enhance biodiversity and ecosystem services as Torralba et al. (2016) have recently written. The difficult situation of the wood-pastures farmers, with stable or decreasing prices for their products while the inputs costs are increasing, is that CAP payment schemes are insufficient, or even counter-productive, as Garrido et al. (2017) conclude from their meta-analysis in the Iberian *dehesas*. The direct payments of the CAP are not applicable to the whole surface area of the farm: at least in some member states, the surface area under the tree canopies is not eligible. (Consistent reasoning could support the view that in, for example, olive orchards the area *not* covered by the tree canopy should *not* be eligible.) The current approach ignores the many benefits derived from the presence of trees, including the protection of soil and its associated services. Torralba et al. (2016) suggested that member states could modify this procedure, but are reluctant to do so. Alternatively, Gaspar et al. (2016) proposed a dedicated payment scheme for the specific services of these ecosystems. Active policies are needed to protect and retain these highly valuable systems in the EU with their soils. These systems face destruction due to short term revenues gained from removing the trees, with the indirect support of the CAP. (Similarly, threats to wood pastures arise from subsidies for afforestation that outweigh current revenues from wood pastures.)

Another 'no regret' action would be the strict protection against land-grabbing of productive farmland. As growing global food demand in coming decades already calls for intensification of all suitable land in current use (Buckwell et al., 2014; Van Ittersum et al., 2016), and loss of productive land is therefore simply unaffordable. Such loss would call for accelerated intensification of remaining farmland, and would increase pressure on extensive systems (e.g., those mentioned above), affecting soil services in both. Ironically, the best soils are often first to disappear under urbanisation, as soils near original habitation have been historically better amended.

Finally, the avoidance of pollution and contamination of soils appears as a rather straightforward action. This aspect seems well covered by the current revision of the Fertilisers Regulation and requires no discussion here.

6.4.4. Some agro-ecological considerations

Although the suitability of any given practice to preserve soils and their services largely depends on local conditions (climate, soil type, topography, farm type, crop type, etc.), and irrespective of the compatibility of practices with farm logistics and economics, there is little doubt about the general validity of the following principles:

- **Soil cover** - in the form of crops, crop residues, cover crops and mulches - protects the soil from the impact of rain; reduces slaking, runoff and water erosion; protects against wind erosion; and provides shelter and sometimes food for aboveground biodiversity; catch-crops can also reduce off-season nitrate leaching, and so increase farm-level nitrogen use efficiency.
- **Fresh organic material**, irrespective of its source - crop residues, manures and slurries, composts, cover crops, grass leys and green manures, sewage sludge, digestates - is the main food source for soil life. Its regular supply promotes bioturbation/burrowing and aggregation (by secretion, excreta, hyphen) and these processes help to build soil structure which promotes water infiltration, retention and drainage; for aeration and root penetration; and for the formation of micro-habitats. Soil structure favors the exploitation of the soil profile by roots, soil workability, seedbed quality, and it may render the soil-crop system more resilient in the face of extremes (drought, waterlogging, soil borne diseases). Besides, the organic material resulting from humification provides some buffering of nutrients and pH.
- **Soil disturbance** (tillage) has a negative impact on soil macro-fauna such as earthworms, which are relevant for soil structure and associated hydraulic properties. (D'Hose et al., 2014).
- High frequency of **host crop species** and the occurrence of specific cropping sequences promote the build-up of soil borne pathogen populations which may cause crop damage
- Heavy **traffic load and high passing frequency** of farm machinery causes damage to soil structure also in deeper layers, which is very difficult to restore. While the process is hard to trace over time, reduced crop stand on headlands relative to main fields illustrates its common occurrence. Damaged soil structure increases the risk of ponding and waterlogging in lowlands, and increases runoff and erosion risks in uplands, causing negative downstream effects (flooding, siltation, pollution).

Practices that promote the above positive mechanisms, or avoid the negative ones, are often referred to as 'soil improving practices', 'recommended management practices', or 'best management practices'. None of these qualifications expresses a general truth: all such practices come with drawbacks, either for soil services or for soil condition itself. Anyway, practices often grouped under those denominators include crop rotation (as opposed to monoculture), forms of reduced tillage (minimum, shallow, non-inversion tillage; versus mouldboard ploughing), the use organic manures (versus mineral fertilisers), the retention of crop residues (versus removal), and the use of catch and cover crops and green manures (versus bare fallow in between main crop seasons). More radical deviations from conventional practices are, for example, the introduction of a ley phase (one or more years) in the arable rotation, and the use of guided traffic ('tram-lines') for

wheels to pass always over established tracks (sometimes practiced in organic vegetable systems).

Below we briefly address the challenges and trade-offs that come with some of the soil-improving practices or with more sweeping changes. Hereby we aim to stress the crucial role that technological innovation must play in successful attempts towards soil preservation. This is not to say that legislation is unnecessary. To the contrary: mandatory requirements, if realistic, can be effective in promoting the development and adoption of innovations previously considered impractical or unnecessary.

Crop rotation

The rotation of crop species is believed to promote soil biodiversity, and helps to keep the pressure of soil borne diseases low. In specialised arable systems (with relatively large fraction of potatoes, beets, and/or vegetables), cereals are usually grown for this purpose. At the same time, cereals contribute more to the maintenance of SOC stocks than, for example, root crops; and they allow for the cultivation of catch or cover crops due to their early harvest. The limited revenues from cereals, however, make this practice costly (e.g., Bos et al., 2016). Often, new crops also require new equipment or machinery. Increasing the variety of crops on farm or in a region may also call for changes in infrastructure (collecting; processing). As such changes are difficult to implement at the single farm level, landscape/regional approaches seem more promising and may need coordinated policies. The adoption of more diverse crop rotations is often hampered by the specialisation of farms or even regions in specific production systems that tend to favour cultivation of specific crops at higher cropping frequencies than beneficial for soil health. Such effects may partly be overcome through development and enforcement by the food industry of good agro-ecological practices at the crop rotation level.

Tillage

Conventional tillage by mouldboard plough is a repair measure to help re-set soil structure after damage by traffic (harvest). It also mixes nutrients and organic matter through the root zone, and is used to bury crop residues, weeds and their seeds, and to prepare good conditions for germination and crop establishment. These benefits are compromised in no-tillage or reduced tillage systems. Sometimes at the cost of yield loss, but this may be balanced by cost savings. In the context of an increase in the size of farm units, reduced tillage has become popular in several EU regions, primarily because of its lower fuel, labour, and sometimes equipment costs (FAO, 201; Ingram, 2010; Morris et al., 2010). Except for saving fuel, there is little evidence for reduced greenhouse gases from no-tillage systems. Its introduction has triggered a massive increase in the use of herbicides, which may cause environmental problems (e.g., see Prosser et al. (2016) and Soloneski et al. (2016) for impacts on amphibians). Crop residues left on the field may promote fungal diseases. The absence of topsoil inversion promotes near-surface accumulation of soil organic matter as well as immobile plant nutrients, notably phosphorus, which may cause larger nutrient loss by runoff and consequent surface water loading. Coarse seedbed structure constitutes limitations for fine-seeded crops; and root crops may suffer from denser topsoil structure. All these limitations call for innovations. Most urgent among these, it seems, are approaches to weed control that do not rely on herbicides.

The effects of tillage and the benefits and problems associated with no-tillage systems vary strongly with climate and soil conditions. Guzmán et al. (2014) thus made specific observations for the Mediterranean region, showing large differences between arable and permanent crop systems. In arable systems, maintaining the cereal stubble during summer and early autumn leaves the large surface cracks undisturbed, allowing for rapid

water infiltration during the first autumn rain events; it also reduces soil losses by water erosion, and reduces evaporative water loss (directly from the soil) until the crop canopy covers the surface. In dry years (annual rainfall below 400 mm), yield is usually greater under no-tillage than under conventional tillage (as confirmed by Pittelkow et al. (2015) for regions with high aridity index). In contrast, no-tillage in permanent (tree) crops enhances runoff and erosion due to compaction, especially during years with frequent rain events. Combining cover plants - cover crops, weeds in between trees - with occasional tillage to alleviate compaction may be an effective measure to protect the soil and conserve water in such systems. This shows that use of less intensive tillage (no tillage) requires local fine-tuning of the entire cropping system (e.g., breeding and selection of cover crops with matching properties; limiting surface load by machinery and transport; proper repair tillage).

Organic manures

The advantages of manures for a range of soil quality indicators are obvious. As addressed in Chapter 4, manures come with disadvantages, too. Nitrogen losses per unit crop nitrogen uptake are usually larger than from inorganic fertilisers. Depending on the reference fertiliser type, this applies to nitrate leaching as well as ammonia volatilisation and denitrification, including for the emissions of nitrous oxide. At the same nitrogen input rate, manures sustain lower crop yield than mineral fertilisers. In spite of this, manures - as a natural outcome of animal production - must of course be utilised to their full potential (Schröder, 2016). While excess manure is available in certain regions, arable farmers in large parts of the EU have limited access to animal manures, as a result of regional specialisation in livestock systems. As long as manure processing costs, to reduce water content or refine nutrients for cheaper transportation, are not borne by the consumer of animal products or otherwise, wider geographical spreading of manures will remain frustrated. Another main source of external organic inputs to arable systems is compost. As any organic product, it brings benefits to soil quality. It is, however, available in limited amounts only. Using all potential sources in the EU will not likely change that (see Chapter 4). Soil carbon accumulation based on manure or compost application brings no net climate benefit (soil carbon sequestration), unless it replaces incineration (Chapter 4). In addition, care should be taken to avoid greenhouse gas emissions of methane (from biogas facilities and manure storage tanks) and of nitrous oxide (from storage or solid manures and field application of manures and composts).

Crop residues

The retention of crop residues contributes to the maintenance of SOC stocks, and may provide shelter for biodiversity, and protect the soil from erosion by wind and water. Standing cereal stubbles, in absence of weed control, also provide a habitat for wild plants supplying food for insects and farmland birds. Whereas residues from fresh harvested crops (vegetables; beet tops) decompose quickly and release nitrogen from the start due to their low C/N ratio, the decomposition of cereal straw (high C/N) takes more time and initially locks up nitrogen which can both be viewed as a service and as a constraint (Heijboer et al., 2016). Keeping residues for soil improvement implies – in the case of straw – income foregone, and brings a climate penalty (missed opportunity for energy from biomass to replace fossil fuel; enhanced N₂O emission). Removal is not always at the cost of SOC stocks on a wider scale, as material used for animal bedding will usually return to the soil, at some point in time and space.

Apart from these short-term economic and climate penalties, farmers often report increased disease pressure (fungi) and increased use of fungicides (Pronk et al., 2015). In some case, antagonisms (soil suppressiveness) may develop which limits this problem in

the longer run (as in the case of the fungus *Gaeumannomyces graminis* var. *tritici*, or 'Take all', that infests cereal roots). Resistance breeding in cereals can help reduce disease pressure as a barrier against straw incorporation. Additional but minor downsides of straw retention are the need for incorporation (traffic, fuel, soil disturbance), although these needs may be reduced by proper crop rotations.

Catch and cover crops and green manures

Catch and cover crops and green manures (CCCGM) are grown in between main crop seasons to protect and improve the soil. They can also reduce nitrate leaching and increase nitrogen use efficiency at farm level, if the nitrogen thus saved is subtracted from the regular fertiliser input to match a given target yield. Moreover - and contrary to manures, compost and crop residues - the introduction of CCCGM implies an increase in net primary production, and so represents net soil carbon sequestration from the atmosphere. The attainable acreage of CCCGM is largely constrained by climate: low temperature after the main crop harvest sets limits to plant growth in northern countries, while CCCGM compete with the main crop for soil water stocks in southern countries. Breeding and selection of CCCGM species hold a potential to shift some of the boundaries, e.g. by increasing cold tolerance during initial growth, or by developing varieties that quickly develop a surface cover and then halt further growth, thus preserving soil water. Breeding should also aim for low host-status of CCCGM crops in view of plant pathogens (notably nematodes). Root crops and silage maize are generally harvested late, which prevents the good establishment of CCCGM crops in temperate climates. Under these conditions, farmers also fear (further) damage to soil structure by traffic. The latter can be avoided by underseeding of the CCCGM during the main season, which is successfully applied in maize and cereal crops on certain soils, but requires tinkering to make it work properly. This may involve, again, breeding and mechanisation efforts to improve soil management and cropping systems.

Grassland and land use changes

The inclusion of a ley phase – ryegrass, grass-clover, lucerne, etc. – is among the most effective options to maintain or build up soil organic matter in arable systems. Mixed leys can bring other services, too, including the promotion of biodiversity, and soil suppressiveness against crop pathogens. The advantages of leys for soil fertility in arable rotations, along with potential gains for the livestock component, have sparked a renewed interest in mixed systems. In practice, the economy of scale and specialisation seem to hamper such revival, at least at the farm scale (Regan et al., 2017). Alternatively, the advantages can be sought by mixing farms at the landscape/regional scale. Without an increased demand for forage, however, to accompany such development, this would be at the cost of the (farmed) permanent grassland acreage. Developments in processing technology may help avoid this consequence, if food for human consumption can be produced from pasture by protein refinement making the current ruminant's role less essential. At the moment, protein recovery rates in such processes are still too low.

Current agricultural policies strongly emphasize the preservation of permanent grasslands. This is absolutely valid from the perspective of soil carbon sequestration, GHG emissions, nutrient leaching and biodiversity conservation. However, such policies implicitly lead to the freezing of present land use, whereas alternative shares of grassland and arable land may better serve multiple services, including primary production. After all, the existing share of grassland, including of High Nature Value grasslands, is to some extent just a reflection of current (or past) dietary preferences, and/or reflects (partly) outdated practices. Even without changes of the human diet, the same volume of meat and milk can probably be produced on a smaller acreage of grassland. Such intensification would free

land area for other services, including for the re-creation of wilderness with appreciated biodiversity, albeit in alternative forms. Obviously, this issue pertains to the much broader debate on 'sharing versus sparing' (Kremen, 2015). Moreover, should future diets change towards less animal products - for the benefit of the environment and human health - then this would likely imply a reduced demand for a vegetation that can only be sublimated by ruminants. Why then would there be an interest in freezing land use?

Wrap up of agro-ecological effects

To wrap up this section, we conclude that 'win-wins' are rare in this domain. All of the so-called 'soil-improving practices' have their trade-offs. The side-effects make that 'what is good for the soil' - by any chosen measure - is not necessarily good for climate (fossil fuel use; fossil fuel replacement foregone; nitrous oxide emission), environment and biodiversity (herbicides; fungicides; nutrient emissions), or farm economy (yield loss; labor; equipment costs). Also, what is good for one aspect of soil (e.g. SOC stocks), is not necessarily good for other aspects (e.g. soil structure). In spite of these trade-offs, we confirm here that all of the above practices can contribute to the protection and improvement of soils, of their state and functioning and of the services they provide. The main challenge for policies is to choose instruments that recognise these trade-offs, and induce or enforce innovations that aim to resolve them. At the same time, proven solutions must be made widely available, both in terms of access to knowledge and availability of tools and products on the market. Many of these solutions need to be adapted to the local situations, and doing so requires innovation within in cropping systems, mechanization and crop and cover crop genetics.

Whereas the above comments relate to merely practices, their implications extend to the whole farming systems level, too. Many soil threats are expressions of a development that we tend to characterize as 'intensification', but its opposite - 'extensification', including approaches known as organic agriculture - is an unfocussed and likely inadequate answer to counter these threats. Instead, specific and probably pedoclimatically tuned measures are needed with respect to organic matter management, machinery weight specifications, hydrological management, the use of agro-chemicals, and landscaping are needed. Still one further level up, a debate is timely on the tendency observed across the wider agro-economy sector: that of specialization. The introduction of improved agroecological practices that favour higher diversity in the cropping system is made difficult by specialisation at farm and regional scales. Policies should, therefore, aim to deal with the drivers of this specialization.

6.4.6. Implications for monitoring and enforcement

Soil differs markedly from other compartments of the biosphere (atmosphere, water bodies) in that it is hardly subject to mixing, while its key processes and governing properties are highly heterogeneous in space due to variation in parent material, exposure and topography, climate and management history. Heterogeneity and absence of mixing imply that the monitoring of soil condition requires high sampling density. The measurement of indicators is usually expensive, time consuming or difficult, and prone to large sampling and measurement error. Interpretation of indicator values observed is complex, and usually requires local calibration to determine which range is acceptable or desirable, and to what level of service such range corresponds. Among the rare cases where agricultural soils are monitored in a mandatory frame - to assess farmer compliance with regulations - are the campaigns of residual (autumn) soil nitrate-N monitoring in Flanders and Baden-Württemberg. They can serve to illustrate the great legal and technical difficulties associated with such monitoring. In this study, we have avoided the enormous body of literature on soil quality indicators used in Europe and worldwide, but

good overviews were provided by the ENVASSO (Huber et al., 2008), RECARE (Stolte et al., 2016), and several other initiatives.

Besides spatial heterogeneity, a feature of soils that complicates the monitoring of their condition is 'resilience': soils tend to gravitate towards equilibria dictated by long term means of external conditions. This, combined with the above characteristics, means that it is difficult to assess the evolution of their state over time, e.g. in response to a change in practice; let alone to verify compliance as a basis for assigning subsidies or penalties, which requires some measure of likelihood. Most soil indicators serve well to express how soil properties and processes respond to contrasted treatments (practices) in controlled trials, and to derive relations between the practice and the level of a service by inference (via modelling), if such relations cannot be established directly (i.e. by observing the change in service value in response to the practice; which is rarely possible). However, very few indicators that reflect a soil property or process are actually suitable for the monitoring of large surface areas as e.g. in farmers' fields, catchments and landscapes, in view of the above purpose (verification of compliance and impact). We therefore advocate that the services themselves be measured instead of soil properties, where possible.

We recommend that soil policies allow for the local ranking of priorities in terms of soil services in need of promotion (or threats in need of mitigation). Soil management must be optimised within the wider frame of sustainability goals, not just focussing on the preservation of the soil itself but rather on the enhancement of the services that are directly or indirectly affected by soil management (see Fig. 3).

Subsequently, a two-pronged approach could combine the promotion of selected practices with the monitoring at the catchment scale of services delivered and/or threats mitigated. Examples of services (or threats mitigated) that can be monitored at catchment scale are the regulation of hydrology (response time in water channels), mitigation of erosion (sediment load in channels), provision of clean water (surface water pollutant load; nutrients; biocides), and aboveground biodiversity in the landscape. Some services, however, require in-situ (per field) measurement for their assessment. These include soil biodiversity, botanical biodiversity, and C-sequestration. We regard it unlikely, however, that it is feasible to monitor these for enforcement or verification purposes.

It may in reality be more feasible to monitor the implementation of the practices, than to monitor the services they aim to provide. The above approach can include the obligation to present and regularly evaluate an integrated multi-annual soil management plan (ISMP) at the farm level, and should then include guidance to implement such plan (see Sections 6.4.7-6.4.8). The choice of practices to be promoted and monitored will need to be prioritised for the local conditions, and for the services in focus. The ISMP should cover all locally relevant aspects of soil management. This would be most usefully implemented in a context of collective action, actively encouraged by extension programs with stepped targets and communication on progress achieved. Compliance of farmers with proposed measures can be verified by remote sensing or other survey methods, which is far less complex than monitoring soil status indicators and comparing these against their target ranges – if such ranges could at all be set to match the service levels desired. The farmer can be held accountable for the implementation of practices, less so for the achievement of the wider goals (services improved).

All of this would require, of course, an a priori assessment of practices most likely to deliver the desired outcomes under the local conditions, and of possible solutions to local barriers that would hamper adoption.

A major advantage of the above approach is that it implicitly accounts for soil-mediated services *as well as* services/threats directly affected by the management practice itself, i.e., those not involving a change in soil condition or soil processes. It could be argued that policy measures are already in place to protect non-soil compartments of the environment (water, air, habitats) and that what we need most are specific measures targeting the protection of soil itself. We argue, however, that so many tradeoffs, synergies and other links exist between the goals of other environmental policies (promoting the quality of air, surface water and groundwater, protecting biodiversity, reducing biocide use, etc.) and those of soil protection, that policies can be more effective and coherent if they target soil protection as part of a well balanced set of locally prioritised environmental stakes. The goals would be pursued at the catchment / landscape scales, the compliance with policy measures at field/farm level.

Finally, for threats which cannot be countered by the individual farmer due to the scale of required measures (e.g. salinization, desertification, peatland degradation, land-slides) it is self-evident that the catchment/landscape scale is the only level permitting effective action.

6.4.7. Agricultural Knowledge and Innovation Systems (AKIS)

Many 'soil-improving practices' are as old as agriculture itself, and for some systems they have been well documented long ago (see for example King (1911), for an extensive treatise on the management of soil, nutrients, and organic matter in traditional farming systems of East Asia). Nevertheless, their re-introduction today, modulated or not, often comes with barriers that complicate proper integration into the farming system (Chapter 5; and Section 6.4.4). While the assessment of awareness or belief is admittedly a highly complex endeavor, evidence from the CATCH-C project strongly suggests that many farmers from across Europe are well aware of the soil benefits that certain practices may bring. At the same time, they have expressed a varied array of expected or experienced difficulties associated with the 'new' practices. Cited difficulties were sometimes shared between adopter and non-adopter farmers, but contrasting opinions between the two groups were also found (Bijttebier et al., 2014). Werner et al. (2016) introduced the concept of 'perceived difficulty', and proposed that efforts to promote alternative practices focus on diminishing this perceived difficulty. Whether difficulties are 'true' or 'just perceived', it appears that combining technological innovation with exchange of experiences and insights by practitioners will be vital to promote adoption of practices that come with extra cost or other difficulties. Paraphrasing the common interpretation of 'intensification', Buckwell et al. (2014) argue that sustainable intensification means more knowledge input and more 'services output' per ha: "Intensification of agriculture, especially in Europe, is not primarily about the use of more fertilisers, pesticides and machinery per hectare, but the development of much more knowledge intensive management of scarce resources to produce food outputs with minimal disturbance to the natural environment, and more environmental outputs too." This definitely applies also to the subdomain of soil management.

Some basic biophysical principles cannot be changed and the barriers defined by them are, in the long run, perhaps the most challenging (Section 6.4.4). Addressing these may require drastic changes in production systems. Other barriers more related to technology – including crop improvement – can be tackled by focused research and innovation efforts. Developments in aerial reconnaissance, computer vision and crop sensing, autonomous transport, GPS guidance, crop breeding and genome editing, and many other fields of technology may bring solutions to long-felt bottlenecks. Turning technological development towards resolving conflicts and trade-offs in soil management is, in our view, the most

promising track to progress. Improved methods for seeding, weed and pest control, harvesting and transport operations, and post-harvest processing of produce can all contribute, depending on local issues, to more sustainable soil management. And so can resistance breeding, for example to address fungal diseases in cereal systems where residues remain unburied overwinter.

Similarly, communication tools and access to existing knowledge for innovation are rapidly improving with the help of web semantics and linked open data infrastructure, decision support via hand-held devices, etc. All of these serve to support learning networks and advisory systems. For an overview of the roles ICT can play in agriculture and participatory innovation, we refer to EU SCAR (2015). We can only confirm that current R&D policy under Horizon-2020 and the EIP-AGRI - with its focus groups setting agenda's, thematic networks surveying bottlenecks and possible solutions, and operational groups connecting the various actor-types per supply chain around shared problems - has set out an ambitious programme deserving of continued support.

The desired evolutions towards more sustainable management can be promoted by legislation that sets requirements and so enforces the search for solutions. For this to be successful it must recognise the large diversity of bottlenecks felt by farmers, as well as for biophysical and economic constraints and their dependency on local factors. For such enforcement to result in desired outcomes, it must likely be accompanied by independent advice. Parts of Europe have seen the collapse of such independent advisory systems over the past decades, which does not help the infusion of knowledge for improving ecosystem services, where focus now is largely restricted to commercial interests.

Advisors can help farmers plan, implement and evaluate soil management to best match the goals - soil services to be promoted - set for the region. Such approach requires, in addition, the local assessment of practices in terms of generating the desired outcomes (services), which in turn calls for applied research on farms representative of conditions in the region (Ingram et al., 2016).

We refrain here from analysing the suitability of various types of policy instruments for the achievement of satisfactory levels of soil services. We find it impossible to make general statements as to whether measures/practices must be coded as mandatory or voluntary, based on tax or subsidy, commanded as Statutory Management Requirements (SMRs) or be seen as Good Agricultural and Environmental Conditions (GAECs), embedded in the Greening requirements or elaborated within the frame of the Rural Development Programs.

6.5. Integrated Soil Management Plan (ISMP)

As discussed in Chapter 4, an EIP-AGRI Focus Group (Molendijk, 2015) compiled a set of recommendations for integrated pest management practices to control soil borne diseases. Among their main recommendations was the proposal to introduce soil management planning as a mandatory tool on all farms. This approach could be effectively extended to include all aspects of soil management, in view of protecting the soil as well as the services they support. Such integrated soil management plan (ISMP) would best be guided by independent experts, and may combine on-farm monitoring of indicators of highly localized importance - such as the occurrence of pathogen in (sections of) parcels - with monitoring of soil-supported services on a wider scale (catchment, landscape) as advocated in Section 6.4.6. Monitoring should go hand in hand with the development of decision support systems, and new tools for both are becoming rapidly available. For IPM

of soil borne diseases, these include high-throughput PFLA, metagenomics (identification of genes for soil functions), hyperspectral remote sensing by unmanned aircraft and satellites, precision technology for reduced pesticide dosage aided or not by imaging sensors, and web services for decision making, data management and visualisation of geo-referenced soil health issues. Similarly, innovations in other sub-domains of the applied soil and crop sciences yield new opportunities that should be seized to make soil management more sustainable, and to support informed decision making. Planning of measures, and regular monitoring and evaluation of progress and innovation barriers should be part of the permanent cycle that would constitute integrated soil management planning.

6.6. A ranking of soil threats

The ranking of soil threats is, admittedly, a tedious task. Some threats have been documented, in this report as elsewhere in the literature, in terms of risks or vulnerability, rather than actual occurrence. For most threats, quantitative evidence is limited and scattered, lacking of uniformity across member states in terms of monitoring and reporting, but also in how risks are quantified and assessed. Risks are often derived by modelling exercises, using partial information on vulnerability as related to soil properties and to the occurrence of drivers.

In spite of the reservations that scientists should foster when ranking threats that cannot be compared directly (for lack of a uniform scale to express them; for lack of certainty about their geographic extent and gravity), we believe that this report to the European Parliament must include an attempt to rank threats by urgency. This is because, as in other domains, society can simply not afford to await full completion of databases, agreement on indicators to be reported, and the standardisation of monitoring procedures across the EU. Such pre-conditions, by themselves, are hard to fulfil on a continent so heterogeneous in many respects. Threats progress by their own pace, unhampered by debate, and most of them are not reversible within the time scale of one or a few human generations. We believe that several of the threats mentioned are sufficiently severe to warrant firm public intervention, in spite of uncertainties.

Table 5 scores the respective threats by a number of criteria: how rapidly they occur, and the time needed to revert to the pre-damage situation (if at all possible); our level of scientific understanding of the degradation process; and the geographical extent of the threat: whether it occurs in specific pockets only, or across the EU. Next, the same table shows the expected impact of the threat on the main soil functions that sustain key services: primary production, climate regulation, regulation of the water cycle (quantity, quality), the provision of habitat for biodiversity, and nutrient cycling. Subsequently, Table 6 presents a scoring in terms of control measures to halt or reduce the threats: whether it is clear what should be done in the field, how sure we can be of the measure's success, and how easy it would be to implement the measure. 'Easy' in terms of policies implies that the implementation requires no or little differentiation to local conditions, including from the perspective of agro-technical feasibility. 'Very easy' measures then have the character of a generic ban or obligation. Finally, ease of implementation by the farmers refers to cost or income lost, and availability of agro-technical solutions. Sometimes the solution is clear and available, but costly to the farmer, for example in the case of downscaling subsoil damage by farm machinery (compaction).

Based on these criteria, we rank highest (most urgently calling for action) the intact conservation of carbon-rich soil and systems. These are peatlands and extensive pasture

systems. Their degradation will have severe impacts on climate, biodiversity and water regulation. The impacts of such degradation will likely be felt across the EU and – in the case of climate – globally.

Within the domain of (more) intensive farming systems, top priority should be the curbing of topsoil erosion by water, the compaction of subsoil by farm machinery and, foremost, the loss of productive land to urbanization and infrastructure (sealing). These three all have a strong bearing in the long run on primary production and therefore on economic sustainability of the farming sector and on food security and will, given rising food demand, increase pressure on remaining productive land as well as on extensive systems of high value for climate and biodiversity. These threats occur widespread across the EU. Their impacts may be felt more locally than in the above case, but are not restricted to the field parcels where the degradation occurs, and they may reach across member state borders.

Among the above three threats (erosion, compaction, sealing), subsoil compaction is generally less well-documented. However, the continuous trend for upscaling field mechanization is likely to cause increasing damage to the subsoil. To repair such damage with lasting benefit is notoriously difficult, besides the immense effort (also in terms of fossil fuel spent) to address this issue. With the increasing frequency and gravity of weather extremes under climate change (drought spells; intense precipitation; flooding), subsoil conditions in terms of hydraulic properties and ability to support deep rooting will increase in importance. From a precautionary perspective, we believe that ignoring the trend of increasing traffic loads on soils would be unwise. It would be a sensible precaution – comparable to speed limits on highways – to introduce a maximum permissible limit to the wheel load carrying capacity (WLCC; Schjønning et al., 2015) for traffic on all agricultural soils. Differentiated limits could be applied, depending on soil vulnerability. This would trigger innovation towards lighter systems, autonomous transport systems for harvesting operations, and similar advances. Reducing tyre pressure and the use of caterpillars bring some solace for the prevention of near surface structure damage, but are largely ineffective against subsoil compaction.

Water erosion occurs in many regions of the EU and the remedies must be chosen to match the local conditions. The formal delimitation of vulnerable zones, and the keeping of effective soil cover types, and/or tillage systems tailored to the local potential and needs of such zones should be considered, in our view, as necessary steps towards the enforcement of practices that reduce the loss of topsoil by erosion.

As for the intention of ‘no-net land take’ by 2050 (COM 571/2011), the projected time span seems too comfortably long, given the expected steep rise in global population and the associated increasing demands for food and biomass during this period. This is particularly worrying since other drivers such as climate change will likely deteriorate conditions for crop production in large parts of the EU and elsewhere.

Table 5. Qualitative evaluation of soil threats

					Impact on services					
	Degrad. rate fast	Recov. rate slow	Science clear	Geogr. extent large	Prim. prod.	Climate (GHG)	Water quant. regul.	Water qual. regul.	Habitat biodiv.	Nutrient cycling
Erosion by wind	XXX	XXX	XXX	X	XX	X	X	X	XX	XX
Erosion by water	XXX	XXXX	XXX	XXX	XXX	X	XXX	XX	XX	XXX
Floods & Land slides	XXX	XXXX	XXX	X	XXX	X	XXX	XX	XX	XXX
SOC-loss peat soils	XXX	XXXX	XXX	XX	-	XXXX	XXXX	XXX	XXXX	-
SOC-loss mineral soils	X	XX	XXX	XXX	XX	XX	XX	XXX	XXX	XX
Compaction	XX	XXXX	XXX	XXXX	XXX	XX	XXX	X	X	XXX
Sodification	X	XX	XXX	X	XXX	X	XX	XXX	XXX	X
Contamination	XXX	XX	XXX	XX	X	X	X	XXX	XX	X
Acidification	X	X	XXX	X	XX	X	X	X	XX	X
Low fertility	X	X	XXX	X	XXX	X	X	X	X	X
Desertification	X	XXXX	XX	X	XXX	XXX	XXX	X	XXX	X
Loss aboveground biodiv.	XXX	XX	XX	XXX	X	X	X	X	XXX	X
Loss of soil biodiversity	X	XXX	X	X	XX	X	XXX	XX	XXX	XX
Infestation S-Borne	XXX	XXX	XXX	XX	XXX	X	X	X	X	X
Sealing	XXXX	XXXX	XXX	X	XXXX	X	XXXX	X	XXXX	XXXX

Scale: X: Little, XX: Some/somewhat, XXX: Very, XXXX: Extremely.

Example on how to read the table: for soil erosion by water, degradation rate is very fast, recovery rate extremely slow, scientific understanding very clear, geographic extent very large, etc.

Table 6. Qualitative evaluation of control measures available to mitigate the respective soil threats

	Control measures clear	Control measures effective	Control measures easy to implement for policy	Control measures easy to implement for farmer (cheap - technol. avail.)
Erosion by wind	XXX	XXX	XX	XXX – XXX
Erosion by water	XXX	XXX	XX	XX - XX
Floods & Land slides	XX	XX	XX	X – X
SOC-loss peat soils	XXX	XXX	XXX	-
SOC-loss mineral soils	XXX	XX	XX	XX - XX
Compaction	XXX	XX	XXX	X - XXX
Sodification	XXX	XX	X	X - XX
Contamination	XXX	XXX	XXX	XX - XX
Acidification	XXX	XXX	XXX	XXX – XXX
Low fertility	XXX	XXX	XXX	XXX – XXX
Desertification	X	X	X	X – X
Loss of above-ground biodiv.	XXX	XX	XX	X – XXX
Loss of soil biodiversity	XX	XX	X	X – XX
Infestation S-Borne	XXX	XX	XX	X - XXX
Sealing	XXX	XXX	XXX	-

Scale: X: Little, XX: Some, XXX: Very.

Examples on how to read the table: for desertification (X symbol in all columns) control measures are not very clear, are little effective, and not easy to implement through policies.

In the last column, the first set of X-symbols refers to affordability in terms of cost (e.g., XXX signifies 'very cheap'); the second set refers to the availability of appropriate/feasible technology (e.g., XXX signifies that the technology is readily available / very easy to apply).

7. CONCLUSIONS AND RECOMMENDATIONS

Conclusions and supporting information are in roman font, *recommendations for policies in italics*.

7.1. Soil functions and services

- For soils under agriculture, the five main ecosystem services include the provision of harvestable crops, clean freshwater and nutrients for plants and animals, conservation of suitable habitats for biodiversity, and maintenance of a benign climate. The corresponding soil functions are primary production, water regulation, nutrient cycling, habitat support and climate regulation.

7.2. Threats to soils and soil services

- Soil threats jeopardize the state and/or functioning of soils. Their unchecked persistence will in the short or long run degrade the level of services provided or supported by soils.
- *Major soil threats at the whole-system level that call for policy action are:*
 - (i) *the degradation of peatland and similar organic soils by drainage under agriculture and forestry, mostly in Northern and North-Western Europe; and the potential encroachment of agriculture and forestry onto pristine peatlands as favoured by climate change in Northern Europe;*
 - (ii) *the drastic alteration of traditional wood-pasture systems in various parts of the EU;*
 - (iii) *the grabbing of productive agricultural land for non-agricultural use (sealing).*

Policies to address the above threats merely need to aim at full conservation of the corresponding natural, semi-natural and agricultural status of these systems.

- *Both peatlands and extensive grassland and wood-pasture systems store large amounts of soil carbon. The protection of these soils and carbon stocks is vital in view of climate change mitigation, conservation of their rich biodiversity (in soil and aboveground), regulation of the water cycle, and (in the case of grasslands and wood-pastures) protection from water erosion. These systems need full protection at the whole-system level. Their loss cannot be reverted on the human time scale.*
- *In relation to the protection of wood-pastures, we call attention to the overly complex Commission Delegated Regulation EU 640/2014 (supplementing EU Regulation 1306/2013), Art. 9-10, concerning the parcel area fraction eligible for direct payments under the CAP in the case of pastures with scattered trees. While MS have certain flexibility in assigning eligibility coefficients, this does not in practice result in adequate protection of the tree component. Rather, certain areas have seen the deliberate removal of trees in order to increase the area eligible for direct payments, which triggers land degradation and dramatic loss of ecosystem services. In other cases, afforestation premiums outweigh the revenues from maintaining the mixed system, and so invite destruction. Current legislation appears inadequate to offer protection to such cross-sectoral (silvopastoral / agro-forestry) systems.*
- *Growing global food demand in coming decades already calls for intensification of all suitable land in current use, and loss of productive land by sealing is therefore simply unaffordable. Such loss increased the need to further intensify production on*

remaining farmland, and would increase pressure on extensive systems with high biodiversity, affecting soil services in both types of systems. Ironically, the best soils are often first to disappear under urbanisation, as soils near original habitation have been historically better amended. As for the intention of 'no-net land take' by 2050 (COM 571/2011), the projected time span seems too comfortably long, given the expected steep rise in global population and the associated increasing demands for food and biomass during this period. This is particularly worrying since other drivers such as climate change will likely deteriorate conditions for crop production in large parts of the EU and elsewhere.

- Other major threats, in contrast, require policy measures that enforce the focussed adaption of farming practices. These threats are, by our ranking:
 - (iv) Soil erosion by water.
 - (v) Compaction of the subsoil by heavy machinery.
- Water erosion occurs in many regions of the EU and the remedies must be chosen to match the local conditions. The formal delimitation of vulnerable zones, and the keeping of effective soil cover types, and/or tillage systems tailored to the local potential and needs of such zones should be considered, in our view, as necessary steps towards the enforcement of practices that reduce the loss of topsoil by erosion.
- In spite of its limited documentation, we regard subsoil compaction as a serious and wide-spread menace, and include it in our shortlist of major threats that call for policy action. Its main driver is the ongoing upscaling of farm mechanisation to increase labour productivity. This trend is likely to cause increasing damage to the subsoil. To repair such damage with lasting benefit is notoriously difficult, besides the immense effort (also in terms of fossil fuel spent) to address this issue. With the increasing frequency and gravity of weather extremes under climate change (drought spells; intense precipitation; flooding), subsoil conditions in terms of hydraulic properties and ability to support deep rooting will increase in importance. From a precautionary perspective, we believe that ignoring the trend of increasing traffic loads on soils would be unwise. It would be a sensible precaution - comparable to speed limits on highways - to introduce a maximum permissible limit to the wheel load carrying capacity (WLCC; Schjønning et al., 2015) for traffic on all agricultural soils. Differentiated limits could be applied, depending on soil vulnerability (texture). This would trigger innovation towards lighter systems, autonomous transport systems for harvesting operations, and similar advances. Reducing tyre pressure and the use of caterpillars bring some solace for the prevention of near surface structure damage, but are largely ineffective against subsoil compaction.

7.3. Soil in the farm economy

- Although the soil is the farmer's most precious economic resource, there may be several reasons why farmers may not take specific actions to protect or improve their soil in their own commercial interest. These reasons include:
 - (i) Soils generally respond slowly to changes in management practices. Even given sufficient time, soil properties can only be improved within certain limits set by climate and parent material.
 - (ii) Effects of soil improvements on crop yields remain often unnoticed. Inter-annual (weather-related) variation in attainable yield, timing of operations and the use of inputs, and crop genotype all have relatively strong effects on actual yield and

produce quality, and can easily mask benefits from improved soil conditions, even in the long run. The low responsiveness of yield to management-induced soil improvements is, we believe, a major reason for farmers to avoid practices that may improve/protect one or more aspects of the soil, but bring disadvantages relative to current management.

- (iii) Benefits or problems may show only under extreme conditions (e.g., drought, waterlogging) or go unnoticed as long as critical thresholds are not crossed (disease suppressiveness)
- (iv) Farmers are unaware of existing solutions to address the problem; or no ready-for-practice solutions really exist;
- (v) Farmers can sometimes correct for soil-induced crop stresses, by using extra inputs (fertilisers; crop protection agents; irrigation) or alternative equipment (deep-plow; drainage) – thus largely externalising the cost of such ‘repair measures’ if their impacts (emissions; lowering groundwater table; loss of biodiversity) are mostly felt elsewhere.
- (vi) Threats are simply not present on the farm.

7.4. Public interest in soils – the EU governance level

We see various justifications for tackling soil threats at the European Union rather than the national level:

- (i) Biophysical impacts of threats do not stop at member states’ borders.
- (ii) Economic impacts of threats, or consequences of the policy measures to counter them, are not limited to the member state either (‘spill-over’).
- (iii) The European Union level allows the spatial optimisation of interventions and the sharing of their costs.
- (iv) The cost efficiency of the necessary research and innovation efforts to make soil management more sustainable increases with the geographic extent of the application domain.
- (v) Local democracy may hamper the implementation of unpopular measures, if these lack central underpinning at higher level.

7.5. Biophysical principles in soil management

- Soil management that aims to protect soils and promote their services can be usefully guided by the following general principles:
 - (i) Soil cover protects the soil from the impact of rain and reduces erosion by water or wind; it provides shelter and/or food for biodiversity; catch crops can also reduce off-season nitrate leaching.
 - (ii) Fresh organic material, irrespective of its source, is the main food for soil life. Its regular supply promotes bioturbation/burrowing and aggregation (by secretion, excreta, fungi hyphen) and these processes help to build soil structure which improves infiltration, retention and drainage of water; aeration and rooting; and the formation of micro-habitats for biodiversity. Soil structure favors soil workability, seedbed quality, and it may render the soil-crop system more resilient in the face of extremes and pressures (drought, waterlogging, soil borne diseases).

- (iii) Soil disturbance (tillage) has a negative impact on soil macro-fauna such as earthworms, which contribute to maintaining soil structure and associated hydraulic properties.
- (iv) Heavy traffic load and high passing frequency of farm machinery causes damage to soil structure. Especially in deeper layers, this is very difficult to restore. Damaged soil structure increases the risk of ponding and waterlogging in lowlands, and increases runoff and erosion risks in uplands, causing negative downstream effects (flooding, siltation, pollution).
- (v) High frequency of host crop species and the occurrence of specific cropping sequences (including cover crops) promote the build-up of soil borne pathogen populations which may cause crop damage.
- *Practices that aim to protect soils and promote their services are often referred to as 'soil improving practices'. However, this qualification does not express a general truth: 'win-wins' are rare and all such practices come with trade-offs: negative consequences (sometimes grave) for either soil condition itself or for soil-supported services, with impacts on crop yield, farm income, climate, air and water quality, or biodiversity. Soil management must therefore be optimised within the wider frame of sustainability goals, not just focussing on the preservation of the soil itself.*
- The suitability and successful application of practices (promoting benefits and avoiding drawbacks) are highly reliant on local conditions (climate, soil, topography, hydrology, crop, farm type). Potential drawbacks are too many and too complex to cite them here, see Chapter 5 and Section 6.4.4 for details.
- Practices often denominated as 'soil improving practices' include crop rotation (as opposed to monoculture); forms of reduced tillage (versus mouldboard ploughing); the use organic manures (versus mineral fertilisers); the retention of crop residues (versus removal); and the use of catch and cover crops and green manures (versus bare fallow in between main crop seasons). More radical deviations from conventional practices are, for example, the use of ley phases (one or more years) in arable rotations, and the use of guided traffic ('tram-lines') for wheels to pass always over established tracks. Organic farming and conservation agriculture strive at the 'whole-farming-system level' to protect soils and promote their services. As stated, all of these come with drawbacks that cannot generally be ignored.
- *Although many soil threats are expressions of a development that we tend to characterize as 'intensification', its opposite 'extensification' - including approaches known as organic agriculture - is an unfocussed and likely inadequate answer to counter these threats. Instead, specific and probably pedoclimatically tuned measures are needed with respect to organic matter management, machinery weight specifications, hydrological management, the use of agro-chemicals, and landscaping are needed.*

7.6. Soil carbon and climate

- *Peatlands are the most efficient carbon stores of all terrestrial ecosystems, containing 455,000 Mt of carbon, or twice the amount found in the world's forest biomass. The majority of this carbon is stored in the saturated peat soil. Pristine peatlands are still sequestering carbon at a rate of 96 Mt carbon per year (Dunn and Freeman, 2014). Louwagie et al. (2009) state that EU soils store more than 70,000 Mt of organic carbon, as compared to about 2,000 Mt of carbon altogether emitted by member states annually, and argue that "releasing just a small fraction of the carbon currently stored in European soils to the atmosphere could wipe out emission savings in other*

sectors of the economy". Schils et al. (2008) concluded that "the largest emissions of CO₂ 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 high content of organic matter." (Quoted from Van den Akker et al., 2016).

- Practices to promote the accumulation of SOM in agricultural soils, or to mitigate its decline, can definitely bring agronomic benefits and contribute to the protection of soils and various soil services. However, the scope for accumulating SOC (a major constituent of SOM) as a climate mitigation measure is very limited, given the following considerations:
 - (i) the availability of additional carbon sources (not currently returned to soil at some point in time or space) is limited, even if all urban waste in the EU were mobilized for this purpose;
 - (ii) nitrous oxide (N₂O) emissions are associated with most practices that enhance SOC accumulation;
 - (iii) farm economy/market demand limit the cultivation of crops with high SOC contribution;
 - (iv) only carbon that originates from extra primary production and carbon that is saved from incineration of residues contributes to climate change mitigation;
 - (v) the use of organic biomass for bioenergy to replace fossil fuel is likely to contribute substantially more to climate mitigation than soil incorporation of the biomass;
 - (vi) gains in SOC made by adjusting farming practices are reversible – i.e., SOC gained by the practice can be lost again rapidly if the practice is discontinued – while fossil fuel savings forgone and N₂O emitted are irreversible.

7.7. Monitoring and enforcement

- Soil differs markedly from other compartments of the biosphere (atmosphere, water bodies) in that it is hardly subject to mixing, while its key processes and governing properties are highly heterogeneous in space due to variation in parent material, exposure and topography, climate and management history. Heterogeneity and absence of mixing imply that the monitoring of soil condition requires high sampling density, is usually expensive, time consuming or difficult, and is prone to large sampling and measurement error. Interpretation of observed indicator values is complex, too, and needs local calibration. For certain properties, the use of geophysical methods may resolve such constraints.
- We recommend that soil protection policies allow for local ranking of priorities: soil services in need of promotion, and/or threats in need of mitigation. Subsequently, a two-pronged approach can combine (i) the promotion/enforcement at farm level of locally suitable practices, with (ii) monitoring and evaluation at the catchment scale of services delivered and/or threats mitigated.
- Soil protection policies would be most usefully implemented if combined with collective action, and extension programmes with stepped targets and communication on progress achieved. Compliance of farmers with proposed measures can be verified by remote sensing or other survey methods, and is generally less complex than the monitoring and interpretation of soil status indicators directly.

- *Integrated soil management planning (ISMP) should be introduced as a mandatory farm-level tool on all farms. The integrated soil management plan should address all aspects of soil management, e.g., crop rotation, IPM for soil borne diseases and weeds, SOM balance, soil fertility management, water management, tillage and soil structure management, ...). It should be geared towards protecting the soil as well as the services the soil supports. Implementing such integrated soil management plan requires guidance by independent experts. The planning of measures, and regular evaluation of progress based on monitoring as well as barriers encountered should be part of the permanent cycle that would constitute integrated soil management planning.*
- *Examples of services (or threats mitigated) that could thus be monitored at catchment/landscape scale are the regulation of hydrology (response time in water channels), mitigation of erosion (sediment load in channels), provision of clean water (surface water pollutant load; nutrients; biocides), and aboveground biodiversity. Some services, however, require in-situ (per field) measurement for their assessment. These include soil biodiversity, botanical biodiversity, and C-sequestration. However, we regard it unlikely that it is feasible to monitor these for enforcement or verification purposes.*
- *The two-pronged approach requires a priori assessment of practices most likely to deliver the desired outcomes, given the local conditions and the services in focus, as well as identification of solutions to local barriers that hamper adoption of the practices.*
- *The above approach accounts implicitly for soil-mediated services as well as services/threats directly affected by the management practice itself, i.e., without involving a change in soil condition or soil processes. Although (some) policy measures are already in place to protect non-soil compartments (water, air, habitats), we argue that so many trade-offs, synergies and other links exist between the goals of those 'other environmental policies' and those of soil protection, that policies can be more effective and coherent if they target soil protection as part of a well-balanced set of locally prioritised environmental stakes. The goals would be pursued at the catchment and landscape scales, and the compliance with policy measures would be pursued at farm level.*
- *Finally, for threats which cannot be countered by the individual farmer due to the scale of required measures (e.g. salinization, desertification, peatland degradation, landslides) it is self-evident that the catchment or landscape scale is the only level permitting effective action.*

7.8. Innovation, communication, learning and extension

- *Changes in soil management come with barriers that complicate proper integration into the farming system. It appears that combining technological innovation with exchange of experiences and insights by practitioners will be vital to promote practices that bring extra cost or other difficulties. "Sustainable intensification means more knowledge input and more 'services output' per ha" (Buckwell et al., 2014). This definitely applies also to the subdomain of soil management.*
- *Some basic biophysical principles cannot be changed and the barriers defined by them are, in the long run, perhaps the most challenging. Addressing these may require drastic changes in production systems. Other barriers can be tackled by focused research and innovation efforts. Developments in aerial surveillance, computer vision and crop sensing, autonomous transport, GPS guidance, crop breeding and genome*

editing, and many other fields of technology may bring solutions to long-felt bottlenecks. Turning technological development towards resolving conflicts and trade-offs in soil management is, in our view, the most promising track to progress. Improved methods for seeding, weed and pest control, harvesting and transport operations, and post-harvest processing of produce can all contribute to more sustainable soil management. And so can resistance breeding in crops, cover crops and green manures.

- *Communication tools and access to existing knowledge for innovation are rapidly improving with the help of web semantics and linked open data infrastructure, decision support via hand-held devices, etc. All of these serve to support learning networks and advisory systems. For an overview of the roles ICT can play in agriculture and participatory innovation, we refer to EU SCAR (2015). We can only confirm that current R&D policy under Horizon-2020 and the EIP-Agri - with its focus groups setting agenda's, thematic networks surveying bottlenecks and possible solutions, and operational groups connecting the various actor-types per supply chain around shared problems – has set out an ambitious programme deserving of continued support.*
- *Innovation towards more sustainable management can be promoted by legislation that sets requirements and so enforces the search for solutions. For such twinned approach (support innovation; legal enforcement) to result in desired outcomes, it must recognise the large diversity of bottlenecks felt by farmers, set by local biophysical and economic constraints. Also, it must likely be accompanied by independent advice. Parts of Europe have seen the collapse of such independent advisory systems over the past decades. This hampers the infusion of knowledge for improving ecosystem services, where focus now is largely restricted to commercial interests. Such commercially driven advisory systems are not likely to foster the innovation needed to drive the needed long-term local innovations in sustainable soil management.*

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ANNEX I. INTERACTIONS BETWEEN SOIL THREATS

Interactions between soil threats, reproduced from Stolte et al. (2016), their Table 14.5. Dot sizes indicate impacts: low, moderate and large for small, medium and large impacts

Soil threat	Water erosion	Wind erosion	SOM decline peat soils	SOM decline mineral soils	Compaction	Sealing	Contamination	Salinization	Desertification	Flooding and landslides	Biodiversity decline
Water erosion											
Wind erosion											
SOM decline peat soils											
SOM decline mineral soils											
Compaction											
Sealing											
Contamination											
Salinization											
Desertification											
Flooding and landslides											
Biodiversity decline											

Source:

ANNEX II. GREENHOUSE GAS LOSSES FROM DRAINED PEAT SOILS

Emissions of greenhouse gases from peats soils under agriculture. Calculations based on: grassland emissions 20 t CO₂/ha/y, cropland emissions 40 t CO₂/ha/y (see Oleszczuk et al., 2008), C/N ratio =20, assuming that the major part of agricultural peat soils are fen peats; 1.25% of mineralised N converted into N₂O (Mosier et al., 1998). Crop and grassland areas are based on Byrne et al. (2004). This table was reproduced from Schils et al. (2008), their Table 7

Country	Agricultural Area km ²	Crop area km ²	Grass area km ²	CO ₂ - C Mt / a	CO ₂ Mt / a	N ₂ O CO ₂ eq Mt / a	Total CO ₂ eq Mt / a
<i>Member states</i>							
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
<i>Other European</i>							
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

Source ^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).

ANNEX III. SELECTED OUTCOMES FROM FRELIH-LARSEN ET AL. (2016)

The following conclusions are cited from the extensive work carried out by Ana Freluh-Larsen and her team, entitled 'Updated Inventory and Assessment of Soil Protection Policy Instruments in EU Member States' (Freluh-Larsen et al., 2016). All text fragments were taken from their Section 10.2 – in some cases we shortened their formulation.

- Many different policy instruments at EU and Member State level exist that either explicitly reference soil threats or soil functions, or implicitly offer some form of protection for soils. 35 EU level and 671 Member State policy instruments were identified.
- At EU level, the instruments range from strategic documents, to directives and regulations as well as funding instruments. At MS level, three quarters of the instruments are regulatory instruments, and the majority (61% out of 671) are binding in nature. Nearly half (45%) of all MS instruments are directly linked to EU policies: their implementation is mandated by the EU acquis. Another 21% are linked partly to EU binding instruments: they implement the EU binding legislation but also go beyond the acquis in either the degree of ambition that they set for EU requirements or they regulate additional areas that do not derive from the EU acquis. In total, 225 identified instruments (35.5%) are 'nationally initiated' policies, i.e. policies partly linked to EU non-binding policies or not linked to any EU requirements. The number and diversity of the MS instruments reflects the cross-cutting nature of soils; it also underscores the importance and challenge of integration and coordination of policy instruments in order to ensure that soil issues are addressed coherently.
- Major opportunities for soil protection can emerge from improved use of existing legislation, or through upcoming EU policy dossiers. For existing legislation: to further build on priorities within the 7th EAP, and promote more holistic soil management as a tool for delivering goals on sustainable land management and more sustainable and resource efficient nutrient cycling; and to pursue a binding legislative proposal.
- The climate and energy package for 2020 – 2030 includes potential opportunities for soil protection linked to GHG emission reduction targets through better soil organic matter protection and management, and the more sustainable use of inorganic (especially nitrogen) fertilisers and manure. Specifically, there is some potential for soil protection in the current proposals for a Land Use and Land Use Change and Forestry Regulation (LULUCF) and an Effort Sharing Regulation (ESR) requiring GHG emission reductions from sectors excluded from the EU Emissions Trading Scheme, including agriculture up to 2030.
- Whether the strengths and opportunities identified above indeed result in benefits for soil protection depends on how soil issues are integrated and prioritised in these policy instruments. There is no guarantee that, for example, GHG mitigation in the agricultural sector will be prioritised under the ESR since the split of effort among non ETS sectors is determined by each Member State. Moreover, even if Member States were to prioritise agriculture as part of their efforts to maximise reductions in net GHG emissions, there is no requirement that this must encompass sustainable soil management techniques that also deliver other soil functions. Similar to climate change policies, water protection policies are also important existing instruments identified for protecting Europe's soils. Nonetheless, there is also no specific requirement in water quality legislation to remediate or protect the soil in situ.

Instead, the goal of water legislation is to prevent negative impacts on water bodies and this could be delivered in multiple ways.

- When looking at the weaknesses of EU level policy instruments in protecting Europe's soils the lack of a coherent, strategic policy framework was highlighted across all policy clusters. This lack of a common and integrated strategic policy frame is an important gap, one that had been intended to be filled by the withdrawn Soil Framework Directive proposal. Therefore, a strategic policy framework is missing that would, in an integrated manner: conceptualise soil issues (including common definitions on good status); set out priorities and targets; define monitoring parameters and desired end points; and define the role of different policy instruments in delivering good soil status. In the absence of a common policy framework, soils are addressed in many policy instruments but there is no EU level political or legislative driver for establishing integration and coherence towards an agreed strategic aim and objectives. Not only does this mean the EU policy frame is limited for soils, it means that existing strengths and opportunities that have been identified cannot be fully explored and exploited.
- The analysis of nationally initiated policy instruments (national initiatives) in the EU-28 Member States confirms that the lack of strategic coordination is an important theme. Some Member States have comprehensive policies in place that take account of soil protection. However, many of the policies that require the integration of and strengthening in relation to soil protection needs and objectives are EU level policies. This includes critical measures relating to agricultural land management, pollution prevention, water and biodiversity protection. Member State opportunities to address these EU level policy questions of integration are more limited.

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