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*Review*

# Soil Degradation and Soil Quality in Western Europe: Current Situation and Future Perspectives

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**Abstract:** The extent and causes of chemical, physical and biological degradation of soil, and of soil loss, vary greatly in different countries in Western Europe. The objective of this review paper is to examine these issues and also strategies for soil protection and future perspectives for soil quality evaluation, in light of present legislation aimed at soil protection. Agriculture and forestry are the main causes of many of the above problems, especially physical degradation, erosion and organic matter loss. Land take and soil sealing have increased in recent decades, further enhancing the problems. In agricultural land, conservation farming, organic farming and other soil-friendly practices have been seen to have site-specific effects, depending on the soil characteristics and the particular types of land use and land users. No single soil management strategy is suitable for all regions, soil types and soil uses. Except for soil contamination, specific legislation for soil protection is lacking in Western Europe. The Thematic Strategy for Soil Protection in the European Union has produced valuable information and has encouraged the development of networks and databases. However, soil degradation is addressed only indirectly in environmental policies and through the Common Agricultural Policy of the European Union, which promotes farming practices that support soil conservation. Despite these efforts, there remains a need for soil monitoring networks and decision-support systems aimed at optimization of soil quality in the region. The pressure on European soils will continue in the future, and a clearly defined regulatory framework is needed.

**Keywords:** soil quality; Western Europe; sustainable soil management

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## 1. Soil Degradation in Western Europe

### *1.1. Geographical Diversity of Soils in Western Europe*

Western Europe (WE) is a loose term for the collection of countries lying in the most westerly part of Europe. However, the definition is context-dependent as it has political and geographic connotations. From a geographical point of view, the United Nations' geoscheme divides Europe into four regions: Western, Eastern, Northern and Southern Europe [1]. In this paper, the term WE is used to refer to countries in the Western half of the continent, including the Western, Northern and Southern regions [1]. Most of the countries border with the Atlantic Ocean and/or the West Mediterranean Sea. The region includes the countries that had joined the European Union (EU) before 2000 (Table 1) and which have therefore implemented Common Agricultural Policy (CAP) regulations and other EU environmental directives affecting soils (*i.e.*, Nitrates, Water and Pesticides Framework Directives) during at least the last 15 years. Iceland, Norway and Switzerland are included because of their geographical location. Historical land government and ownership also have common traits in the region. As a result, the trends in agricultural and forest soil management, and the strategies for mitigating soil degradation are somewhat similar in these countries.

In WE, interactions between climatic, geological and topographic conditions have resulted in a large natural diversity of soils. Twenty-three of the 32 reference soil groups included in the World Reference Base [2] occur in WE. Soil groups are presented for each country in Table 1, following the Soil Atlas of Europe [3]. The most common soils across WE are Cambisols, Podzols, Leptosols, Luvisols, Fluvisols Gleysols, Regosols and Calcisols. However, distribution of the soils is uneven. Cambisols and Podzols each occupy 12%–14% of the total land in Europe. Cambisols occur in a wide variety of environments and under all types of vegetation, and they are present in almost all countries. Podzols are mainly present in the boreal and temperate zones of Northern countries (Norway, Sweden, Finland, Denmark, Scotland, N Germany and some areas of France and the Alps). Leptosols (9% of the European land) are mainly present in mountainous regions of Spain, France, Switzerland, NE Italy, and Norway. Luvisols, Fluvisols, Histosols, Gleysols, Regosols and Calcisols each occupy 5%–6% of the land. Fluvisols are common in river fans, valleys, and tidal marshes in all climate zones. Histosols, which mainly comprise organic matter, are common in boreal and sub-artic regions, Scotland and Ireland. Gleysols occur in lowland areas that have been saturated with groundwater for long periods, mainly in the United Kingdom (UK) and Ireland. Regosols are widespread in arid and semi-arid areas, as well as in mountainous regions of Portugal and Spain. Finally, Calcisols are also common in Spain. They appear in regions with calcareous parent materials and distinct dry seasons, and in dry zones where carbonate-rich groundwater appears near the surface.

**Table 1.** Major soil types in each country according to the Soil Atlas of Europe [3]. Soils shown in bold type are predominant in the particular country.

| European Region | Country * | Main Soil Types [2] |
|---|---|---|
| North Western | Denmark * | Cambisol, Gleysol, Luvisol, Podzol |
| | Finland * | Cambisol, Gleysol, Histosol, Leptosol, Podzol |
| | Iceland | Andosol, Histosol, Luvisol, Podzol, Umbrisol |
| | Ireland * | Cambisol, Gleysol, Leptosol |
| | Norway | Albeluvisol, Cambisol, Leptosol, Phaeozem, Podzol |
| | Sweden * | Cambisol, Histosol, Leptosol, Podzol, Regosol |
| | United Kingdom * | Cambisol, Gleysol, Histosol, Leptosol, Luvisol, Podzol, Umbrisol |
| Central Western | Austria * | Cambisol, Chernozem, Fluvisol, Leptosol, Luvisol, Podzol |
| | Belgium * | Albeluvisol, Cambisol, Fluvisol, Luvisol, Podzol |
| | France * | Albeluvisol, Andosol, Calcisol, Cambisol, Leptosol, Luvisol, Podzol |
| | Germany * | Cambisol, Chernozem, Fluvisol, Luvisol, Podzol, Umbrisol |
| | Luxembourg * | Arenosol, Cambisol, Fluvisol |
| | Netherlands * | Fluvisol, Gleysol, Histosol, Podzol |
| | Switzerland | Albeluvisol, Cambisol, Leptosol, Luvisol, Podzol, Umbrisol |
| South Western | Greece * | Cambisol, Fluvisol, Leptosol, Luvisol, Vertisol |
| | Italy * | Andosol, Calcisol, Cambisol, Fluvisol, Leptosol, Luvisol, Podzol, Vertisol |
| | Portugal * | Cambisol, Fluvisol, Luvisol, Podzol, Regosol, Umbrisol, Vertisol |
| | Spain * | Calcisol, Cambisol, Fluvisol, Gypsisol, Leptosol, Luvisol, Regosol, Umbrisol, Vertisol |

* Countries indicated with an asterisk are part of the European Union.

Different risks and soil degradation processes occur within the various areas of WE because of significant differences in the intrinsic properties of these types of soils and local variations in each soil group.

### *1.2. Historical Soil Management and Present Land-Use Patterns*

Land and soil are fundamental pillars of agricultural economies and are essential for industrial and urban development. In WE, agriculture and forestry have traditionally been the most widespread types of land use and have shaped the rural landscape [4]. Their relative importance has changed throughout history. The historical relationship between these types of land use has been shaped by socioeconomic and technological changes, demographical fluctuations and environmental variability. At the beginning of the Middle Ages, more than 80% of the population in WE was working in agriculture [4]. In the 14th century, the European population was drastically reduced as a result of the Black Death, and the cultivated area thus became smaller. The agricultural area then expanded until the end of the 18th century, when a new period of contraction began due to the increased productivity per hectare. During the 19th century, the agricultural area again expanded because population growth rate exceeded the agricultural productivity per hectare. Forests expanded and contracted in the opposite fashion to agricultural land. About half of WE forests are estimated to have been cleared prior to the Middle Ages [5]. The highest rates of deforestation occurred on the land best suited for farming, especially in France, Germany and the UK. Since then, the periods of most intense deforestation have coincided with those of high economic activity. Trees were felled when grain prices rose and forest land was converted to cropland. The use of wood for construction and shipbuilding also contributed to forest degradation and eventual deforestation in France, Portugal and Spain. Wood was also needed to fuel foundries and smelters early on in the Industrial Revolution, resulting in further forest degradation and deforestation, even on land not suitable for agriculture. Old-growth, primary forests essentially disappeared from WE in these periods. In the last 150 years, forests planted to produce raw materials expanded dramatically in WE. Requirements for food and/or timber, as well as town and country planning, were therefore the drivers of land use in WE until the mid 20th century.

In the early 1960s, the European Union's CAP enforced some degree of harmonization in agriculture and influenced land use patterns in many WE countries. The CAP was designed to guarantee the supply of sufficient food for EU citizens, to support the price of agricultural products, and to provide farmers with an acceptable level of income. Implementation of the CAP together with technological progress caused a sharp increase in agricultural productivity, which led to overproduction of agricultural goods in the 1970s and 1980s. In the following decades, the CAP has been reformed several times to regulate agricultural production and stabilize agricultural markets (see Section 3.2). As a consequence, the recent history of land use patterns again shows contraction of agricultural areas and expansion of forests. This process, which is particularly notable in the extensive margins (*i.e.*, alpine regions), has somewhat counterbalanced the historical loss of forest.

Nowadays, rural European landscapes are generally still strongly linked to agriculture and forestry. In 2009, agriculture (43% of the surface) and forestry (30%) were the most common primary land use categories in the EU [6]. As shown in Table 2, the land cover pattern in WE shows a clear North-South gradient [6–9]. Semi-natural forest prevails in the Northern countries (Norway, Sweden and Finland), while agricultural land (arable land, permanent crops and grassland) dominates the rural landscape between Denmark and Southern Europe, as well as in the UK and Ireland.

The proportion of arable land that is actually cultivated varies greatly among countries. This is related to population density and traditional land use. In the tilled arable area, although the most widespread system across WE is conventional tillage, the tillage systems vary greatly between countries [10] (see below, Section 2.1). Regarding woodland, the most common function of forests in WE is for wood production, although different patterns also exist among regions and countries. The Atlantic area of WE is characterized by forest plantations, which cover more than 5 million ha in Portugal, Spain, France and the UK, producing 33 million $m^3$ of wood in 2012. As a region, WE is the sixth industrial roundwood producer of the world, just behind Chile and in front of New Zealand. In the WE, the proportion of industrial roundwood produced in plantations is 31% in France, 52% in Spain, 65% in UK and 99% in Portugal [5]. The most common species in planted forests in WE are *Eucalyptus* spp., *Populus* spp; *Picea sitchensis*, *Pinus radiata* and *Pinus pinaster*.

Land management in agriculture and forestry has had a strong impact on the natural environment, including soils. On the one hand, over the centuries, farming has created and maintained a variety of valuable semi-natural habitats on which a wide range of wildlife depends for survival. On the other hand, land use changes and farming practices have also had negative impacts on natural resources, such as habitat fragmentation, loss of biodiversity and soil degradation. In particular, more than half of the land in Europe has suffered some type of soil degradation in the last few decades [11].

**Table 2.** Land use in each country in Western Europe (as a percentage of total land area).

| European Region | Country | Woodland | Cropland | Grassland | Artificial Land | Other * |
|---|---|---|---|---|---|---|
| **North Western** | Denmark | 18.3 | 48.5 | 21.1 | 7.1 | 4.9 |
| | Finland | 71.8 | 4.9 | 4.4 | 1.6 | 17.4 |
| | Iceland | 0.3 | 0.1 | 2.3 | 0.4 | 96.9 |
| | Ireland | 13.2 | 4.7 | 67.1 | 3.9 | 11.2 |
| | Norway | 37.5 | 2.9 | 0.7 | 2.1 | 56.8 |
| | Sweden | 75.6 | 4.3 | 4.6 | 1.8 | 13.7 |
| | United Kingdom | 19.8 | 21.7 | 40.1 | 6.5 | 11.9 |
| **Central Western** | Austria | 47.5 | 17.7 | 22.9 | 5.8 | 6.1 |
| | Belgium | 24.7 | 27.5 | 32.3 | 13.4 | 2 |
| | France | 31.8 | 30.6 | 26.9 | 5.8 | 4.8 |
| | Germany | 32.9 | 33.1 | 22.5 | 7.7 | 3.8 |
| | Luxembourg | 30.5 | 18.3 | 37.1 | 11.9 | 2.2 |
| | Netherlands | 12.6 | 23.1 | 38.0 | 12.2 | 14.1 |
| | Switzerland | 31.3 | 11.1 | 24.8 | 7.5 | 25.3 |
| **South Western** | Greece | 37.4 | 23.2 | 11.4 | 3.8 | 24.2 |
| | Italy | 34.5 | 32.2 | 15.4 | 7.8 | 10.1 |
| | Portugal | 44.2 | 17.6 | 15.1 | 6.2 | 16.9 |
| | Spain | 36.7 | 28.0 | 13.9 | 3.9 | 17.4 |
| **Average** | | 33.4 | 19.4 | 22.3 | 6.1 | 18.9 |

* Includes wetlands, shrubland, bare land, water bodies and other semi-natural areas; Sources: [6] for EU members (data for 2012); [7] for Finland (data for 2010); [8] for Norway (data for 2011); [9] for Switzerland (data for 2009).

The objective of this review paper is to examine soil degradation problems in WE, along with the different strategies and policies implemented to protect soil (with special focus on agricultural and forest soils) and future perspectives, in light of present legislation and technical support to soils. With this review, we intend to provide an up-to-date summary of soil degradation, soil management, soil quality (SQ) and SQ evaluation in the region.

### *1.3. Soil Degradation Issues in Western Europe*

Damage to Europe's soils from modern human activities increased in the second half of the 20th century and led to irreversible losses due to a number of causes, which vary in importance and intensity across WE [12]. These include increasing demands from almost all economic sectors, mostly agriculture and forestry [13], but also households, industry, transport and tourism.

The European Environment Agency (EEA) and the Joint Research Center (JRC) of the European Commission have published numerous papers and reports describing soil degradation problems in Europe, in some cases, with special emphasis on WE (e.g., [11–13]). From a general perspective, soil degradation problems can be classified into four major groups: chemical, physical and biological degradation (including soil organic matter decline), and soil loss. Land-use changes can be considered as a cross-cutting factor that also affects soils. The following sections explain the present situation in relation to these problems in WE. It is important to note that many soil degradation problems usually occur together in many areas of WE [14].

The ENVASSO Project (Environmental Assessment of Soil for Monitoring), which involved 37 partners drawn from 25 EU Member States, represents a significant step in the identification of these problems and in the quantification of their spread and importance [15]. The aim of this section is not to repeat this information, but to provide a summary of these problems in WE, adding up-to-date information and giving significant and recent examples.

#### 1.3.1. Chemical Degradation

The three major problems of chemical degradation in WE are soil contamination, soil salinization and acidification, and nutrient depletion [14]. The causes of these problems are varied, as are their relationships with agricultural management.

When addressing soil contamination, distinction must be made between local soil contamination (contaminated sites) and diffuse contamination over large areas [11]. For local contamination, in 2003, the European Environment Agency (EEA) reported soil contamination as a growing problem in WE, despite the existence of national and international legislation controlling sources of contamination and waste management [13]. A recent review on contaminated sites in Europe identified most of these as being located close to landfill sites, industrial and commercial installations emitting heavy metals, oil installations and military camps [16]. Mineral oil and heavy metals are the main contaminants, with the metal industries, gasoline and vehicle service stations being reported as the most frequent sources of local soil contamination [16,17]. The most recent available EEA report on the management of contaminated sites shows that the sizes of these vary widely across WE [17]. While in the UK, industrial waste treatment accounts for 31% of the contaminated sites, this is reduced to 20% in Italy and 0% in The Netherlands. On the other hand, municipal waste treatment and landfill sites account for just 1% in the Netherlands and up to 41% of contaminated sites in Switzerland.

The significance of this problem also differs between WE countries. The Netherlands, Belgium (Flanders), Denmark, France, Germany and the UK all have more than the EU average of 2.46 identified contaminated sites per 1000 inhabitants [16]. The corresponding figures are much lower in Greece, Norway, Ireland and Italy. This is clearly related to past and present industrial and commercial activities in these countries. Although reclamation and remediation of these sites has increased significantly in recent years, many potentially contaminated sites are still not clearly identified as such [11]. In addition to the references given, detailed information for most WE countries can also be found in [18].

In general, agriculture and agricultural soil management are not related to this type of contamination. Data on diffuse contamination, which is in many cases related to agriculture, are scarce and inaccurate owing to the lack of harmonized requirements for gathering this type of information in the different countries [11]. The overuse of plant protection products and fertilizers are usually highlighted as significant sources of diffuse soil contamination associated with agricultural production in WE [19]. Unlike in Eastern Europe, the use of fertilizers generally decreased in WE (in ton $ha^{-1}$ and total ton) during the last decade (2000–2012). Much of this decrease is due to the implementation of legislation to prevent contamination of fresh water due to agricultural activities in the EU (such as the Nitrate Directive 676/1991 and the Water Framework Directive 60/2000). However, the rates of application of N and P vary widely between different regions. The highest average inputs of N and the highest share of manure to the total N fertilizer application have traditionally been observed in

The Netherlands and Belgium (>300 kg/ha in 2008) [20]. On the other hand, Spain and Portugal, while using only an average of 89 and 79 kg N $ha^{-1}$, respectively, in 2008, used less manure, although the amounts appear to be increasing. Differences in the way the data are reported by each country make straightforward comparisons difficult. The variability within countries is also high, with irrigated agriculture accounting for much higher N and P doses than dryland areas, especially in the Mediterranean region [21].

Another source of diffuse soil contamination in agricultural soils is the use of sewage sludge. Since 1986, the use of sludge as a soil fertilizer has been regulated in the EU by a specific directive (Directive 278/1986/CEE). In relation to the risk of soil degradation, this Directive mainly focuses on heavy metal concentrations. However, since publication of the Directive, more stringent legislation has been adopted by several European countries for sludge disposal on soil, with lower limits established for heavy metals and limits also established for pathogens and organic pollutants [22]. There is currently increasing concern about the presence of emerging pollutants in sewage sludge that may contaminate soil when used in agriculture, and this should result in an up-grade of this legislation in the near future (see for instance [23]). Nonetheless, in WE, the prevailing destination for sewage sludge is recycling in agriculture [24], which accounts for 44% of the total sewage sludge production overall, although this varies in different countries. For instance, the proportion is >60% in Portugal, UK, Ireland and Spain, and <30% in Germany, Sweden, Italy, Austria and Belgium. In addition, sludge management practices vary greatly between regions within the same country [22]. These differences arise from local political, social and legal conditions, such as the adaptation of legal restrictions on toxic element concentrations. Future trends in the WE seem to be for stable or increased agricultural use as the most frequent option, although some shifts may be seen as biogas and energy production from sludge is also a current trend. In relation to the potential effect of this practice on soil contamination, several local scale studies have shown different trends depending on soil type, agricultural or forest management, *etc.* (e.g., [25]). On a regional scale, a study involving the distribution of heavy metals in European soils showed that concentrations of Cd, Cu, Hg, Pb and Zn were closely correlated with agricultural practices and some parent materials [26]. In France, diffuse contamination with heavy metals was identified close to industrial sites and also associated with sewage sludge amendments carried out in agricultural areas before the present legislation was implemented [27]. In Mediterranean Spain, Co, Cr, Fe, Mn, Ni and Zn in agricultural soils have been associated with parent rocks, while Cd, Cu and Pb have been related to human activities [28].

In WE, **soil salinization** (understood as the accumulation of soluble salts in soils as a result of human activities) is mainly of concern in the Mediterranean region, where it is most frequently caused by inadequate irrigation techniques, including the use of saline water or salinized groundwater and/or poor drainage conditions [13]. In contrast to other soil degradation problems, the thresholds and baselines of salt concentrations used to assess salinization are well defined and almost universally accepted, because of the importance of this issue for the development of irrigated agriculture (e.g., [29,30]). A map of salt-affected soils in the EU reveals that these are particularly important in Spain and Greece [31]. Coastal areas of France also have a high proportion of saline soils. However, the map does not indicate which of these areas are naturally saline or have soils with poorly soluble salts such as gypsum. There is evidence that at least some parts of the Ebro Basin in Spain and smaller regions of Italy, Greece, Portugal and France have been salinized due to improper irrigation strategies; however, data on present trends in this problem are not available on a continental scale [11].

Finally, **soil acidification** may occur as a result of atmospheric acid deposition and/or the use of acidifying amendments. No systematic national and continental-level studies on soil acidification are available for non-forested soils [11]. Acid deposition has decreased drastically in WE since 1980 ([13,32]). However, the effect of this reduction on soil acidity and acidification is not evident, because while some studies report declining levels of acid (see [11]), others indicate no or very slight reductions in acidity despite much lower rates of acid deposition.

In forest soils, **nutrient depletion** due to intensive soil management has been reported in several areas of WE [33]. The depletion depends on the level of biomass removal [34]. For instance, in weathered, acidic soils with low reserves of nutrients, stem-only harvesting in *Eucalyptus sp*. stands was found to involve the export, every 15 years, of more than 80% of the nutrients available in the soil [35]. Stem-only harvesting of *Pinus radiata* and *Pinus pinaster* in Southern Europe also involved high exports of K, Mg, P and Ca, leading to losses of 60%–100% of the soil available stores [35]. In a study in the UK, it was concluded that the removal of N, P and K in the tree biomass by whole tree harvesting was three to four times greater than by stem-only harvesting of the first rotation of Sitka spruce (*Picea sitchensis* L) [36]. It was also observed that after 23 years of growth of the next rotation of trees, the plots where whole trees were harvested had a significantly lower basal area on average [36]. Furthermore, the removal of tree stumps and coarse roots from felling sites as a source of woody biomass for bioenergy generation is being established in parts of WE such as Aquitaine (France) [37]. However, harvesting roots may be unsustainable if soil fertility is reduced, with consequences for future forest production [37,38].

### 1.3.2. Physical Degradation

**Soil compaction** has been widely studied in WE. It affects the air capacity, the permeability and the water-holding capacity of soils, as well as root development and soil biological activity, and it has therefore been observed to determine plant growth and agricultural yields [39]. The two most significant human activities responsible for soil compaction in Europe are agriculture and forestry [40]. Two major causes are identified [41]: ground pressure from machinery and/or animals, and soil management in agricultural land (including tillage systems). A complete survey of the surface affected by compaction has not yet been conducted in WE. The most important work is the elaboration of a map showing soil susceptibility to compaction [42]. This map shows that some areas of Belgium, NW France and The Netherlands are highly or very highly susceptible to compaction, although parts of England and South Scotland in the UK, and some Mediterranean areas such as the Ebro and Guadalquivir basins in Spain, and the Veneto region and some parts of Lombardy and Piedmont in Italy are also affected [13]. An earlier report declared 37% of European soils as being highly or very highly sensitive to compaction [43]; however, the map was created using pedotransfer functions, and it must therefore be interpreted with caution. In addition, many of the susceptible areas in the map correspond to peatland and other types of soil that are not cultivated or are managed with heavy machinery [44]. Nonetheless, the increasing weight of agricultural machinery, the introduction of irrigation and the use of farm equipment when soil water content is high suggest that some WE agricultural soils will be increasingly compacted to ever-greater depth [44].

Forest management can result in significant compaction problems because of the weight and size of forest machinery. In planted forests in WE, different rotation schedules involve more or less frequent use of heavy machinery. *Eucalyptus* spp., which occupy 1.5 million ha of land in the Iberian Peninsula (Spain and Portugal), are cultivated through a coppice system in short rotation forestry (10–12 year rotations), generally for three consecutive rotations. Poplar (*Populus* spp.) plantations, which are mainly found in France (230,000 ha) followed by Italy, Germany and Spain, cover between 100,000 and 125,000 ha of land [5]. This species, which has a deep root system and requires rich soils and large amounts of water [45,46], is usually also managed intensively in short rotations (12–16 years), with weed control techniques (mainly surface ploughing) used regularly during the first six years. Sitka spruce (*Picea sitchensis* L) plantations, mainly located in the UK and Ireland (1.2 million hectares) [47], are typically grown on 35–45 year rotations, but rotation lengths of 25 years have been proposed. Among pine species, the rotation lengths of *Pinus radiata* are typically between 35 and 40 years and may include both pre-commercial and commercial thinning and mechanical weed control. *Pinus radiata* plantations are mainly located in N Spain (290,000 ha) [48]. After clear-cut felling, the trunks are harvested with the help of skidders that are sometimes driven over the plantation area. The conventional method of site preparation consists of the partial removal of logging residues followed by down-slope ripping or blading, which consists of pushing the logging residues and the humus layer away from the site [49]. *Pinus pinaster*, which covers 2.6 million hectares of land in Portugal, NW Spain (Galicia) and SW France (Aquitaine), is characteristic of an Atlantic climate [50] and is well adapted to sandy soils [51]. The rotation lengths are typically between 30 years in NW Spain and 45 years in France and always include thinning and mechanical weed control. These management techniques have consequences for both soil compaction and erosion risk (see Section 1.3.4). Although the effects are assumed to be most pronounced on clayey or loamy textures [52], it has been suggested that a single pass of a harvester is enough to induce a large increase in bulk density and penetration resistance in sandy soils [53].

Finally, another important factor in relation to the soil physical status is the **loss of structural stability**, which can also favor erosion and greatly reduce soil porosity and the capacity of soil to store and conduct water. In many agricultural soils in WE, this is also related to the formation of crusts (e.g., [28]). However, this topic is seldom addressed as such on regional, national or continental scales in WE Europe. This is probably because soil structure is considered as a diagnostic soil property related not only to physical degradation, but also to chemical and/or biological problems, such as organic matter decline and salinization [54]. The risk of soil crusting has only recently been addressed in relation to potential wind erosion [55]. Sandy soils, which are common in the glacial deposits of Denmark, N Germany, the Netherlands, Scandinavia and the Baltic area, as well as in some areas of the NW of the Iberian Peninsula and SW France, are less affected by the formation of a soil surface crust.

1.3.3. Organic Matter Decay and Soil Biological Degradation

Soil organic matter (SOM), in particular organic C (SOC), has been in the spotlight of soil research for decades. At the European level, an overwhelming amount of research on SOC storage, gains and losses in soils has been conducted on different scales. However, the high variability and diversity of data make comparisons difficult [56]. A general view on the average content in SOC of European soils is that most of South Europe is covered by soils with less than 2% SOC [57]. This is related to both climate and historical land use. Many areas of France fall also below this threshold. The average SOC contents are higher in northern countries, the UK and Ireland.

The reasons for the generally observed **decline in SOC** in agricultural soils in WE Europe have been summarized [15]. These include conversion of grassland, forests and natural vegetation to arable land, deep ploughing of arable soils, intensive tillage operations, overfertilization [11], drainage, liming, fertilizer use and tillage of peat soils, crop rotations with reduced proportion of grasses, soil erosion, and wildfires. The latter two are of particular importance in Mediterranean countries [58].

At a national level, some long-term studies have reported changes in the SOC contents of agricultural soils. For instance, losses of 0.5–2 g SOC/kg soil per year were observed in England and Wales between 1973 and 2003 [59]. A large-scale inventory in Austria revealed that croplands were losing 24 g C/m$^2$ annually [60]. In S Belgium, losses of 0.12 t/ha per year were reported for croplands, but with an increase 0.44 t/ha in grasslands between 1955 and 2005 [61]. Grasslands on sandy soils in the Netherlands displayed a non-homogeneous trend, with some gains and some losses of SOC between 1984 and 2004. Continuous maize crops on the same soils systematically lost SOC in the period mentioned [62]. A slight average increment of 0.10 and 0.08 g SOC/kg soil in grasslands and arable land was reported for the same period [63]. In France, long-term observations (e.g., [64]) show decreasing stocks in many regions, because of deforestation, conversion of grassland into cropland, increasing cropping intensity or climate change. Vineyards and arable land display the lowest SOC contents overall [28]. An overall decrease in SOC was also recently observed in Bavarian cropland, although the variability was high, with some plots showing no change or a net increase between 1986 and 2007 [60]. This was also reported in France, where some intensely cultivated areas showed stable or slightly higher SOC stocks over time [28].

Despite these regional-scale studies, consistent figures for SOC stocks and how they change at European level are still scarce [65]. The interaction between SOC and climate change is an important issue that complicates predictions about SOC changes in relation to future land-use changes in WE in [66]. Recent simulations predicted an overall increase in this pool in agricultural soils in Europe, with a non homogeneous distribution [66], including C losses in the South, which could be compensated by a gain in Central and Northern regions. This model also showed pastures in the UK, Ireland, the Netherlands and France as the dominant SOC reservoirs, while permanent crops (olives, vineyards and orchards) accounted for only 3% of the total SOC stock, despite being widespread in Southern Europe. Arable land was predicted as containing 43% of the total stock of C, while it represents 53% of the total agricultural surface. In forest soils, harvesting activities and site preparation may lead to the removal of the humus layer from more than 80% of the surface [67].

Change in **soil biodiversity**, understood as the variety of all living organisms found within the soil system, is directly related to soil degradation in WE [68]. Some authors have suggested that it is essential to establish the present extent and distribution of soil biodiversity and to identify current threats [69]. The strong correlation between these threats and problems related to soil degradation problems described in the study becomes evident as the major challenges to soil biodiversity in Europe are land-use changes, intensive human exploitation of soils, soil compaction, soil erosion, soil organic matter decline and soil pollution [69]. Other issues of importance are invasive species and the use of genetically-modified crops (GMCs), climate change, salinization, desertification and wildfires [69–71]. The intensity of land exploitation has been identified, both in terms of agricultural use intensity and land-use dynamics, as the main factor affecting soil biodiversity in the EU [71]. The extent and intensity of these factors enable identification of the areas of WE most at risk. The areas at high, very high, and extremely high risk are concentrated in the UK, the Netherlands and Belgium, where almost 100% of the territory appears within these categories in different maps [69,71]. Most of Central and Northern France, Denmark and Germany also fall within these categories. This is attributed to the combined effect of intense agriculture, with a relatively large number of invasive species and an increased risk for the soils to lose organic carbon. In general, Mediterranean countries and areas of Southern Europe dedicated to intensive agriculture are at a lower risk. However, some areas of Italy and Spain under intensive land-use have been identified as being at high risk of suffering a decline in soil biodiversity. A recent modeling study of the susceptibility of European soils to antibiotic contamination from cattle also shows an uneven distribution in WE [72]. The Netherlands, Ireland and Belgium displayed by far the highest risk, while the risk was much lower in Mediterranean countries.

These characteristics refer to the relative pressure exerted to soil biodiversity, but not to the actual state of soil biota. At the national level, some WE countries have carried out a systematic evaluation of soil biodiversity in national soil monitoring networks [70]. These countries include France, Germany, the Netherlands, Switzerland, Ireland and the UK [11,70,73,74]. Diverse types of soil organisms are targeted in these studies, and data on the changes in soil biodiversity are scarce. Earthworms and soil fungi are some of the most commonly studied organisms [11]. The abundance and diversity of earthworms and other soil organisms have been found to be related to land use (dairy farms display the highest numbers and arable land the lowest among non-natural sites) and soil type [75]. In France, most of the soil biological groups exhibited lower values of abundance and community richness in cropland than in meadows [76]. Within agricultural land, the intensity of the management system also affected most biological soil properties; however, the type of tillage, fertilization and pesticide use were only related to the total microbial biomass and earthworm diversity, which were lower in sites in which fertilizer use is restricted, ploughed soils and sites with high inputs of pesticide. The use of fungicides and herbicides generally increased between 1992 and 2003 [77], but their use has decreased in most WE countries [78] following the adoption of strategies encouraging low-input or pesticide-free cultivation to reduce the risks and impacts of pesticides on the environment (EU Directive 128/2009). National action plans are being developed in most WE countries following this Directive [11]. It appears from these data that the relationship between land-use and soil biodiversity is a much-needed but still pending topic in WE Europe [74].

#### 1.3.4. Soil Loss

The two major processes that cause soil loss are soil erosion and landslides. Although both have been clearly identified as soil degradation problems in WE [11], the nature, potential remediation and severity of both are rather different, and their extent and degree of significance in Europe also differ.

Soil erosion is a key environmental problem that has long been recognized in WE. The first extensive essay on soil conservation known to the Western world was published in Germany in 1815 [79]. The paper focused on depicting the most appropriate methods of preventing soil erosion in mountainous arable land. Since then, an enormous body of work has been developed in WE, as elsewhere, with the aim of understanding, defining, modeling, measuring and preventing soil erosion (see e.g., a comprehensive review of keynote references in N Europe in [80]). Nonetheless, soil erosion by water remains one of the most widespread forms of soil degradation in WE, especially throughout Mediterranean Europe [11,13]. Soil erosion has also traditionally been strongly associated with desertification. Desertification does not only affect soil and is now perceived as a wide-ranging issue associated with the loss of SOM, salinization and other soil degradation problems in WE [15], where it mainly affects the Mediterranean countries (Spain, Portugal, S France, Greece and S Italy).

Estimated mean annual erosion rates are higher than 6 t/ha in Italy and Portugal, between 4 and 6 t/ha in Austria and the UK, and between 2 and 4 ton/ha in Spain, France, Luxembourg, Belgium and Germany [81]. However, these data are average values of very different patterns within each country. In Portugal and Italy, for instance, some regions display average rates of >20 ton/ha and others of <2 ton/ha. This leaves few regions in WE with average erosion rates below the threshold of 1 t/ha/year, which has been repeatedly cited as the safe or tolerable value for soil erosion [11,41,82,83]. If non-arable land is excluded from the analysis, the highest rates correspond to Central-East and S Italy, S and SW Spain and N Portugal, SW France and Brittany. Of these, only 2% of the areas suffering from moderate to severe erosion are permanent grassland and pasture in the EU. The trend in the period 2000–2006 was towards no change in potential erosion rates in WE, except for some areas of Italy, Portugal, Austria and the UK (the countries with the highest average erosion rates) in which increments have been detected [81]. These data may be underestimated because of the short time period considered. It is also important to note that all these data are estimates derived from erosion models (in some cases RUSLE (e.g., [84–87]) and also PESERA (Pan–European Soil Erosion Risk Assessment [88]), and they should therefore be considered as such and not as actual measured data.

A review of the relationship between land use and actual erosion rates using measured erosion in plots at the country level confirmed the dominant influence of land use and cover on soil erosion rates in many countries in WE [89]. This study concluded that high erosion rates occurred in hilly loess-rich areas of West and Central Europe (2–10 ton $ha^{-1}$ per year) and in agricultural areas in the piedmonts of the major European mountain ranges. Within agricultural land, cropping systems in which soil lay bare for long periods displayed the highest erosion rates. The period of year when soil is covered by crops is therefore an indicator of erosion problems [90]. The magnitude of erosion varies greatly across WE Europe, mainly due to climatic differences that induce different agronomic practices. For example, the percentage of soil covered by green crops on arable land varies from 25% to 50% in Spain, Italy and Southern France, and to more than 50% above a latitude of 46°N. Conversely, bare soil, which accounts for 20%–30% of arable land in France, Spain and Italy, represents less than 10% in the UK [90]. In areas of Southern Europe dominated by vineyards, in which the soil is often bare, erosion rates are high (12.2 ton $ha^{-1}$ per year but with a great variability (standard deviation = 27.8 ton $ha^{-1}year^{-1}$). Measured erosion rates on arable and bare land were also related to slope steepness and length and to soil texture, while this was not the case for plots with permanent land cover [90].

From the above-mentioned measured data, estimated average rates were calculated per country and land use [89]. The rates were lower than those calculated using model simulations (Denmark and Italy were the only two countries in WE with rates >2 ton $ha^{-1}year^{-1}$). In particular, erosion rates in the Mediterranean were much lower than predicted, mainly due to the stoniness of many Mediterranean soils [84]. Areas of intense erosion were, however, found in Spain (in the Guadalquivir and Ebro river basins) and Italy (Apennines and Sicily).

In the Mediterranean region, wildfires are of particular concern in relation to soil erosion [13,83]. The characteristics of post-wildfire erosion in Mediterranean countries have been summarized, showing a strong influence of the topography, slope orientation and erratic rain distribution in the observed erosion rates at different observation points in Mediterranean WE [58]. Peak and average erosion rates were reported to be similar to or lower than those reported for fire-affected soils elsewhere or even those reported for common land uses in this region. Human-induced soil degradation since prehistoric times was suggested as the most likely factor explaining low erosion rates. Human occupation, abandonment, forestry use, *etc.*, have resulted in shallow, stony and weak soils.

In general, the loss of soil in forestry operations has also been related to a significant decrease in the water holding capacity [49]. Thus, although forest roads are essential structures that provide access to forests for wood extraction, their construction is the most destructive operation in the forest environment, causing soil compaction, increased surface run-off and soil erosion. Studies in which the degree of soil disturbance has been directly evaluated have shown that mechanized labor typically produces alteration of more than 80% of the soil surface, basically through the removal of the topsoil, compaction and soil displacement in *Pinus radiata* plantations in N Spain [67]. In the same region, the use of heavy machinery increased the erosion rate from 15 to 1600 kg $ha^{-1}$ $year^{-1}$. Other types of soil erosion that can affect or have affected soils in WE are wind erosion and tillage erosion (the displacement of soil masses through intense tillage). Wind erosion has been less well studied in WE [11,81]. However, soils are highly susceptible to wind erosion in N Europe (Belgium, the Netherlands, N Germany and Denmark) and some areas of SW France and Spain [55]. This has been related to a lower tendency of soils in these regions to form crusts that would prevent wind erosion (see above). Tillage erosion has recently been studied, and although available data are scarce, studies in Europe highlight the importance of the magnitude of tillage erosion relative to water erosion, with mean annual rates in the order of 3 ton/ha for Belgium, northern France, and eastern England [82]. Tillage erosion rates were measured in an experimental field in Belgium and were found to be a more important soil redistribution factor than water erosion at present [91].

In summary, soil erosion remains a matter of concern in WE, where it is related to natural characteristics, land use and soil management. However, despite significant efforts form European institutions (e.g., [43]), harmonized data on actual erosion rates for the European continent are not yet available [11,58]. Identification of vulnerable areas that are affected by erosion is required in WE [82,87]. Current efforts on standardizing national erosion measures and estimates show discrepancies between countries, models and actual plot data [86] that must be overcome for better assessment of soil erosion and soil erosion risk in WE.

**Landslides** represent a serious natural hazard in many areas of WE and particularly affect the mountainous regions (Alps, Pyrenees, Apennines, and other). Although the total affected area in Europe is not known, the Italian national database states that more than 400,000 ha of land in Italy is affected [11]. From the point of view of soil degradation, landslides represent dramatic but highly localized problems of soil loss.

#### 1.3.5. Land-Use Changes

In addition to changes within agricultural and forest systems, land take and agricultural land abandonment can also have important effects on soils.

**Land take** can be defined as the loss of agricultural land to non-agricultural sectors. It represents an increase in settlement areas (or artificial surfaces) over time, usually at the expanse of rural areas [92]. Land take has increased significantly in WE in the last few decades [93]. In the EU, about 1000 $km^2$ of land is taken for housing, industry, roads or recreational purposes every year [92]. In WE, the countries in which the largest areas of agricultural land were converted to artificial surfaces between 2000 and 2006 were Spain (104,706 ha) and France (76,000 ha). In the same period, the highest proportion of the total area of agricultural land converted (1.42%), mostly for construction work, occurred in The Netherlands [94]. A high percentage of land was converted to alternative uses in 2012 in Belgium and the Netherlands (13.4% and 12.2%, respectively), which is directly related to the high population densities in these countries [95]. Many aspects of the consequences of such change are serious. Thus, although soil sealing is the most evident consequence, land take also affects soil erosion, soil productivity, water storage and biodiversity in neighboring areas, as it interrupts the exchange between soils and other ecological compartments [92]. Soil sealing has been described as the destruction or covering of soils by buildings, constructions and layers of completely or partly impermeable artificial material. It has also been defined as a process that changes the nature of soil so that it becomes impermeable. Soil sealing is the most intense form of land take in WE and is essentially an irreversible process [92]. The effects of soil sealing are much more complex than those of other soil degradation issues because the process affects other ecological compartments (biosphere, atmosphere and hydrosphere) apart from soils [92]. In addition, the drivers and potential control of soil sealing are beyond the reach of soil conservation policies and must be addressed in an integrated approach implemented at all policy levels [11,13]. Understanding the potential services that urban and other sealed soils can provide is undoubtedly a first step [96].

**Agricultural land abandonment** has occurred as the result of the decline in the viability of extensive systems in some areas. This process has mainly affected farmland in marginal land, such as cold, wet and/or mountainous areas of WE [97]. In France and the UK, it is a marginal process, except in French vineyards (11% of the surface lost between 1990 and 2000). The rates of agricultural land abandonment are higher in Italy and Germany (2% of the total utilizable agricultural area, and 250,000 ha, respectively). It has recently been estimated that around 8% less land will be farmed under the new trade policies in the EU, if agricultural subsidies are further decoupled from production, which will particularly affect livestock grazing farms [98]. Regions where agriculture is limited by natural climate and/or soil limitations, such as some areas of Finland and Sweden, the Pyrenees, NW Spain and Portugal, the Massif Central and Brittany in France, the Apennines (Italy), the Alps, and other upland areas of Germany and arid zones of SE Spain, would be the most strongly affected. It has also been argued that there is reasonable evidence from trends in the drivers of abandonment that significant amounts of farmland will be abandoned in Europe over the next few decades, particularly in extensively grazed areas [99]. Eurostat identifies a higher risk of land abandonment in Southern states (Portugal, Spain, Italy and Greece) and also in some areas of N Finland and Sweden and NW Ireland [100].

The consequences of land abandonment on soil resources in WE are not yet clear. On the one hand, land abandonment has been associated with soil degradation issues such as wildfire and erosion in S Europe and biodiversity loss in N Europe [98]. On the other hand, it can provide an opportunity for restoration of natural habitats. The final balance seems to be both site and time-dependent. For instance, it has been noted that erosion problems tend to decrease and disappear as vegetation colonizes cropland following abandonment of sloping land [97]. The same study also established that the degree of intensification of agriculture before abandonment is also important: abandonment of intensively managed farmland has more positive effects as it reduces soil erosion and use of pesticides and fertilizers to a greater extent than abandonment of extensively managed farmland. In semi-arid mountainous areas of Mediterranean Europe, land abandonment has been shown to reduce soil loss and sediment delivery in surface water flows, despite slower plant colonization. When the abandoned farmland comprises terraced areas, this trend can be reversed, as small-scale landslides and erosion may increase as terraces collapse [101,102]. Slow recovery of soil properties has been noted in abandoned vineyard soils in semi-arid Spain and attributed to intense soil degradation under the vines [103]. In the Alps, small differences have been observed in the soil biological properties of abandoned land and organically fertilized meadows and grasslands [104].

Finally, a particular case of land-use change can be observed within agricultural land upon **irrigation adoption** (e.g., [105–107]). Overall, the socio-economic relevance of irrigation is considerable in WE [108]. In France, Greece and Spain together, the total area of irrigated land was 7.4 million ha in 2000, which represents an increase of 28.8% between 1990 and 2000 [109]. Across the entire European continent, irrigated land increased from 14.5 to 25.2 million ha in 2003 [110]. In 2007, more than 30% of the total area occupied by agriculture in Italy and Greece was irrigated. The proportion in other Mediterranean countries such as Spain and Portugal was close to 15%, and Northern countries such as the Netherlands and Denmark it was also >15% [111]. In 2011, irrigated land already represented 13% of the agricultural land in Central and Western Europe [112]. In Spain, 20.2% of the agricultural land was irrigated, which in absolute terms (in total, an irrigated area of 3,780,000 ha) represents the largest area of irrigated land in WE [113].

According to available data in Eurostat, the trend is for irrigation land to be stabilized in WE. In the Mediterranean region, present irrigation projects also include improvements in terms of water-use efficiency and soil conservation [108]. However, the environmental impacts of irrigation are variable and still poorly documented in many WE countries [108]. In relation to soil degradation, problems involving erosion and salinization are most often associated with irrigation in Southern Europe [108,113–115], although some cases of organic matter loss have also been reported [105]. Recent studies have shown that deficit irrigation strategies must be implemented with caution because they may lead to soil salinization and sodification problems, especially when moderately saline waters are used [116,117].

## 2. Mitigation of Soil Degradation in Agricultural and Forest Soils in Western Europe

Different strategies promoting management systems that attempt to reduce or mitigate soil degradation in agricultural and forest areas have been adopted in WE. The degree of implementation and the effectiveness of these strategies vary greatly between countries. In the EU, the European Parliament launched a pan-European project in 2007 to evaluate the status of soil degradation and appropriateness of relevant policy measures and the so-called "soil friendly farming practices". This project, which remains one of the most comprehensive studies of sustainable agricultural management at the EU level [118], summarized the actual and potential effect of some farming practices on soil degradation processes. The study found that actual and potential effects of farming practices on soil degradation depend on the type of degradation considered and that the implementation of farming practices is a complex and site-specific process that requires the cooperation of farmers and depends on local conditions (soil and climatic). These factors determine the potential success of changes in farming practices in the prevention and mitigation of soil degradation. A pan-European meta-analysis has recently revised the effects of some sustainable farming practices on soil chemical, physical and biological properties, also concluding that the success and adoption of such strategies is highly site-dependent [119].

### *2.1. Conservation Farming*

Conservation farming (CF) comprises a series of techniques granting minimum mechanical soil disturbance, permanent soil cover with crop residues and diversification of crop species grown in the same field [120]. These techniques have been reported to reduce soil degradation in different agroecological situations (and in some cases are designed for this purpose). As excessive tillage and/or crop residue removal and uncontrolled grazing have been the cause of many soil degradation-associated problems, CF seems an appropriate strategy for solving such problems. Statistics on the share of conventional, conservation and zero tillage on the total arable area and the tilled arable area show that CF techniques are less frequently adopted in Europe than in other areas (especially the US and Latin America), and reduced tillage is more common than no-tillage (NT) and the use of cover crops [121]. The uptake of NT varies widely in WE; for example, in 2009 it varied from 4.5% to 10% of total arable land in Finland and Greece and from 2.5% to 4.5% in Spain and the UK [118]. Reduced tillage was practiced on 40%–55% of the arable land in Finland and the UK, and on 20%–25% of arable land in France, Germany and Portugal. Although the major driver for CF adoption in WE in recent decades has generally been the need to reduce costs in extensive agriculture [119,122], soil management and soil degradation issues have been given as reasons for both adopting and abandoning CF in different areas. In Northern Europe (including France), soil erosion, soil crusting in loamy soils and the need to increase SOM and soil trafficability are widely cited as reasons for adopting CF [123]. In the Mediterranean countries, soil water storage and water-use efficiency can be added to this list of reasons [124]. Different studies show that the effectiveness of CF in solving these problems is highly site-dependent because the contrasting soil and climate types existing in WE exert a strong influence on the success of these techniques [119,125]. Numerous studies have examined the effect of CF on soil properties across WE. Some general trends can be drawn from reviews and pan-European projects conducted on the topic in recent years.

There is some agreement about the positive effect of CF in reducing erosion in WE [119,121]. The most widely reported benefits of CF in relation to erosion are the increased soil infiltrability and/or the protective effect of crop residues on the soil surface. This is of particular importance in Southwestern Europe [125]. The capacity of reduced tillage and NT to increase SOC stocks is currently under discussion [126]. SOC increments have been reported under NT in many WE countries (see [125] for cases in UK, Spain, Portugal, France and Germany). In one study, NT increased SOC concentrations and SOC stocks by on average 3% and 7%, respectively, although differences were observed depending on the soil type and climate area [119]. Other studies reported no significant differences in the long term (e.g., in Scotland and Switzerland [125], in NE Spain after 20 years [127], and in N France in a 41-year experimental trial [128]) or inconsistent results (e.g., in Denmark [129]). This has been attributed to CF favoring stratification of SOC over accumulation in depth [130], or to NT being able to stock more SOC only when it induces higher yields and inputs of crop residues to the soil [131]. Climate and soil constraints may therefore modulate the effect of CF on crop yields and SOC accumulation in WE [125]. Other cultivation practices associated with low intensity soil management have been observed to have a greater potential for SOC accumulation than NT in WE. For instance, ley cropping systems and cover crops have been shown to perform better than straw incorporation and reduced tillage [66].

The effects of CF on physical properties have also shown variable results across WE. The soil water-retention capacity has been observed to be greater in semi-arid land under NT in Spain [132,133] and other Mediterranean countries in WE [124], but not in Germany [134]. In general, increased infiltration rates under NT have been reported [125], although this seems to be related to the type of soil and to the presence and activity of earthworms. It has been reported that NT induced soil physical degradation in some cases (higher bulk density, lower aggregate stability and permeability [119]). Soil compaction, usually understood as increased bulk density relative to that of non-CF soils, has generally been reported during the first years of NT and reduced tillage adoption, especially in Germany and Scandinavia [135,136], although it sometimes seems to decrease over time [125]. This problem is directly related to waterlogging and soil ponding in many areas in Northern and WE, for instance in Denmark and Germany [123] and Finland [137]. The positive effect of CF on the soil biological activity, especially earthworms, is generally observed all across WE [119,121,135,136,138].

The above-mentioned studies generally suggest that the adoption of NT and other CF techniques have had different consequences on WE depending on the type of soil, climate, the CF activities and the crops involved. When negative effects have been observed, these can be included in the reasons for fluctuations in the interest in CF shown by WE farmers [121]. The greatest soil limitations for CF adoption in N Europe are soil texture (sand and silt soils and heavy clays are difficult to cultivate without ploughing), soil drainage and the soil water regime (cold and wet conditions hinder organic matter incorporation into the soil and favor waterlogging) [121,122,125]. Also directly related to soils, reduced yields are also frequently cited as a reason for abandoning CF techniques, especially in Northern Europe [121]. A meta-analysis revealed that the effect of CF on crop yields in Europe is highly site-dependent, with NT tending to decrease yields to a greater extent than reduced tillage, and that the effect varies depending on climate, soil characteristics, type of crops and management under conventional farming [139]. Adoption of CF in the UK is mainly under systems of non-inversion tillage because the generally reduced yields under NT do not favor direct drilling [140]. In Mediterranean land, higher yields are commonly obtained under NT than under other tillage systems only in dry years [119,132,141].

Increases in weeds, pests or diseases, although not directly related to soil characteristics, are agronomic limitations also often cited as reasons for rejecting CF in WE. The combined use of CF techniques, especially NT, and intensive use of herbicides and other phytosanitary products (increasingly discouraged by European environmental and agricultural policies, see Section 3.2), together with the reduced acceptance of GMCs by European consumers are undoubtedly also related to the lower uptake of these techniques in WE than in other regions. Some authors acknowledge that future restrictions on the use of herbicides may deter NT adoption in WE [125]. However, others report cases of lower mobility and persistence of herbicides in soils under conservation tillage in Mediterranean Europe [124]. Finally, other circumstances not related to soil or environmental issues, such as the existence of subsidies, technical problems with weeds and soil management, scarce technical expertise, unfavorable market conditions and other socio-economic aspects are very often cited as explaining the lower use of NT and adoption of CF in WE [119,121,122,142]. For example, although CF adoption (especially the use of permanent soil cover [143]) in olive groves in Mediterranean land has proven to be efficient in reducing erosion and increasing soil fertility (e.g., [144]), it has been shown that the factors that determine the adoption of such practices include the socio-demographic characteristics of olive growers and the role of social capital as well as the characteristics of the olive groves [145].

Considering that CF is not equally suitable for all WE agrosystems [121], there is a need to define which regions are the most suitable for its implementation, which entails the need for soil databases and decision support systems [119,122]. Because of the complexity and site-specificity of CF implementation, training of farmers and the adaptation of CF techniques to local circumstances are still required in WE [118,119].

### *2.2. Organic Farming, Agroecology and Agroforestry*

Organic farming (OF) in WE is mainly regulated by national and regional rules issued from a common legislative framework within the EU (Regulations 834/2007 and 889/2008). OF is defined therein as a system of farm management […] that combines best environmental practices, a high level of biodiversity, the preservation of natural resources […] and production […] using natural substances and processes. These regulations clearly state that OF must adopt those farming practices that aid the conservation and improvement of soils, with special emphasis on organic matter management. Since the beginning of the 1980s, agroecology has emerged as a distinct conceptual framework with holistic methods for the study of agroecosystems [146]. Agroecology, which includes organic management of soils and other approaches, has been defined as a means of protecting natural resources with guidelines for designing and managing sustainable agroecosystems. It is difficult to obtain accurate data from farmers following these principles as a whole in WE, and therefore only OF data 15% can be used to analyze its potential impact in the region.

Organic farming has developed rapidly during recent years in the EU. According to Eursotat [147], in 2012 the EU-27 had a total area of 5.8 million ha cultivated as fully-converted organic land plus 4.2 million ha under conversion, representing an increase from 2.6 million in 2003 (2.8 including fully converted and in conversion). However, in 2012, the whole OF area represented only 5.7% of the total agricultural area in the EU. In absolute terms, the countries with the most extensive fully converted areas in 2012 were Spain (1.4 million ha), Italy (0.9 million ha) and Germany (0.9 million ha). In relative terms, in 2012, the importance of the OF sector was highest in Austria (18.6%), followed by Sweden (15.8%), Italy (15%) and much further behind by Spain (7.5%), Denmark (7.3%) and Germany (5.8%). The number of studies and reports on the effect of OF on soils in WE is much fewer than for CF. Effects on the soil organic carbon and soil biodiversity have been reported, although these were not consistent in all parts of WE [118]. A meta-analysis revealed that, on average, SOC was significantly higher in organic plots than in conventional plots, although some paired studies reported no differences [148]. Another study also reported that organic farms in Europe tend to have higher SOC contents and lower nutrient losses per unit of field area [149]. However, both studies report that the differences may be lower when expressed per product unit. Gains in organic matter under OF have been reported in many Mediterranean agrosystems (e.g., in olive orchards [150] and in dryland crop rotations [151]) and attributed to the use of organic amendments (e.g., [152]).

The effects of larger organic C stocks on other soil properties reportedly vary in different sites and for different crops. Organic fertilization (relative to mineral N) yielded significant increases in SOC, N availability, earthworm density and activity, microbial biomass and diversity in European soils [119]. Microbial biomass was also found to be significantly higher under organic than conventional management [118]. Other authors observed better structure and porosity in soils under OF than in conventionally cultivated soils in the UK, but this effect was found to be scale and time-dependent [153]. In a study evaluating the long-term effects of organic viticulture in France, the authors found that OF led to an increase in SOC, potassium content, soil microbial biomass, and nematode densities [154]. However, OF also increased soil compaction, decreased endogeic earthworm density and did not modify the soil micro-food web evaluated by nematofauna analysis. Similar values of organic matter were associated with OF and conventional management in agroforestry (*dehesa*) soils in Spain, where differences in the soil physical condition were more dependent on the soil type than on management [155].

In relation to OF and other low-intensity farming systems, changes in land allocation as a result of social and land-use policy changes has resulted in extensification of agriculture in some areas of WE, especially those where revenues from agricultural production are low and/or land costs are high and decrease because of lower demand [156]. Some of these low-intensity areas support agroforestry systems. In a modeling study of the environmental benefits in the Mediterranean and Atlantic regions of Europe, soil erosion and nitrogen leaching were found to be lower than in conventional cropping land, and carbon sequestration was enhanced [157]. In a review of the benefits of alley cropping systems combining agriculture and short rotation coppices by growing trees in agricultural sites in temperate Europe, these systems proved efficient for soil carbon sequestration, improving fertility, controlling erosion, storing water and regulating its quality, and increasing the overall productivity compared to conventional farming [158]. In a review of alley cropping agroforestry systems in Europe, other authors also reported overall increases in the soil organic C stocks [159].

The *dehesas* and *montados,* which are characteristic of Western Spain and Portugal, represent a particular case in agricultural land management. These systems, which are the most extensive agroforestry systems in Europe, integrate forestry (usually with evergreen holm oak (*Quercus ilex* L.) and cork oak (*Quercus suber* L.) with agricultural and livestock management practices [160]. They cover more than 3 million ha [161]. They are usually developed on acid, sandy, poor soils in semi-arid land, and yield significantly higher primary productivity than forestry or crops alone. Some authors found that soil fertility increased near the trees, with a significant increase in organic C, total N, available P, CEC and exchangeable calcium and potassium in *dehesa* systems in Central-Western Spain [161]. Similar systems in Sicily (Italy) have also been described as effective for maintaining relatively good soil condition [162]. They generally yield smaller erosion risks than croplands in the area, although they may be prone to severe erosion if stocking rates and/or tillage frequency become excessive [163].

### *2.3. Other Sustainable Crop Management Strategies*

A significant number of agricultural practices that may have positive impacts on soils have been developed in the last few decades. Some of these practices have been implemented to a variable extent and with variable success in WE. Among those not specifically included within CF or OF, ridge tillage, contour farming and buffer strips, bench terraces and subsoiling have been identified [118]. Cover and catch crops, and the efficiency of crop rotations have been studied together with NT and organic fertilization [119]. These practices may be beneficial for addressing different soil degradation problems, and they have been identified as affecting the soil physical, chemical and biological parameters at different levels in WE [118,119]. As a result, they can help to address some soil degradation and related environmental issues. Their adoption in WE is variable.

**Ridge tillage** may favor retention of water in the soil. It is frequently used for crops such as potatoes and beet that are not suitable for CF. Reduced forms of ridge tillage have been shown to improve the soil physical quality and favor N mineralization in Belgium [164,165]. **Contour farming** involves developing cropping practices along the field contours, especially in sloping areas. It has been shown to be an effective measure for controlling erosion in England [166], especially when combined with minimal tillage [167]. When including hedgerows, as in the Armorican Massif in W France, gains in SOC were also significant both under the hedges and upslope [168].

**Bench terraces**, which are mainly built to prevent erosion and to enable cultivation in steep slopes, greatly modify the soil profile by leveling the slope to the contouring strips. They are very common in many mountainous areas in WE, especially in Alpine and Mediterranean regions, where they constitute a historical and traditional landscape feature [169]. Despite differences due to the heterogeneity of soil-forming factors across the area, terraced soils have some common elements. They display improved water availability, together with better nutrient conservation, which are known to increase crop yield in arid or semi-arid environments. Improved soil structure, physical quality, fertility, organic matter and porosity have also been reported for terrace soils, and they are often of better agricultural quality than the surrounding undisturbed soils [169]. However, significant nutrient losses have been reported to occur from terrace taluses when these are not properly built or not protected by vegetation [170]. Bench terrace systems require intensive maintenance to retain their sustainability [101,118] because if abandoned they may be subjected to gradual decay due to erosion processes and slope failure resulting in the loss of organic C and soil fertility [171].

As an intermediate system between contour farming and bench terraces, detention ponds can be constructed along the borders of the fields and perpendicular to the main slope. These ponds can store run-off and sediments. This system is common in many smallholdings in areas of Central and WE, and has been observed to be effective in reducing soil and nutrient loss through erosion [172].

**Subsoiling** is a cultivation practice generally used to loosen compacted or hardpan subsoil horizons with the aim of improving soil infiltration and/or root penetration [118]. It is widely used in agricultural soils that are prone to compaction in Central and N Europe [173] and also before planting vineyards in many wine-producing areas in WE (e.g., [174]). Its final effect depends on multiple factors, especially soil texture, the type of tools used in the operation and the crops planted.

The design and implementation of adequate **crop rotations**, and where possible, the introduction of **cover crops** have generally been observed to have positive effects on soils across Europe, especially in relation to SOM and biological soil properties [119]. However, trade-offs in terms of decreased crop yields and/or increased greenhouse gas (GHG) emissions must be considered for a complete assessment of the consequences of their implementation.

### *2.4. Sustainable Forest Management and Afforestation*

As outlined above, there has been a significant increase in forested areas in WE in recent years. In the period 1990–2005, the gain in forest surfaces in Europe was around 10,939,000 ha [97]. The greatest increases in this period were observed in Ireland (+52%, with 10% of the total country surface under forest), Spain (+33%), Portugal (+22%), Italy (+19%), Greece (+14%), Denmark (+12%), and the UK (+9%). Afforestation has been strongly encouraged by EU funds and regulations, such as the Council Regulation 2080/92, and afforestation of agricultural land and the development of forestry activities on farms have been promoted [175].

In planted forests, the Ministerial Conference on the Protection of Forests in Europe (MCPFE) is a pan-European policy process for the sustainable management of the continent's forests. It develops common strategies for its 46 member countries, which include all of the EU Member States, other European countries and Russia. Cooperation between countries, which began in 1990, has produced guidelines, indicators and criteria for sustainable forest management (MCPFE, 2007). In the resolution of the Helsinki Conference in 1993, the signatories explicitly stated that "Human actions must be avoided which lead, directly or indirectly, to irreversible degradation of forest soils and sites, the flora and fauna they support and the services they provide" (MCPFE, 1993). In the subsequent Ministerial Conference held in Lisbon in 1998, the participants adopted six criteria for sustainable forest management from the Pan-European Criteria and Indicators for Sustainable Forest Management, and endorsed the associated indicators as a basis for international reporting and for development of national indicators (MCPFE, 1998). Among these criteria, soil conservation is mentioned several times. In particular, SOC stocks are included as indicators for the maintenance of appropriate enhancement of forest resources and their contribution to global C cycles, soil condition is used an indicator for the maintenance of forest ecosystems health and vitality, and soil functions are used as indicators for the maintenance and appropriate enhancement of the protective functions in forest management.

Some countries in WE have designed good forest practice guidance for soil protection. In the UK, the government's approach to sustainable forestry involves specific good forestry practice requirements and guidelines for soils. In Northern Spain, specific regulations are implemented to avoid damage to soil in forestry operations [176].

In addition to national and regional guidelines, many planted forests in WE are established under different certification frameworks that in most cases include requirements in relation to soil management. Forest certification is a voluntary process conducted by an independent third party who issues a written statement or certificate guaranteeing that forest management is carried out according to standards considering ecological, economic and social aspects [177]. The two principal objectives of certification are to improve forest management and to ensure market access for products from certified forests, allowing both consumers and companies who sell forest products to play an important role in forest conservation [178]. The proportion of certified forest area decreases significantly from North to South in WE: in Ireland and the UK more than 50% of the forest surface is certified, in Portugal and Spain it is only around 6% and in France around 32% of the forest surface is certified [178].

## 3. Soil Quality and Future Trends in Western Europe

Unlike for air and water, environmental issues associated with soil degradation have been given marginal consideration in environmental regulations in WE. Soil protection has been addressed indirectly through measures aimed at the protection of air and water or developed within sectoral policies [13]. The most important initiative that partly redresses the lack of explicit soil protection is, undoubtedly, the proposal for the development of a Thematic Strategy for Soil Protection. Officially launched in 2002 (COM (2002) 179), this has led to a significant research effort and yielded an impressive amount of information on soil degradation in the EU. The final aim of these efforts was the implementation of a EU Directive for soil protection within the EU (*i.e.*, in most WE countries). Unfortunately, after several years of discussion between different European institutions, the proposal for a Soil Framework Directive similar to those existing for Air and Water was finally withdrawn from the European Commission agenda in May 2014 [179].

Although great efforts are being undertaken to recover this EU initiative, at present soil protection in WE mainly relies on national-level policies and indirect policies such as the Nitrates Directive and the Water Framework Directive [118], and the agri-environmental measures included in the CAP regulations, as explained below. As a result, many soil degradation issues are not completely covered by legislation at present. The only field in which soil protection is directly addressed in national laws is soil contamination and the management of contaminated sites. Between 1980 and 2006, most WE countries developed specific laws to address issues related to soil contamination and contaminated sites (see [180]). For instance, in Germany, the Soil Protection Act acknowledges the ecosystem functions of soils and states that they should be preserved over time. However, the Act focuses on and limits the threats to these functions derived from chemical contamination [181,182]. In the Netherlands, the Soil Protection Act states the importance of prevention, reduction and reversal of changes in the soil quality that imply a reduction or threat to the functional properties of soil has for humans, plants and animals [181]. The German Act mainly focuses on degradation risks associated with contamination of soil by toxic compounds.

### *3.1. Soil Quality and Ecosystem Services of Soils in Western Europe*

The formal concept of soil quality (SQ) was developed in the second half of the last century [14], in response to the need to assess soil degradation problems from a holistic perspective [31,183,184]. Assessment of SQ is complicated by the fact that soil is a heterogeneous resource for which it is difficult to establish quality standards. Thus, SQ has not been defined by established universal criteria, but as the capacity of a given soil to function [185]. Proper soil functioning is understood as the capacity of a soil to accomplish its natural (ecosystemic), social and economic functions in a sustained way over time [186]. Defining soil functions was one of the goals of the European Thematic Strategy for Soil Protection. Five critical soil functions have been identified: production of food and other biomass; storage, filtration and transformation of minerals, water and other elements including C; supply of habitat and gene pool for a variety of organisms; acting as the physical and cultural environment for mankind (present and past); and as a source of raw materials. As the Commission's Communication (COM (2002) 179) states, most of these functions are inter-dependent and the development of some of them (raw materials, physical environment for mankind) may imply a reduction in the ability of soils to accomplish the others.

Since this Communication was launched, some efforts have been made to develop SQ monitoring systems, mainly within the EU. At the continental level, the basis for SQ and sustainability evaluation was established via definition of a common framework to assess soil functions, degradation threats and soil-use options [187]. This framework proposed a three-step evaluation in which the capacity of a given soil to accomplish a selected function is first evaluated. The existing threats for the considered soil and soil function are then determined, and finally, the capacity of the soil to accomplish the function is evaluated for different levels of pressure from the threats identified in the second step. This approach acknowledges that the results of the three steps, and especially the sensitivity of a soil to different threats, is soil- and site-dependent. This implies that the soil functional ability (number of functions that a soil can accomplish) and the soil responses to different levels of human-induced or natural threats (soil response capacity) must be evaluated to define SQ for a given soil [188]. The development of this framework requires detailed information on soil types, soil characteristics and threats to soil in each area studied. Its full development in detail therefore seems complicated. A first step is the identification of risk areas based on clearly described criteria (such as in [189]) for the identified threats to soil. Strategies for evaluating the risk of SOM decline, soil erosion, soil compaction, salinization and landslides in WE have been suggested [189]. For each of these, the authors provided the information needed to evaluate the risk of soil degradation based upon soil/topography/climate parameters in each site. For most sites, it was concluded that determining quantitative scores or thresholds requires more accurate information than is currently available.

The ENVASSO project is another important pan-European attempt to advance towards the identification of SQ indicators (SQI) and baseline values. The main aim of this project was the creation of a comprehensive, harmonized soil information system in Europe via the design and testing of an integrated and operational set of indicators [70,74]. Its output ([15,41]) includes selected indicators, threshold and baseline levels for the major soil threats identified in the European Thematic Strategy for Soil Protection (COM(2002) 179 final) and its subsequent evaluations (e.g., COM(2012)46). For each soil threat, three parameters were selected from an initial base of 290 indicators [15]. Some of the selected soil parameters are actually measured values, and others are estimated through modeling. The indicators were selected by experts, following these criteria: relevance for assessing each soil threat, ease of application, link to policy aims and applicability in a pan-European context. Baseline and threshold values were established for some of these indicators. However, it is recognized that such values may have to be established separately for different areas in Europe because of the variety of soil types and the variability in environmental conditions and land use. Table 3 summarizes the soil threats and properties suggested as indicators by the ENVASSO Project team, and which of those were finally selected as the best indicators for each threat.

The performance of those indicators was tested in different pilot areas in Europe, and the results of these tests have been reported in detail [190,191]. Complete descriptions of the protocols that should be used in each case have been published [84]. The purpose of drawing up this list of indicators was to establish a monitoring network in which changes in soil characteristics can be periodically controlled [192].

Development of the European Soil Data Centre provides additional mechanisms for reporting information on soil and SQ data and adequate definitions of SQ, SQI and monitoring networks [85]. The spatial density of soil monitoring networks is very non-homogeneous in WE, with no or very few systematic sampling sites available for many of the indicators shown in Table 3 [192]. In fact, some of those SQI (e.g., those related to soil erosion or soil organic C) have been monitored with much higher intensity and frequency than others such as soil biodiversity [70]. The LUCAS (Land Use/Land Cover Area Frame Survey) represents the first effort to build a consistent spatial database of the topsoil (0–30 cm) cover across Europe, based on standard sampling and analytical procedures [193]. The aim of LUCAS is to gather harmonized information on land use/land cover and several soil properties, such as soil texture, organic carbon, nitrogen content, pH and cation exchange capacity. The survey also provides territorial information for the analysis of the interactions between agriculture, environment and countryside, such as irrigation and land management. LUCAS field surveys have been carried out every three years since 2006. The next LUCAS field survey is planned for 2015.

**Table 3.** Proposed and selected soil properties for monitoring soil quality in the Environmental Assessment of Soil for Monitoring (ENVASSO) Project.

| Soil Threat | Soil Indicator * | Source $ | Baseline | Threshold |
|---|---|---|---|---|
| **Soil contamination** Diffuse and local | **Heavy metal content** | M | National background levels | National legislation |
| | Nutrient balance | M | Average national balance | Defined at a regional level |
| | Organic pollutant concentration | M | National background levels | National legislation |
| | Topsoil pH | M | Not defined | Not defined |
| **Salinization** | **Salt profile** | M | EC saturation extract < 2 dS/m | 0.10% salt content or $EC_e$ < 4 dS/m |
| | **Exchangeable sodium percentage (ESP)** | M | <5% | >15% |
| **Soil compaction** | **Density (bulk, packing and total density)** | M/E | Measured in non-compacted soils | Packing density: 1.75 $g/cm^3$ |
| | **Air-filled pore volume at a specified suction** | M | Measured in non-compacted soils | Air-filled pore vol. at 5 KPa >10% |
| | Permeability | M | Not defined | Not defined |
| | Mechanical resistance | M | Dependent on soil structure status | Penetration resistance < 2–5 MPa |
| | Structure status | E | Not defined | Not defined |
| | **Vulnerability to compaction** | E | Not defined | Persistent (not recoverable) |
| | Drainage | E/M | Not defined | Not defined |
| | Precompression strength | E | Measured in non-compacted soils | <90–120 KPa |
| **Decline in SOM** | **Topsoil organic carbon content** § | M | Not defined | Not defined |
| | **Soil organic carbon stock** | M | Not defined | Not defined |
| | **Peat stocks** | E | Not defined | Not defined |
| | Topsoil C:N ratio | M | Not defined | Not defined |
| **Soil biodiversity** | Microbial and fungal diversity | M | Not defined | Not defined |
| | **Earthworm diversity and fresh biomass** | M | Not defined | Not defined |
| | Macrofauna diversity | M | Not defined | Not defined |
| | **Collembola/Enchytraeid diversity** | M | Not defined | Not defined |
| | Acari diversity | M | Not defined | Not defined |
| | Nematode diversity | M | Not defined | Not defined |
| | **Microbial respiration** | M | Not defined | Not defined |
| | Microbial activity (enzymes) | M | Not defined | Not defined |

**Table 3.** *Cont.*

| Soil Threat | Soil Indicator * | Source $ | Baseline | Threshold |
|---|---|---|---|---|
| **Erosion** | | | | |
| Water, wind and tillage erosion | **Estimated soil loss** | E | Water and tillage erosion:<br>N Europe 0–3 ton/ha*year<br>S Europe 0–5 ton/ha*year<br>Wind: N&S Europe: 0–2 ton/ha*year | Water and tillage erosion:<br>N Europe 1–2 ton/ha*year<br>S Europe 1–2 ton/ha*year<br>Wind: N&S Europe: 2 ton/ha*year |
| | **Measured or observed soil loss** | M | Water: 0.5 ton/ha*year | Water: 1–2 ton/ha*year |
| **Soil sealing** | No soil properties as indicators | | | |
| **Landslides** | No soil properties as indicators | | | |

* Indicators shown in bold type are within the three selected for each threat; $ M: measured; E: estimated or calculated; § For desertification: SOM in desertified land, salt content in desertified land and soil biodiversity in desertified land; Sources: [15,41,192].

Finally, within agricultural soils, the above-mentioned Communication of the European Commission on the development of agri-environmental indicators for monitoring the integration of environmental concerns into the CAP (COM(2006) 508 final) also established SQ as a state/impact indicator set [194] (see Section 3.2). This indicator set, which has not yet been completely assessed, aims to describe the following: (i) the soil capacity for biomass production; (ii) the input required for optimal productivity; (iii) the soil response to climatic variability; and (iv) carbon storage, filtering and buffering capacity. These four issues are to be integrated in a SQ index aimed at quantifying the ability of soils to provide agri-environmental services by performing their functions and responding to external influences. The index is determined following a similar previously described approach [187]. The SQ index is calculated from four sub-indicators of similar weight, which are relevant either to the agricultural and/or to environmental performance of soil: (i) soil productivity index; (ii) soil fertilizer response rate; (iii) production stability; and (iv) soil environmental services. So far, only indicators (i) and (ii) have been calculated for most regions and countries within the EU. The productivity index (i) takes into account both the inherent soil properties, and the climate and topography of each territory. When climate is more limiting for rainfed agricultural production, soil properties supporting productivity gain more weight in the index than in areas with fewer or no climatic limitations. The index therefore does not only represent soil characteristics and should be considered as a land quality index rather than a SQ indicator. In WE, the most productive croplands are found in NW France, Belgium and the Netherlands, together with W England and Scotland. The most widespread areas of land of low productivity are in Spain and SE Italy.

The response to fertilization (ii) was calculated by assigning a fertilizer response score for each soil unit based on soil properties. The areas with a high response value matched those with high productivities in (i). Conversely, the areas with soils displaying a low response to fertilization in rainfed croplands were found in Spain, especially in the Ebro and Guadalquivir river basins. The stability of crop production will be estimated from soil characteristics that explain higher variability under limiting water and climate conditions. Finally, four soil functions will be considered for evaluating the environmental services of agricultural soils: organic C storage, the filtering capacity, the transforming capacity of the soils, and their biodiversity and biological activity. Further development of this index will be of use in land-use planning and environmental protection.

At a national level, some attempts have been made to establish standard systems for the periodic control of SQ and SQI, such as the National Soils Indicator Consortium in the UK (UKSIC) and the Soil Quality Monitoring Network (Réseau de Mesures de la Qualité des Sols, RMQS) in France. In the UK, the UKSIC has established a minimum set of SQI for broad-scale soil monitoring. This includes soil organic carbon, soil pH, heavy metals (Cu, Zn, Ni), Olsen P, potentially mineralizable N and bulk density [195]. These indicators were selected by experts and constitute a basis for periodical comparisons in a network of sampling sites across England and Wales. Scotland is at present developing its own soil monitoring system within the Scottish Soil Framework (SSF) [196].

In France, the development of the RMQS was designed as a periodic (every 10 years) collection of soil samples in a regular template (16x16 km). As in the UKSIC, soil properties (texture, organic C, nutrient contents and some other physical and chemical properties) are measured at each sampling point where soil and soil use are characterized [29,197].

Similar soil monitoring networks exist in other WE countries such as Germany and Austria. Many of these networks aim to monitor soil degradation at a national level by making regular and comprehensive comparisons of the selected indicators over time. They are not intended for field-scale application to detect main soil constraints and thus to derive soil management and conservation recommendations for particular sites [198].

One problem associated with studying SQ in such a way is the difficulty in evaluating the absolute values and observed changes in SQI. This problem arises from two sources. First, the heterogeneity of soils and soil uses makes it difficult to establish baselines and thresholds (see gaps in Table 3, and the case for SOM in [62]). Second, the same factor may have a different score depending on the soil function or ecosystem service being evaluated.

In this sense, to our knowledge, no systematic and normalized strategy for SQ evaluation at the national level equivalent to the Soil Management Assessment Framework (SMAF) in the USA [183,199,200] exists in WE. This type of evaluation focuses on the selection of a minimum data set of SQI that must include soil physical, chemical and biological attributes of soil. Different scores can be assigned to these indicators on the basis of their average values and the relationship between these values and the performance of each soil function. The scoring therefore depends on the soil type and the soil function(s) considered [183]. In contrast to SQ evaluations focused only on monitoring soil properties in time, this type of system defines different scores and quality attributes depending on the soil function and/or soil ecosystem service considered. Although these studies are complicated to carry out because they are site specific, they are very valuable for assessing SQ response to different types of agricultural management (e.g., [200]). Some initiatives are currently being developed for considering the multi-functionality of soils for land management decision (see Section 3.3 for those by Schulte *et al.* and by Volchko *et al.* [201,202]).

At regional and local levels, many studies have addressed SQ monitoring systems with a holistic approach in WE. Most of these focus on the evaluation of SQI for SQ monitoring under particular soil uses and/or under particular conditions, such as soils under OF ([203,204], forest soils under different types of management [49,205], extensive rainfed cereal crops in semi-arid land [206], Mediterranean mountain agrosystems and vineyards [207,208], and many others. Since biological SQI are generally not considered in large soil inventories or monitoring networks [209], many of these studies identify suitable biological SQI such as microbial parameters, soil fauna, earthworms and other macro invertebrates, *etc.* (e.g., [76,208,210]). The results of studies using holistic SQ evaluation systems in agricultural soils are diverse, as are the agrosystems studied. In general, different management systems are compared. To cite two examples, enhanced soil quality was observed under NT in extensive rainfed cereal systems in semi-arid Spain, but the impact of organic farming in vineyards in S France was not detectable in the overall SQ [154,206].

In summary, the development of SQ monitoring programs and SQ evaluation systems that enable accurate assessment of the soil ability to accomplish functions or ecosystem services is an ongoing and promising strategy for soil protection in WE. Two considerations are important in this framework: inclusion of the farmers' perspective and evaluation of economic trade-offs in the different evaluations [211].

### *3.2. Soil Protection and the EU's New Common Agricultural Policy (CAP)*

Although there is a shift towards including the multi-functionality of soils into the legislation in many WE countries (e.g., The Netherlands [212], Belgium [213] and France [214]), a specific legislative framework for unpolluted agricultural soils is so far lacking. However, as most countries in WE belong to the EU, they are affected by the CAP. The CAP is based on two groups of measures or pillars. Pillar one corresponds to the legislative framework in relation to agricultural production subsidies. Pillar two includes the support policies for rural development in the EU.

Since 1999, in the so-called Cardiff process, environmental protection measures have been integrated into the CAP. This implies that the successive reforms of the CAP established a list of statutory management requirements and a reference level of good agricultural practices that should be respected by European farmers being supported by the CAP. Different requirements and reference levels have been established for different local conditions by member states or competent regional or local authorities. The cross-compliance character of these measures implies that they are mandatory for farmers receiving CAP subsidies. From the perspective of soil conservation, cross-compliance links direct payments with compliance by farmers with the obligation of keeping land in *good agricultural and environmental condition*, including standards related to soil protection (namely protecting soil from erosion and the maintenance of soil organic matter and soil structure) (EU Council Regulation 73/2009). Table 4 shows the different measures adopted in this framework for different WE countries in 2006, as compiled by GEIE Alliance Environment [215].

The CAP has also encouraged sustainable soil management by funding the provision of environmental public goods and services beyond mandatory requirements to those farmers adopting the so-called *agri-environmental measures* (AEMs). In many cases, this implies adopting agricultural activities or levels of production intensity that deliver positive environmental outcomes, while not necessarily being the first choice from the point of view of profitability. Some of these measures are related to management systems that can promote SQ. As a result, throughout its successive reforms, soil protection measures have been reinforced in the CAP and expanded to encourage organic and integrated farming, extensification, maintenance of terraces, safer pesticide use, use of certified composts, and afforestation, among others [13]. The flexibility of AEMs allowed WE countries in the EU to develop different measures or schemes to reflect different bio-physical, climatic, environmental and agronomic conditions and therefore to tailor management options to suit the characteristics of their agricultural sector. As described in a case study in Brandenburg (Germany) [182], AEMs and cross-compliance measures associated with the CAP are often the only significant official policies addressing soil conservation in agricultural land in WE [53].

**Table 4.** Soil-related measures adopted for maintaining arable land in good agricultural and environmental condition within the Common Agricultural Policy (CAP) framework in Western Europe (WE) countries within the EU.

| | Country | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Measure** | **DE** | **AT** | **BE *** | **DK** | **ES** | **GR** | **FR** | **IE** | **IT** | **LU** | **NL** | **PT** | **UK** | **SE** |
| **Soil erosion control** | | | | | | | | | | | | | | |
| Minimum soil cover | x | x | x | x | x | x | x | x | | x | x | x | x | x |
| Minimum land management | x | | x | | x | x | | x | x | x | x | x | x | |
| Terrace conservation | x | x | | x | x | x | | | x | x | | | | |
| Other measures for erosion | | | | | | x | x | | | | x | | x | |
| **Organic matter (SOM) management** | | | | | | | | | | | | | | |
| Crop rotation | x | | | | | x | | x | | x | | | x | |
| Management of crop residues | x | x | x | | x | x | x | | x | | | x | x | |
| Other measures for SOM | | | x | | | | x | | | x | x | | | x |
| **Soil structure protection** | | | | | | | | | | | | | | |
| Use of adequate farm machinery | | x | | | x | x | | x | | | | | x | |
| Other measures for structure | | | x | | | | x | | | x | x | | | x |
| **Other measures** | | | | | | | | | | | | | | |
| Livestock density control | x | | x | x | x | x | x | x | x | x | x | x | x | x |
| Grassland protection | x | | x | x | x | x | x | x | x | x | x | | x | x |
| Slope assessment | x | x | | | x | x | | x | x | | | | x | x |
| Wild vegetation control | | x | x | x | x | x | x | x | | x | | x | x | x |
| Olive-grove preservation | | | | | x | | | | x | | | | | |
| Other | | x | x | x | x | | x | | | | | x | x | |

* Data for Belgium include Flanders and Walonia; Source: Adapted from [215] (Data for 2006).

Within this framework, the development of agri-environmental indicators for monitoring the integration of environmental issues in the CAP was introduced in 2006 (COM 2006-508 final). As explained above, some of these indicators involve soil protection. These have been selected for monitoring farm management practices, agricultural production systems, pressures and risks to the environment and the state of natural resources. Their level of development differs greatly: while some are already operational, others are only defined and lack data. Table 5 shows these indicators and their development to date.

The changes in CAP towards more environmentally-oriented policies had different results in relation to SQ [175]. In most cases, measures included in AEMs, such as contour and reduced tillage, led to reduced erosion rates, higher biodiversity and generally improved SQ in arable land and grasslands across WE. However, promotion of set-aside, for example, may have the opposite effect in arid and semi-arid land. The difficulty in fulfilling the requirements of cross-compliance also stimulated land abandonment in some areas. The CAP has also encouraged the use of soil cover systems and crop rotations, and has contributed to the dissemination of CF [121].

**Table 5.** Agri-environmental indicators for CAP monitoring in relation to soil protection in Western Europe. Indicators shown in bold type directly address soil conservation issues.

| Domain | Indicator | Field of control | Related soil protection issue |
|---|---|---|---|
| **Response** | | | |
| Public policies | Agri-environmental commitments (AEC) | Surface under AEC | Soil degradation due to agricultural management (depending on each AEC in each WE country and region) |
| | Agricultural areas under Natura 2000 | (To be developed) | |
| Technology and skills | Farmers' training level and environmental advicing | Farmers in training programs | --- |
| Market signals and attitudes | Area under organic farming | (To be developed) | |
| **Driving force** | | | |
| Input use | Mineral fertilizer consumption | Mineral and organic N and P | Diffuse contamination |
| | Consumption of pesticides | Plant-protection products | Diffuse contamination, biodiversity decay |
| | | | Erosion, salinization |
| | Irrigation | Irrigable and irrigated surface | |
| | | Irrigation methods | |
| | | Source of irrigation water | --- |
| | Energy use | Use of energy by fuel type | |
| Land use | **Land use change** | Agricultural land take | Soil sealing |
| | Cropping patterns | Share of arable land, permanent grassland and permanent crops | Erosion, organic matter decay, compaction |
| | Livestock patterns | Share and density of major livestock types | Compaction |

**Table 5.** *Cont.*

| Domain | Indicator | Field of control | Related soil protection issue |
|---|---|---|---|
| Farm management | **Soil cover** | Time when soil is under crops | Erosion, organic matter decay, compaction |
| | **Tillage practices** | Share of arable land in CT, CoT and ZT * | Erosion, organic matter decay, compaction, biodiversity |
| | Manure storage | (To be developed) | |
| | Intensification/extensification | Inputs per factor production | --- |
| Trends | Specialization | Number of products in farm | --- |
| | Risk of land abandonment | Cessation of agricultural activities | Erosion, organic matter, biodiversity |
| **Pressures and risks** | | | |
| Pollution | **Gross nitrogen balance** | Surplus N risk | Diffuse contamination |
| | **Risk of pollution by P** | Surplus of P risk | Diffuse contamination |
| | **Pesticide risk** | Pesticide consumption | Contamination, biodiversity |
| | Ammonia emissions | Ammonia emitted by agriculture | --- |
| | Greenhouse gas emissions | GHG emitted by agriculture | Soil organic matter decay |
| Resource depletion | Water abstraction | For agricultural use | --- |
| | **Soil erosion** | Estimated water erosion | Soil erosion |
| | Genetic diversity | (To be developed) | |
| Benefit | High Nature Value farmland | (To be developed) | |
| | Renewable energy production | From agriculture and forestry | --- |
| **Impact/state** | | | |
| Biodiversity/habitats | Population trends in farmland birds | Monitoring 37 birds species | |
| Natural resources | **Soil quality** | Productivity index, fertilizer response rate, production stability and soil environmental services. | Organic matter, biodiversity, physical degradation, chemical fertility, biodiversity |
| | Water quality—Nitrate pollution | Nitrates in rivers and groundwater | |
| | Water quality—Pesticide pollution | (To be developed) | |
| Landscape | Landscape—state and diversity | Dominance and diversity | As in land use and soil cover. |

* CT: Conventional Tillage; CoT: Conservation Tillage; ZT: zero tillage; Source: [216].

The latest reform of the CAP (for the period 2014–2020) includes significant changes in relation to environmental protection: a new policy instrument of the first pillar (*greening*) is directed to the provision of environmental public goods [217]. This instrument has been designed to reward farmers for respecting three obligatory agricultural practices: (i) maintenance of permanent grassland; (ii) maintenance of ecological focus areas (land left fallow, terraces, landscape features, buffer strips and afforested areas); and (iii) crop diversification (which includes having at least three crops on the same agricultural exploitation, or including agronomic practices with minimum soil disturbance and green coverage of the soil surface in permanent crops). Implementation of these measures across the EU is expected to increase soil protection, as many of the measure directly involve soil. For instance, in Spain, ecological focus areas include set-asides, N-fixing crops, afforested surfaces and land devoted to agroforestry. The aim of this reform is also to extend and reinforce the environmental component to Pillar 2, by including agri–environmental-climate measures, OF, forestry measures and investments that are beneficial for the environment or climate (amongst others) in rural development policies.

Nevertheless, the final net effect of the new CAP on SQ in WE will depend on multiple factors, both at local and national level, and it is possible that trade-offs between conflicting agricultural sector policies will appear. For example, a policy aimed at mitigating soil erosion (achievable through CF) may conflict with another policy discouraging the use of herbicides (often critical to the initial success of CF practices [121]). Similarly, CAP measures designed to promote increased agricultural production may diverge from those developing environmental policy objectives [201].

In addition to environmental issues affecting terrestrial and aquatic systems, CAP reforms included since 2010 support climate action. The reduction of GHG emissions from farmland, when including soil management strategies and the stabilization of organic C in soils, may affect SQ in WE. The efficiency of these strategies may differ both in terms of the abatement of GHG emissions and economic costs, as shown for ten possible measures in French farms [218]. Among these measures, those directly affecting soil and soil management had positive (cover crops, hedges), very little (legume crops, agroforestry, reduced tillage) and negative (organic fertilizer application) effects in terms of net $CO_2$ abatement. Conversely, some agricultural practices that improve SQ have been observed to increase GHG emissions [119].

The new CAP structure offers the possibility of including climate action instruments in both Pillar 1 and Pillar 2; however, in some cases the impact of such measures is still uncertain. Nevertheless, according to [219], the new CAP will probably be one of the most important opportunities for the EU-28 to tackle the climate change issue. Implementation of CF and use of cover crops are included among the proposed measures to be adopted at farm level.

### *3.3. Promising Strategies for Soil Quality in Western Europe*

From the above it can be concluded that SQ monitoring, assessment and protection are currently at different levels of development in WE. Some promising strategies for increasing the awareness of SQ and improving its consideration in future policies in WE include (i) the development of soil status and SQ monitoring networks at a continental scale; (ii) the inclusion of SQ and soil functionality issues in environmental and agricultural legislation; (iii) the research for accurate and, if possible, simple SQ monitoring tools; and (iv) the implementation of multi-actor and multi-target strategies for promoting and increasing SQ awareness and the effective implementation of SQ-improving management practices.

**Monitoring SQ** (i) is essential to measure soil degradation and to develop appropriate strategies for soil protection. This includes the creation of international networks to address critical and crosscutting soil issues. In addition to national initiatives, Europe has several projects that address these issues. For example, the above-mentioned LUCAS survey and the recently launched (2013) European Soil Partnership (ESP) are added to previous initiatives under the support of the Joint Research Center of the EU, such as the European Soil Bureau and the European Soil Data Center, from which information on soils in the EU can be retrieved at the European Soil Portal (http://eusoils.jrc.ec.europa.eu/). All these initiatives are supported by the JRC of the EU. The ESP is one of the regional partnerships of the Global Soil Partnership. The objective of this regional network is to bring together the various scattered networks and soil-related activities within a common framework, open to all institutions and stakeholders willing to actively contribute to sustainable soil management in Europe. The ESP has five main pillars of action, which include promoting sustainable management for soil protection, encouraging investment, technical cooperation, policy, educational awareness and extension in soil, promoting soil research related to productive, environmental and social development actions, enhancing the quantity and quality of soil data and information, and harmonizing methods, measurements and indicators for the sustainable management and protection of soil resources. These initiatives must account for the fact that sampling schemes suitable for inventory are not necessarily also suitable for monitoring [220]. Monitoring information on farm management practices, on how these practices affect the environment, and whether they correspond to recommended (or legislated) practices and standards may also contribute to early detection and assessment of SQ issues [221].

In relation to the **inclusion of SQ issues in legislative and assessment tools** (ii), the functional land management strategy recently proposed by Schulte and coworkers is a complete and promising model for developing policies that enable achievement of goal targets for productivity by considering and enhancing the capacity of soils to provide ecosystem services [201]. This strategy is based on optimizing five basic soil functions (biomass production, water purification, C sequestration, habitat for biodiversity and recycling of nutrients) by studying the potential of soils to supply these, as well the present and future demands by taking into account growth goals and environmental restrictions. Although the study was proposed for Irish agricultural soils, it could be expanded to other European regions (including the EU).

Another example of a decision tool that considers soil functions and that can be used to evaluate remediation alternatives for contaminated soils has been described by Volchko and coworkers [202]. This is based on the inclusion of selected ecological soil functions (basis for primary production, cycling of carbon, water, nitrogen and phosphorus) in a multi-criteria decision analysis. The degree to which these functions are fulfilled in remediated sites is determined using a minimum data set of SQI (soil texture, coarse material, organic matter, available water, pH, potentially mineralizable nitrogen, and available phosphorus), which are scored and integrated in a SQ Index, in an approach very similar to that described in the SMAF [199].

A good example of the incorporation of soil functionality criteria in legislative frameworks is the ongoing process in the above-mentioned SSF in Scotland. This framework aspires to develop the EU

Soil Thematic Strategy for Scottish soils, providing a legislative framework for soil protection that accounts for the inherent soil quality and the multi-functional roles. The declared aim of this SSF is to promote the sustainable management and protection of soils consistent with the economic, social and environmental needs of Scotland. The framework identifies more than 35 actions in different fields (research, soil conservation, land management, *etc*.) linked to expected soil outcomes. Each action has a delivery date and designates the persons or bodies responsible for its accomplishment. These actions include the development of a Scottish soil-monitoring network and review of the land capacity for agriculture. The monitoring network focuses on the functions of soils related to ecosystem services. This strategy will be included in the more general strategy for land use [222], which includes for instance, the rationale for woodland expansion. In this rationale, a soil-based evaluation of land is made in order to protect sensitive soils (such as peatland soils) or high quality agricultural land, which provide essential services such as C sequestration and food production, from being converted into forest plantations. Similar systems in which the soil types (and therefore their inherent quality) are considered for management decisions have been developed in England for forestry management. These guides (e.g., Whole-tree harvesting guide [223]) determine the type of practice to be implemented as a function of soil characteristics in each forest plantation.

In this sense, the positive and negative market services provided by forests (including those related to soil protection) play a significant part in decisions about forests, but they are notoriously difficult to quantify, and people seldom agree on their value. The EU needs a policy framework that coordinates and ensures coherence of forest-related policies and allows cooperation with other sectors that influence sustainable forest management. For example, the efficient use of wood biomass as a renewable energy resource and increased utilization of domestic, renewable resources of biomass has been identified as an opportunity for many European countries to increase their energy security. Significant impacts on roundwood and wood residue markets are expected as the energy sector becomes a major consumer of wood biomass. There has already been a rapid increase in the production of energy from harvested forest residues (small diameter tree stems, branchwood and foliage) in some Nordic countries [224,225].

Another challenging aspect in many areas in the future is the **development of adequate SQI** (iii) [16] and in particular establishment of the relationship between their levels and soil functions for different areas and land uses across WE, as indicated for levels of soil organic matter [62]. In this sense, although much work has been done in some aspects (for instance in relation to climate change mitigation and adaptation strategies), research for developing SQ assessment tools and SQI in other aspects is still pending in WE. For example, new methodological approaches for soil biodiversity measurement are being developed [73], as well as new tools to assess land susceptibility to wind erosion [55]. New techniques such as near-infrared reflectance spectroscopy are being considered for the evaluation of SQ and soil properties [226,227].

Finally, attempts to **broaden the participating agents (multi-actor) and the objectives (multi-target) of new soil management strategies** that enable improved SQ (iv) also exist in WE. For example, the LIFE project series devoted to soil degradation problems and soil protection aims to translate science and policy into practice [228]. These measures enable the involvement of stakeholders in the launching and demonstration of new techniques and systems for sustainable soil use in the EU. There is a special need for developing cost/benefit analysis, such as that recently developed for GHG abatement in French agriculture [218].

At a different level, certification of agricultural and forest goods has been a successful strategy in some cases. In addition to public policies or official certifications, some initiatives such as integrated farming and GlobalGAP are provided to producers who apply some soil-friendly management practices, with certified labels awarded for their products. This type of labeling increases the awareness of consumers and can indirectly promote the adoption of production strategies that preserve or improve SQ.

## 4. Concluding Remarks and Future Threats to Soils in W Europe

The objective of this review paper was to consider soil degradation problems, the policies and strategies for soil protection and the future perspectives for SQ assessment in WE, with the aim of providing a summary of SQ problems and evaluation in the region.

The review of soil degradation showed that different population trends, economic activities, local legislative conditions and historical land use have created different types of SQ-related problems. Problems related to soil chemical, physical and biological degradation have been identified in different areas of WE. Soil losses through erosion are also significant in many regions. Many of these problems are related to agricultural and forest management, which are the predominated activities on non-urban land.

The strategies implemented in WE to overcome these problems in cultivated and forest land include conservation agriculture, organic farming and other sustainable agriculture and forest management systems. These have had results of varying success, mainly because of the site-specificity of their effectiveness. This highlights the fact that universal solutions are difficult to design and achieve for a complex problem such as soil degradation.

Moving towards strategies that consider soil functions within the framework of SQ assessment and regulation in WE would help to optimize the assessment of soil degradation problems and the search for effective strategies for sustainable soil management adapted to the characteristics of European regions. However, although promising examples exist at a local level, and a significant amount of work has been done in pan-European projects, more information than is currently available seems necessary for a national or continental-scale SQ assessment. Some platforms such as the European Soil Database, the LUCAS framework and development of the European Soil Portal will undoubtedly contribute to the harmonization of soil data and soil monitoring across Europe, for instance in the task of identifying adequate SQI that should be calibrated across the continent. Developing cost/benefit analysis and multi-actor approaches to address SQ and soil protection strategies in WE seems a promising approach.

However, so far only some problems related to soil degradation have been considered in the legislation (*i.e.*, contamination), and specific soil legislation is lacking in WE. As a result, most of the attempts involving soil protection and SQ enhancement are found in normative frameworks that affect soils only indirectly, mostly in the form of environmental-oriented restrictions to the CAP subsidies both to agricultural production and in rural development plans. Thus, the sustainable use and conservation of WE soils are not yet fully guaranteed. This is especially true for forest soils, which are not directly affected by most CAP regulations and mainly depend on certification strategies as indirect mechanisms of ensuring sustainable management.

The need for specific legislation and holistic SQ assessment strategies becomes clear in light of evidence of future threats to soils and SQ in the region. In 2012, the European Commission identified increased future soil degradation problems for European soils if several aspects are not properly addressed [229]. The greatest challenges cited were land use issues (including the increasing demand for productive soils and land take for urbanization and infrastructure areas), the preservation of SOM (especially in peatland, pastures and forest soils) and more effective use of fertilizers and organic waste that may help to optimize soil fertility without leading to SQ degradation. These three aspects are closely connected to most of the soil physical, chemical and biological degradation processes and soil loss problems described in the Soil Thematic Strategy. A review of the topics addressed suggests that, as recently noted [201], greater scientific knowledge and management of soils will be critical in meeting the challenges of food security and environmental sustainability in the forthcoming years in WE and worldwide.

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**Author Contributions**

Iñigo Virto is the corresponding author of this work. He coordinated the other authors and was responsible for the final writing and edition, and for Sections 1.3, 2.3, 3.1–3.3. María José Imaz was responsible for Section 2.2 and all aspects related to organic farming and agroecology in the other sections. Oihane Fernández-Ugalde was responsible for Sections 1.1 and 1.2 and collaborated in the other sections. Nahia Gartzia-Bengoetxea was responsible for Section 2.4 and all aspects related to forest management and forestry. Alberto Enrique collaborated in Sections 2.1 and Sections 3.1–3.3. Paloma Bescansa was responsible for Section 2.1 and assisted in the final editing process. All the authors contributed to the conclusions (Section 4) and enriched the discussion in Section 3.3.

**Conflicts of Interest**

The authors declare no conflict of interest.

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