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# Linking ecosystem condition and ecosystem services: A methodological approach applied to European agroecosystems

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ABSTRACT

Agriculture has been identified as one of the main drivers of environmental degradation in the European Union (EU). It can have negative impacts on air, water, soil and biodiversity. The condition of agroecosystems is affected by soil degradation, especially by soil erosion, which reduces agroecosystems' capacity to provide essential ecosystem services. Therefore, it is necessary to explore synergies and trade-offs between pressures, ecosystem condition and services to create relevant information for policy and decision-makers to implement sustainable response actions.

As part of the EU environmental policy, the Mapping and Assessment of Ecosystems and their Services (MAES) Working Group developed appropriate concepts to assess and link ecosystem condition and services. This study aims to test the indicators proposed by MAES to assess ecosystem condition and link it with the ecosystem services provision. For this, we applied a suggested operational framework developed in the context of the Biodiversity Strategies 2020 and 2030 for the integrated assessment of agroecosystems and regulating ecosystem service control of erosion rates supply at European scale. We quantified and mapped indicators for ecosystem condition, environmental and anthropogenic pressures and soil erosion control. We explored the relationships between the respective indicators and the capacity of agroecosystems to control soil erosion across the different Environmental Zones (EZ).

Our results indicate that, in general, European agroecosystems have a high capacity to control soil erosion with some variations within the EZ. Supply capacity is positively, negatively and not correlated with the various pressure and condition indicators. Management and climate indicators play a significant role in the assessment of this service. These results highlight that conservational management practices are fundamental to reduce soil loss and improve agroecosystem condition. However, the design and implementation of such management practices must consider regional and local landscape characteristics, agricultural practices and the needs and opportunities of stakeholders.

## 1. Introduction

The degradation of ecosystems in the European Union (EU) has considerable economic and environmental consequences. Poor land management, unsustainable farming practices and urbanization are the principal causes (Panagos et al., 2018). Additional to these pressures, climate change increases the effects of soil erosion and loss of organic carbon (Borrelli et al., 2020). In this sense, it is necessary to ensure that ecosystems are in good condition and resilient to sustain human well-being in the long term (Maes et al., 2020a). An ecosystem is in good condition if it supports biodiversity, provides a balanced supply of ecosystem services and if abiotic resources such as soil, water and air are not depleted (Maes et al., 2018). Additionally, an ecosystem is resilient if it can maintain its structures and functions after a possible disturbance (Müller et al., 2010; see Box 1).

The EU environmental policies integrate the related ecosystem condition and ecosystem services concepts to address environmental threats. Thus policy makers require knowledge about ecosystem condition, the factors that affect it, the pressures to which they are subjected and the effects on ecosystem services to design effective policies and

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management strategies (Maes et al., 2020a). The EU Soil Thematic Strategy (expected to be updated in 2021) is part of the efforts to protect soil fertility, reduce soil erosion and increase soil organic matter (European Commission, 2020a). Therefore, a better understanding of ecosystem condition and its relationship with ecosystem services would sustain the implementation of actions that contribute to achieving land degradation neutrality through soil health and functions restoration (see Box 1 for definitions).

The research on ecosystem condition in the EU has increased in the last decades (see review paper Rendon et al., 2019). The Working Group on Mapping and Assessment of Ecosystems and their Services (MAES) of the European Commission, which was installed in the European Union's Biodiversity Strategy 2020 implementation, has done comprehensive research on ecosystem condition. MAES developed, mapped and quantified a series of pressures and condition indicators for European ecosystems (Maes et al., 2020a, 2018). Other initiatives include the System of Environmental Economic Accounting (SEEA) from the United Nations (UN) (United Nations, 2020) and the ongoing work of the European Environment Agency (EEA). At the national and regional levels, there have been some studies assessing the condition of wetlands and forests in Greece (Hatziiordanou et al., 2019; Kokkoris et al., 2018), urban ecosystems in Bulgaria (Nedkov et al., 2017), and agroecosystems in Northern Germany (Rendon et al., 2020). Despite this ongoing research on ecosystem condition (or comparable concepts such as ecosystem health, state, quality), it is still necessary to explore the synergies and trade-offs between ecosystem condition and ecosystem services (Rendon et al., 2019).

As an initial step to fill this research gap, we assessed agroecosystem condition in the EU and related it with the ecosystem service *control of erosion rates*. We selected this ecosystem type because agriculture is among the main drivers of environmental degradation in the EU. It is responsible for a great deal of greenhouse gases emissions, water consumption and soil erosion in the European territory (Recanati et al., 2019). At the same time, intensive agriculture affects soil and ecosystem condition reducing their capacity to provide essential services such as food, fodder and water provision, water quality regulation, and organic carbon storage (European Environment Agency, 2019a).

We focused on the soil erosion narrative, particularly on erosion by water, because this is one of the principal pressures on European soils (Panagos et al., 2015e). Other threats include soil erosion by wind and tillage, soil compaction, sealing, salinization and contamination. All these pressures are increasing and are causing negative impacts on soil functions and services (Solte et al., 2016). Apart from the effects on ecosystem services and soil functions, soil erosion has significant repercussions on the economy, with an estimated annual cost of EUR 1.25 billion for the agricultural sector (Panagos et al., 2018). Furthermore, soil erosion is likely to increase in the future due to more heavy rainfall events associated with climate change (Panagos et al., 2017) and an increase in land use intensity and increasing field sizes (European Environment Agency, 2019a).

We based our assessment on the framework for integrated mapping and assessment of ecosystems and their services proposed by Burkhard et al. (2018) and the indicators proposed by Maes et al. (2018). These frameworks are part of the initiatives developed to support EU Member States in implementing Action 5 of Target 2 of the EU Biodiversity Strategy to 2020. Our objective was to test the indicators for pressures, condition and the ecosystem service *control of erosion rates* in agroecosystems at the European scale, by quantifying and mapping them and by analysing the relationships between them.

The paper is structured as follows: First, we describe the methodological approach. We then present the maps of the three different groups of indicators. Later, we analyse the relationships between pressure and

**Box 1**
. Definitions used in the assessment.

| | |
|---|---|
| Agricultural area | Area already used for farming or that could be brought back into cultivation using the resources normally available on an agricultural holding (EUROSTAT, 2020). |
| Arable land | Land worked (ploughed or tilled regularly), generally under a system of crop rotation excluding berry plantations, land taken out of cultivation and cultivated mushrooms (EUROSTAT, 2019a). |
| Ecosystem capacity | The ability of a given ecosystem to generate a specific 'Ecosystem service' in a sustainable way (Potschin-Young et al., 2016). |
| Ecosystem condition | The overall quality of an ecosystem unit in terms of its main characteristics underpinning its capacity to generate ecosystem services (Potschin-Young et al., 2016). |
| Ecosystem function | Subset of the interactions between biophysical structures, biodiversity and ecosystem processes that underpin the capacity of an ecosystem to provide ecosystem services (Potschin-Young et al., 2016). |
| Ecosystem health | Capacity of an ecosystem to maintain its organization and autonomy over time and to resist external pressures in relation to a desired (sustainable) reference condition or target (Rendon et al., 2019). |
| Ecosystem quality | Norm or a state with reference to what is considered as a good state for humans and societal needs (Roche and Campagne, 2017). |
| Ecosystem services | Contributions of ecosystem structure and function – in combination with other inputs – to human well-being (Burkhard et al., 2012). |
| Environmental Zones | Aggregation of environmental strata of Europe based on environmental variables such as climate, geomorphology, oceanicity and northing (Metzger et al., 2005). |
| Provision Capacity (PCAP) | In the assessment of the ecosystem service *control of erosion rates,* provision capacity (PCAP) is defined as the fraction of the structural impact that is mitigated by the service provision (Guerra et al., 2014; Steinhoff-Knopp and Burkhard, 2018). |
| Resilience | Ability of a system to reorganize after a disturbance and remain in the previous basin of attraction (Müller et al., 2010). |
| Soil health | The capacity of soil to function as a vital living system, within ecosystem and land use boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and promote plant and animal health (Doran and Zeiss, 2000). |
| Utilized Agricultural Area (UAA) | Total area taken up by arable land, permanent grassland, permanent crops and kitchen gardens used by the holding (EUROSTAT, 2019b). |

condition indicators with the ecosystem service provision capacity based on the environmental stratification (Environmental Zones) developed by Metzger et al. (2005). We discuss the results and the benefits of linking ecosystem condition and ecosystem services to potential policy implementation.

## 2. Material and methods

We followed a stepwise approach consisting of nine methodological steps based on the operational framework for integrated mapping and assessment of ecosystems and their services from Burkhard et al. (2018). The steps are (1) theme identification, (2) identification of ecosystem types, (3) mapping of ecosystem types, (4) definition of ecosystem condition and identification of the ecosystem services to be delivered by (agro)ecosystems, (5) selection of indicators for ecosystem condition and ecosystem services, (6) quantification of indicators, (7) mapping of indicators, (8) integration of results, (9) dissemination and communication of results (Fig. 1).

### 2.1. Theme identification (Step 1)

The first step of the operational framework is the identification of a question or a theme that must be addressed in the ecosystem assessment to be relevant for policy, society, business or science. In our study, we identified the policy objective of *maintaining healthy soils*, focusing on agricultural areas. Healthy agricultural soils have high biodiversity and fertility and provide multiple soil-related ecosystem services (e.g. food, fibre, climate and water regulation, water purification, carbon sequestration, nutrient cycling, and habitat) in a sustainable way (Paul et al., 2021; Rinot et al., 2019).

Agricultural soils in the EU face many threats, including erosion by water and wind, sealing, soil organic matter decline, contamination, compaction, salinization, loss of biodiversity, floods, and landslides (Turpin et al., 2017). In an attempt to mitigate these threats, prevent further soil degradation and improve soil health, the EU adopted the Soil Thematic Strategy in 2006. A new EU soil strategy closed a public consultation in April 2021 to receive input from various stakeholder groups, including citizens, environmental and non-governmental organisations (NGOs), industry, and researchers (European Commission, 2021).

Other policy instruments addressing the issue of degrading soils and agroecosystems are the Common Agricultural Policy (European Commission, 2019), the Farm to Fork Strategy (European Commission, 2020b) and the EU Biodiversity Strategies 2020 (European Commission, 2011) and 2030 (European Commission, 2020a). These policies and strategies highlight the relationships between agricultural production, including subsidies for farming, the environment and climate issues and identify the actions needed to reduce the pressures on agroecosystems. Such actions include the protection of soil fertility, the increase of crop diversification and soil organic matter, avoidance of soil erosion, reduction of pesticides and fertilisers use, and expansion of area under organic farming. The aim is to improve the condition and diversity of agroecosystems and to increase the resilience of the agricultural sector to climate change, environmental risks and socio-economic shocks. The instruments used to achieve these goals are, for instance, subsidies via the Cross-Compliance schemes of the Common Agricultural Policy (European Commission, 2020a).

### 2.2. Identification and mapping of ecosystem types (Steps 2 and 3)

The study area covers around 2.02 million km$^2$, corresponding to the agricultural area of the EU Member States (EU-27 and the United Kingdom) described as class 2 of the land use/land cover map CORINE (Coordination of Information on the Environment) for the reference year 2012 (European Environment Agency, 2012). Arable land covers 1.28 million km$^2$ (around 55% of the agricultural area), and the remaining agricultural area is covered by heterogeneous agricultural areas (22%), pastures (17%), and permanent crops (5%).

As climatic and natural conditions, and thus the characteristics of agriculture (e.g. agricultural production) in Europe, vary significantly, it was necessary to further subdivide the area to achieve more meaningful and reliable results. We used the environmental stratification of Europe

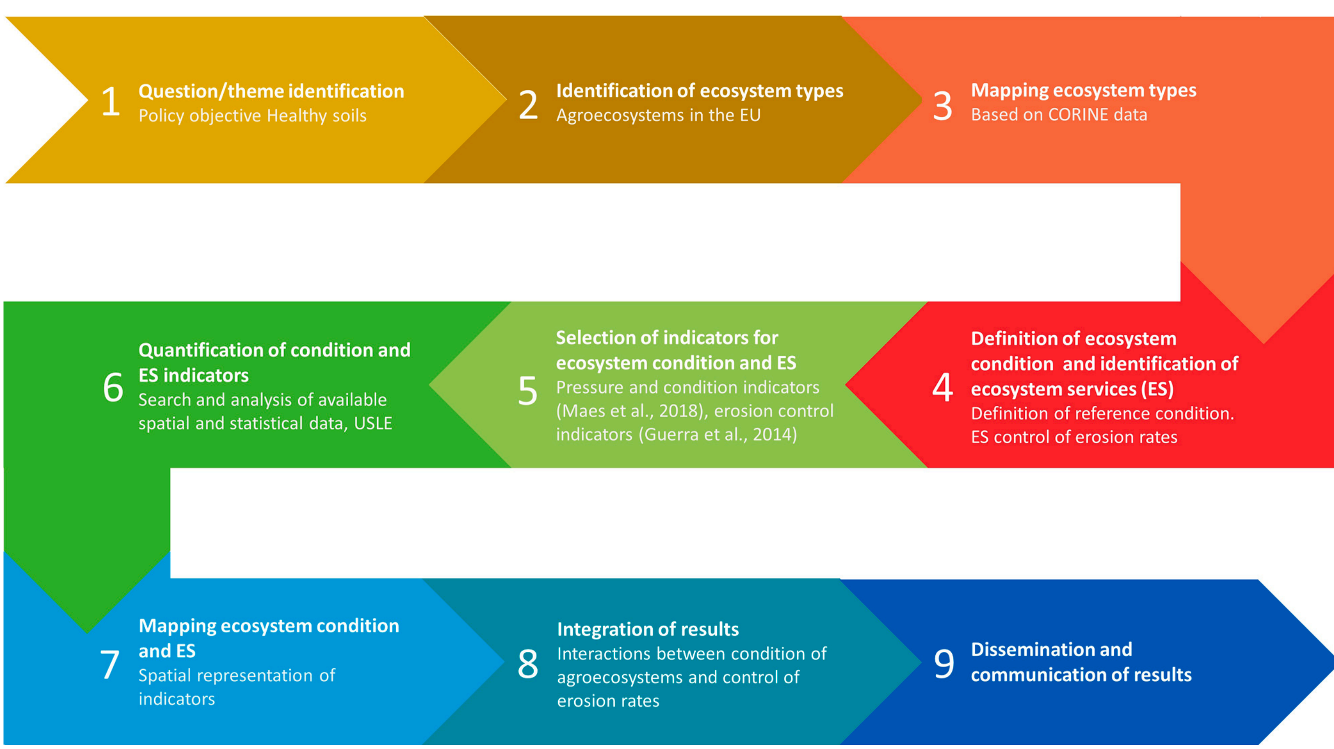


**Fig. 1.** Framework for integrated mapping and assessment of ecosystems and their services. (Step 9 is not included in this study). Based on Burkhard et al. (2018).

developed by Metzger et al. (2005) to identify relationships between indicators within zones with similar environmental characteristics. This stratification was created by statistically clustering climate and topographical variables aggregated into thirteen Environmental Zones (*EZ*). There are twelve *EZ* covered in the study area (see the map in Supplementary Information S2). Additionally, we performed a cluster analysis of the averages or each indicator per *EZ* to identify similarities and patterns between them (see Supplementary Information S3).

### *2.3. Definition of ecosystem condition and identification of ecosystem services delivered by agroecosystems (Step 4)*

Agroecosystems are heavily dependent on human management activities and so their condition cannot be compared to ecosystems in a natural state (Maes et al., 2018). However, for an agroecosystem to be in good condition, it requires a balance between the use of natural resources, the maintenance of biodiversity, the supply (and at the same time use) of various ecosystem services and the fulfilment of the needs of current and future generations (Maes et al., 2020b). Such services include provisioning services such as food, timber, fibres, fuels, pharmaceuticals, and energy crops (Power, 2010); regulating services such as climate and water regulation, water purification, carbon sequestration, nutrient cycling and habitat maintenance (Balmford et al., 2011); and cultural services such as landscape aesthetics and knowledge systems (Bernués et al., 2014; Burkhard et al., 2014). For this assessment, we used the median values of each indicator per *EZ* as reference values to determine the agroecosystem condition when there were no reference values or thresholds available in the literature (Table SI3).

We focused on soil erosion by water because it is the main threat to agricultural soils in the EU mainly through fertility and biodiversity loss. Soil erosion has other negative off-site effects such as sedimentation, siltation, eutrophication of water bodies, and increased risk of flooding and landslides (Borrelli et al., 2017; Panagos et al., 2015e). In this context, soil erosion control is an essential ecosystem service provided by agroecosystems quantified by two indicators: soil retention and the capacity of ecosystems to avoid soil erosion (PCAP, see Box 1) (Maes et al., 2015).

### *2.4. Selection, quantification and mapping of indicators for ecosystem condition and control of erosion rates (Steps 5 to 7)*

To assess the relationships between agroecosystem condition and the provision of the ecosystem service *control of erosion rates*, we selected 31 EU-scale indicators related to pressures (19 indicators), condition (10), and the ecosystem service *control of erosion rates* (2). We chose 13 of these pressures and condition indicators from the 5th MAES report (Maes et al., 2018). We based the selection on the following criteria: (i) relevancy (i.e., indicators are relevant for the provision of the ecosystem service control of erosion rates); (ii) availability of data at EU scale (i.e., data are spatially explicit and available at EU scale); (iii) quantifiable (i.e., indicators are quantifiable and can be compared among regions); and (iv) reliability (i.e., data are obtained from official sources) (see description of the criteria in Table SI1).

We imported spatially explicit data for all indicators into ArcGIS 10.7 for representation and analysis. When necessary, we resampled and aggregated data to grids of 1 km resolution to facilitate their comparison. Table 1 shows a detailed description of the indicators. The Supplementary Information (S1) explains the methodology used to calculate each indicator and its spatial representation.

### *2.5. Integration of results (Step 8)*

For the analysis of the data and integration of results, we created a grid over the study area with a pixel size of 1 km × 1 km containing altogether 1.145.550 pixels. We excluded pixels with missing values of any of the selected indicators. We analysed the data from two different angles: the analysis of the correlation structure in the dataset and the correlation between indicators. For the correlation structure, we randomly selected 0.01% of the pixels in each *EZ* to avoid spatial autocorrelation and bias and normalized the chosen indicators to a 0–1 scale to perform a Principal Component Analysis (PCA) (see Supplementary Information SI3). Additionally, we plotted the spatial data in a two-dimensional principal components biplot explaining a high percentage of the variance in the data. The significant number of factors in the PCA were those with eigenvalues > 1. To assess the correlation between pressures and condition indicators, and provision capacity, we conducted a Spearman correlation (see Table SI5 in the Supplementary Information). For this analysis, we considered all the pixels in the study area. We created two-dimensional box plots to graphically show the results per *EZ*. In each axis, the first and third quartile of the distributions indicates the limits of the boxes. The middle line (horizontal or vertical) inside the boxes indicates the median value (see Fig. 2). The statistical work was conducted in RStudio (version 1.2.1335) (RStudio Team, 2015).

## 3. Results

### *3.1. Mapping and assessment of pressures, condition and control of erosion rates in agroecosystems*

We calculated and mapped the indicators for pressures, ecosystem condition and control of erosion rates using the datasets described in Section 2. Table SI3 presents the median and mean values of indicators per *EZ*. The maps covering the distribution of the indicators in the entire study area are compiled in Fig. 3.

A clustering based on the average values per indicator in the twelve *EZs* revealed some similarities between them according to Euclidean distance metrics. The analysis showed three distinct clusters. The first cluster, Northern Europe and *Alpine South*, comprises four *EZs* with similar climatic characteristics: *Alpine North, Alpine South, Boreal and Nemoral.* The second cluster entails five *EZs* in Central Europe: *Atlantic Central, Atlantic North, Continental, Lusitanian* and *Pannonian.* The third cluster comprises the Mediterranean zones *North, South* and *Mountainous.* The results of the indicators per cluster are summarized in the following subsections.

#### *3.1.1. Pressure indicators*

The *changes in agroecosystems extent* in the study area were comparably low for the selected reference period (2012–2018). We observed a median value of 0% and a mean value of −0.06% related to 2012 in all *EZ* (Fig. 3-1), indicating a low decrease in the agricultural area. The precipitation indicators *mean daily precipitation*, and the *number of days with precipitations ≥ 10 and ≥ 20 mm*, showed an uneven distribution throughout the continent. The *mean daily precipitation* and the *number of days with precipitation ≥ 10 mm* were the highest in the first cluster (Northern Europe and *Alpine South*) (Figs. 3-2 and 3-3). Whereas Mediterranean zones showed the lowest values for these two indicators but the highest for the *number of days with precipitation ≥ 20 mm* (Fig. 3-4).

As expected, we observed the highest *maximum, minimum* and *mean temperatures* in the Mediterranean zones with values way above the average. In contrast, the northern regions showed the lowest minimum temperatures with average values as low as −8.79 in *Alpine North* (Figs. 3-5 to 3-7). The *growing season length* is strongly correlated to the temperature. Hence, we observed a similar distribution of this indicator in the study area. The Mediterranean, and central zones had the longest growing season and Northern Europe had the shortest (Fig. 3-8).

*Soil moisture* had the lowest values in Northern Europe and *Alpine South*. The opposite occurred in the central and southern regions (see Fig. 3-9). On the other hand, *land-use intensity*, indicated by the agricultural energy input, was low in the *Alpine North* and *South* and high in the Mediterranean zones. We observed the highest input levels especially in Italy, Spain, The Netherlands, Belgium and Germany (Fig. 3-

**Table 1**
Indicators used for the assessment of environmental pressures and condition of agroecosystems and the ecosystem service *control of erosion rates* in the EU.

| Indicator class | Indicator | Code | Description | Units | Spatial resolution original data | Reference period / year | Source |
|---|---|---|---|---|---|---|---|
| **Pressure indicators** | | | | | | | |
| Habitat conversion and degradation | Change in ecosystem extent | CE | Change in the area (size) of agroecosystems within the years 2012 and 2018. | % per year | 100 m | 2012–2018 | (European Environment Agency, 2019b) |
| Climate | Mean daily precipitation | RR | Mean daily precipitation | mm per day[1] | 0.1 degrees | 1980–2018 | (Cornes et al., 2018)[1] |
| | Precipitation $\geq$ 10 mm | P10 | Heavy precipitation days: Average number of days per year where precipitation was equal or higher than 10 mm. | Number of days per year[1] | | | |
| | Precipitation $\geq$ 20 mm | P20 | Very heavy precipitation days: Average number of days per year where precipitation was equal or higher than 20 mm. | | | | |
| | Maximum temperature | TX | Average of the daily maximum temperature. | C° | | | |
| | Minimum temperature | TN | Average of the daily minimum temperature. | | | | |
| | Mean temperature | TG | Average of the daily mean temperature. | | | | |
| | Growing season length | GSL | Average number of days between the first occurrence of at least 6 consecutive days with a daily mean temperature higher than 5 °C, and the first occurrence after 1 July of at least 6 consecutive days with temperatures lower than 5 °C. | Number of days[1] | | 1950–2018 | |
| | Summer soil moisture | SM | Mean daily soil water content in the upper soil horizon (up to 1 m). | l $m^{-3}$ | 0.25 degrees | 1980–2016 | (Kurnik et al., 2015) |
| Over-exploitation | Land-use intensity | LUI | Energy inputs as proxy for land-use intensity. | MJ $ha^{-1}$ | 1000 m | 2015 | (Pérez-Soba et al., 2015) |
| Others | Soil erosion | SE | Amount of soil loss per hectare in a year (Actual soil loss). | t $ha^{-1}$ per year | 100 m | 2010 | (ESDAC, 2015; Panagos et al., 2015f, 2015e, 2015c, 2015a, 2015b, 2015d, 2014a) |
| | Loss of organic matter | SOCL | Soil organic carbon (SOC) eroded from agricultural areas. | t $ha^{-1}$ per year | 1000 m | 2000–2010 | (ESDAC, 2014a; Lugato et al., 2016) |
| Management | Share of conventional tillage | CNT | Percentage of arable land under conventional tillage. | % | NUTS2 | 2016 | (EUROSTAT, 2019c) |
| | Share of conservation tillage | CST | Percentage of arable land under conservation tillage. | | | | |
| | Share of zero tillage | ZT | Percentage of arable land under zero tillage. | | | | |
| | Soil cover: bare soil | SCB | Percentage of arable land with bare soil in the winter season. | | | | |
| | Soil cover: plant residues | SCPR | Percentage of arable land covered with plant residues in the winter season. | | | | |
| | Soil cover: intermediate crops | SCI | Percentage of arable land covered with cover crops or intermediate crops in the winter season. | | | | |
| | Soil cover: winter crops | SCW | Percentage of arable land covered with winter crops in the winter season. | | | | |
| **Condition indicators** | | | | | | | |
| Structural ecosystem attributes (general) | Crop diversity | CD | Average number of crops in a 5 arcmin raster cell. | Number of crops | 5 arcmin | 2010 | (EUROSTAT, 2010a) |
| | Density of semi-natural areas | SN | Number of 25 m cells classified as woody vegetation in a 100 m agricultural cell. | Number of cells (0 to 16) | 100 m | 2012 | (Rega et al., 2018) |
| | Density of hedgerows | DH | Hedgerow units: number of intersections with linear landscape elements. | Number of intersections | 1000 m | 2015 | (Joint Research Centre (JRC) (2015)) |
| | Share of fallow land in Utilised Agricultural Area (UAA) | FL | Percentage of arable land that is not being used for agricultural purposes within the UAA. | % | 5 arcmin | 2010 | (EUROSTAT, 2010a) |
| | Share of arable land in UAA | AL | Percentage of land used for the production of crops within the UAA. | | | | |
| | Share of permanent crops in UAA | PC | Percentage of land used for permanent crops within the UAA. | | | | |
| | Livestock Density | LD | Stock of animals (cattle and ovine) converted in livestock units (LU) per hectare of UAA. | LU $ha^{-1}$ | 5 arcmin | 2010 | (EUROSTAT, 2010b) |
| Structural soil attributes | Soil Organic Carbon | SOC | Concentration of topsoil organic carbon in the 0–30 cm layer. | % | 250 m | 2010 | (ESDAC, 2014a; Lugato et al., 2014) |
| | Soil erodibility | KF | Susceptibility of soil to erosion by runoff and raindrop impact. | K factor [t $ha^{-1}N^{-1}$] | 500 m | 2014 | |

(continued on next page)

**Table 1** (*continued*)

| Indicator class | Indicator | Code | Description | Units | Spatial resolution original data | Reference period / year | Source |
|---|---|---|---|---|---|---|---|
| | Bulk density | BD | Weight of soil per cubic meter. | t m$^{-3}$ | 500 m | 2015 | (ESDAC, 2014b; Panagos et al., 2014a, 2012) (Ballabio et al., 2016; ESDAC, 2016) |
| **Ecosystem service indicators** | | | | | | | |
| Control of erosion rates | Soil retention | CER | Actual ecosystem service provision: tons of soil not eroded. | t ha$^{-1}$ per year | 100 | 2010 | (Maes, 2010) |
| | Provision capacity | PCAP | Share of mitigation of soil erosion (0 to 1). | Dimensionless | | | |

[1] E-OBS dataset from the EU-FP6 project UERRA (https://www.uerra.eu) and the Copernicus Climate Change Service, and the data providers in the ECA&D project (https://www.ecad.eu).

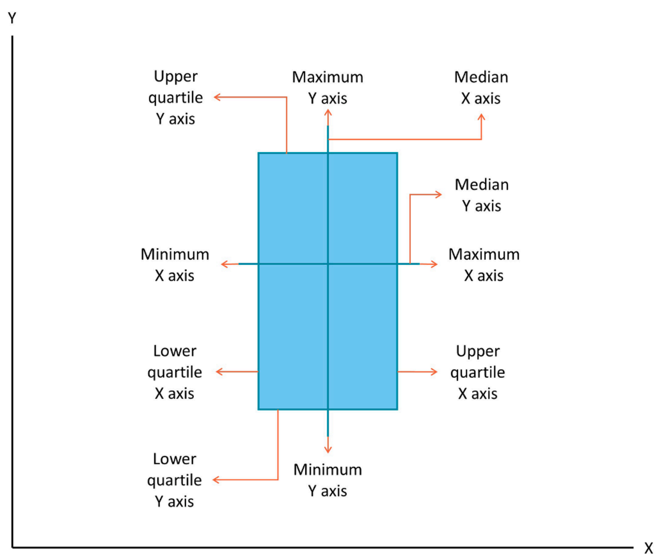


**Fig. 2.** Example 2D Box plot.

10).

*Soil erosion rates* recorded the highest values in the Mediterranean zones and the lowest in Central Europe. Based on the classification of soil erosion from EUROSTAT (EUROSTAT, 2019d), we found that areas classified as having low erosion rates (<5 t ha$^{-1}$ per year) represented about 85% of the total agricultural area in the EU-27 and the UK. Moderate soil erosion values (5–10 t ha$^{-1}$ per year) accounted for about 9% of the area, and the remaining 6% were under severe erosion (>10 t ha$^{-1}$ per year) (Fig. 3-11). *Loss of organic matter* is closely related to soil erosion and showed average values of 0.10 t ha$^{-1}$ per year. *Alpine South* and *Mediterranean Mountainous* showed the highest rates, and Central Europe the lowest ones (Fig. 3-12).

Conventional tillage was the most widespread tillage practice in the study area, covering around 65.2% of the arable land. Conservation tillage covered 17.7% of the arable land, while zero tillage covered only 3.32 %. Central Europe had the highest mean share of conservation tillage, and Mediterranean zones had the highest mean share of arable land under zero tillage. Cyprus had the highest percentage of conservation tillage, with more than 50% of the arable land under this practice. In Estonia and Romania, the proportion of zero tillage was the highest in the study area (Figs. 3-13 to 3-15).

The different types of soil cover in the winter season on agricultural areas were quite varied. An average of 23% of the arable land was left bare, with the highest shares in Northern Europe (>29%). Plant residues and intermediate crops occupied shares of 8.41% and 7.36%, respectively. Soils were covered by plant residues principally in the Mediterranean zones. Intermediate crops had higher proportions in Central Europe. Winter crops, on the other hand, covered more than 41% of the agricultural area, with the highest shares observed in the *Mediterranean zones* (Figs. 3-16 to 3-19).

### 3.1.2. *Ecosystem condition indicators*

The spatial distribution of crop diversity in the agricultural area was strongly influenced by climatic conditions. We found the highest number of crop types in Central Europe, with average values of 14 crops per 5 arcmins raster cell (about 4.6 km × 4.6 km in Europe). Medium to low diversity in Mediterranean zones, and low diversity in the northern part and *Alpine South*. Poland, parts of Romania and Austria had the highest crop diversity with cells with values > 20 (Fig. 3-20). Cereals were the second most common crop category after permanent grasslands and meadows with more than 54 million ha. The least common category was fodder roots and brassicas with 44840 ha.

The *density of semi-natural areas* was higher in Northern Europe with average values of 1.49 out of 16 (see Supplementary Information S1 for a detailed description of the indicator). We also observed high shares in the Iberic Peninsula and Italy with a high proportion of cells with values > 10 on the scale from 0 to 16. In contrast, Central Europe showed a low share, with average values of 0.63 (Fig. 3-21). The *density of hedgerows* was high in Central Europe and the Mediterranean zones, with the highest number of intersections with linear landscape elements on 250 m transects. We observed the highest densities in Ireland, north-west Spain and north-west France. The opposite occurred in Northern Europe, which showed very low averages (Fig. 3-22).

The share of *fallow land* in *Utilized Agricultural Area (UAA)* was small in the whole study area, with an average value of 3.85% (around 6.7 million ha). The *EZs* with the highest share are located in the Mediterranean zones and Northern Europe. We found high shares of fallow land in Greece, Spain, Portugal and Romania that had cells with values higher than 70%. In contrast, we observed the lowest values in Central Europe with mean shares below 2% (Fig. 3-23). On the other hand, the *share of arable* land in UAA was high in most of the study area, with an average of 85.65%. Central Europe had the highest percentages with average values higher than 89%, whereas Northern Europe and Alpine South had the lowest with an average of 80.16% (Fig. 3-24).

*Permanent crops* covered an area of 10.4 million ha, with the most common crops being olives (4.3 million ha) and vineyards (2.9 million ha). Permanent crops concentrate in the Mediterranean zones, with an average share of 15.6% of the agricultural area. Parts of Spain, France, Italy, Greece and Cyprus had the highest number of 1 km raster cells with shares of permanent crops greater than 85%. The rest of the EU had an average share of 1.56% (Fig. 3-25). *Livestock density*, including only cattle and ovine, was relatively low in the EU, with an average value of 0.63 LU ha$^{-1}$. It was highest in Central Europe, especially in Ireland, Denmark, The Netherlands, northern Germany, southern England and France. We observed low livestock density in Northern Europe and mountainous regions (Fig. 3-26).

*Soil Organic Carbon* (SOC) levels in agricultural areas were relatively

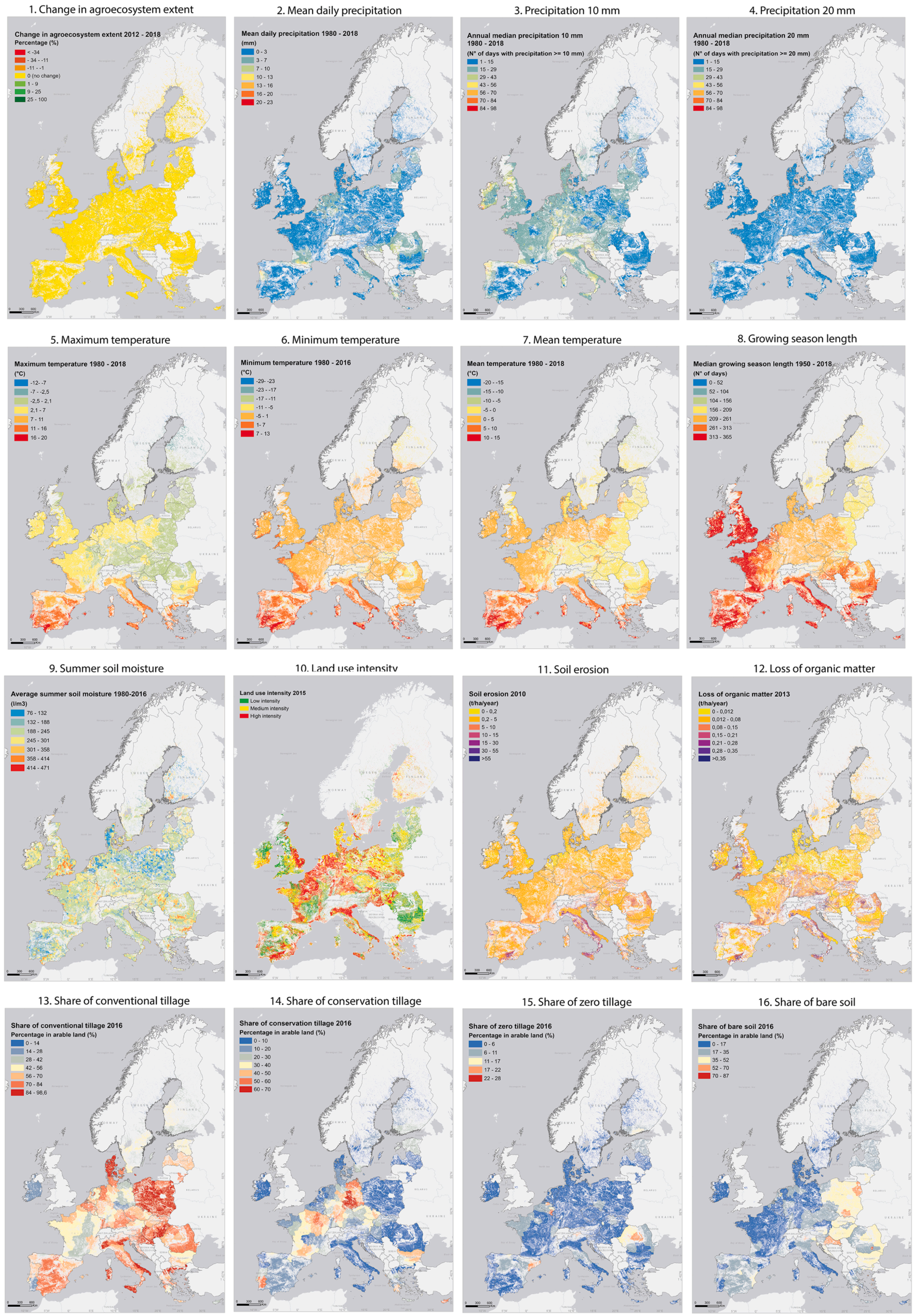


**Fig. 3.** Maps of indicators of environmental pressures, ecosystem condition and control of erosion rates in the EU (larger maps are provided in the Supplementary Information S4).

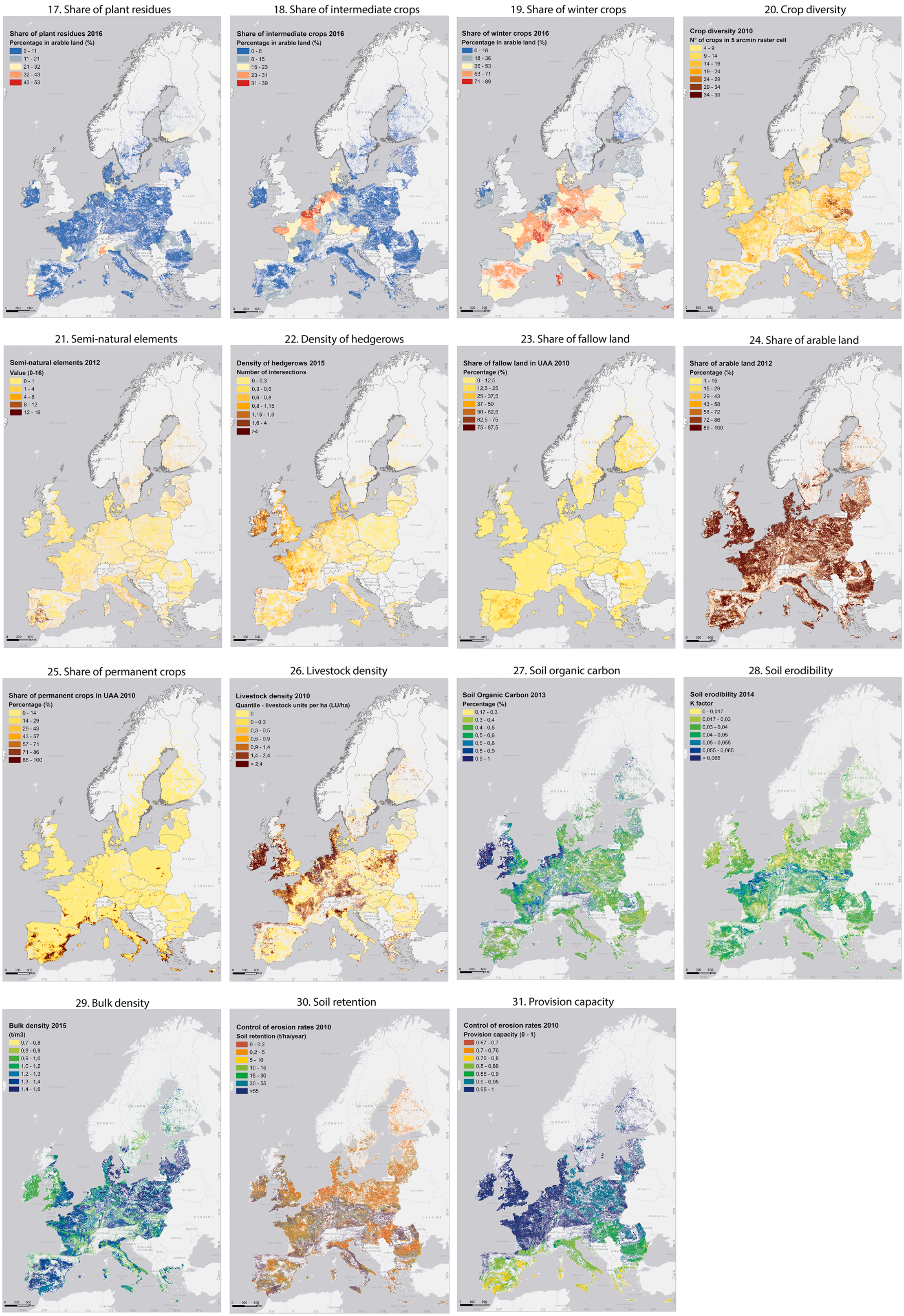


**Fig. 3.** (*continued*).

low (mean value of 0.57%) with significant regional differences. We observed the highest SOC in Northern Europe and *Alpine South*. In contrast to the Mediterranean zones that had the lowest SOC (Fig. 3-27). *Soil erodibility* values ranged from 0.004 to 0.075 t $ha^{-1}N^{-1}$, with an average of 0.03 t $ha^{-1}N^{-1}$. We found the highest soil erodibility in the Mediterranean and central zones, mainly in Spain, France, Belgium, Czech Republic, Hungary, and Slovakia. Areas with high SOC principally in Ireland, Denmark, the UK, The Netherlands and Finland, had the

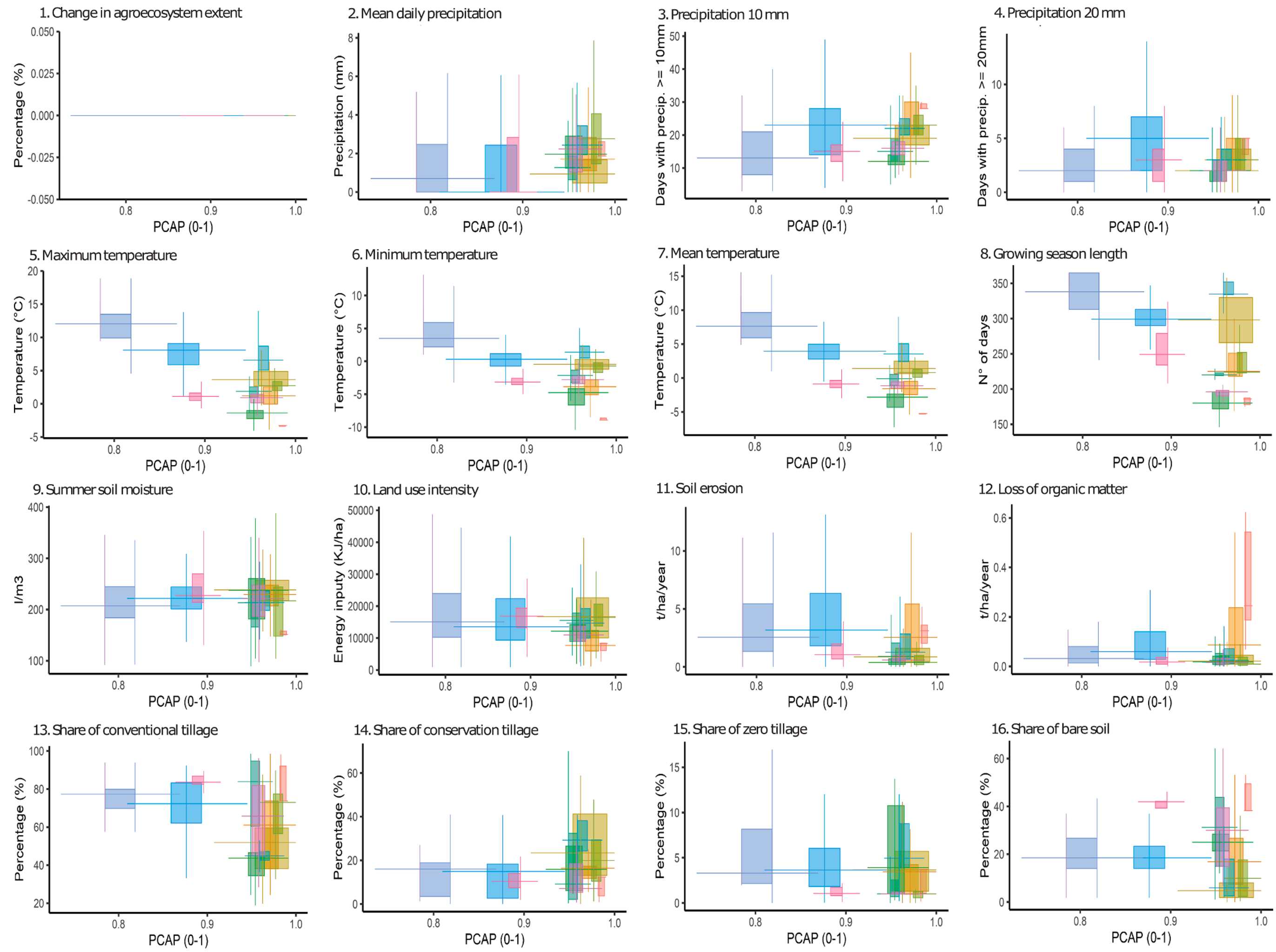


**Fig. 4.** Relationships between the indicators of environmental pressures and condition and the capacity of the ecosystem to control erosion rates (PCAP) per environmental zone (*EZ*). ALN: Alpine North, ALS: Alpine South, ATC: Atlantic Central, ATN: Atlantic North, BOR: Boreal, CON: Continental, LUS: Lusitanian, MDM: Mediterranean Mountainous, MDN: Mediterranean North, MDS: Mediterranean South, NEM: Nemoral, PAN: Pannonian.

lowest soil erodibility (Fig. 3-28). *Bulk density* values ranged from 0.66 to 1.56 t $m^{-3}$ and had an average of 1.22 t $m^{-3}$. Northern Europe and *Alpine South* had values below the average, whereas we observed the highest average values in Central Europe and Mediterranean Zones with 1.25 t $m^{-3}$. Estonia, Latvia, Lithuania, Poland and northern Germany had the most cells with bulk density higher than 1.5 t $m^{-3}$ (Fig. 3-29).

### 3.1.3. Ecosystem service indicators

*Soil retention* was on average 32.3 t $ha^{-1}$ per year. We observed the highest mean values in Northern Europe and *Alpine South* with an average of 49.69 t $ha^{-1}$ per year. Central Europe had the lowest soil retention values with an average of 20.75 t $ha^{-1}$ per year. Median soil retention was lowest in *Alpine North* (0.00 t $ha^{-1}$ per year) and highest in *Alpine South* (30.15 t $ha^{-1}$ per year) (Fig. 3-30).

*Provision capacity* was relatively high in agricultural areas, with an average of 0.93 on the scale from 0 to 1, indicating high provision capacity. However, there are considerable variations within the *EZs*. The Mediterranean zones had the lowest mean provision capacity (0.83). In contrast, Northern and Central Europe and *Alpine South* had the highest provision capacity with average values > 0.95. Median values range between 0.78 (*Mediterranean South*) and 0.99 (*Alpine North*) (Fig. 3-31).

## 3.2. Relationships between agroecosystem condition and control of erosion rates

We performed a systematic analysis of the relationships between the pressure and ecosystem condition indicators with the capacity of agroecosystems to control soil erosion. Fig. 4 shows two-dimensional box plots of the pressures, ecosystem condition and soil retention indicators against provision capacity for the twelve *EZs* in our study area. Each box shows data for an *EZ* and summarizes two distributions: the specified pressure, condition or soil retention indicator in the vertical axis and the *provision capacity* in the horizontal axis.

Our results show that *change in ecosystem extent* was very low and had a median of 0%. Therefore, the correlation between this indicator and provision capacity was not significant ($p = 0.36$, Fig. 4-1). Comparisons between *EZs* showed that the precipitation indicators had a positive correlation with provision capacity ($p < 0.05$, Fig. 4-2–4-4), whereas temperature indicators and *growing season length* had a negative correlation with it ($p < 0.05$, Figs. 4-5 to 4-8). *Soil moisture* was positively correlated, but this correlation was not significant ($p < 0.05$, $r_s = 0.1$, Fig. 4-9). *EZs* with higher *land use intensity*, *soil erosion* and *loss of organic matter* had lower *provision capacity* ($p < 0.05$, Fig. 4-10 to 4-12). However, these correlations were not significant ($r_s = -0.14$, $r_s = -0.4$ and $r_s = -0.1$, respectively).

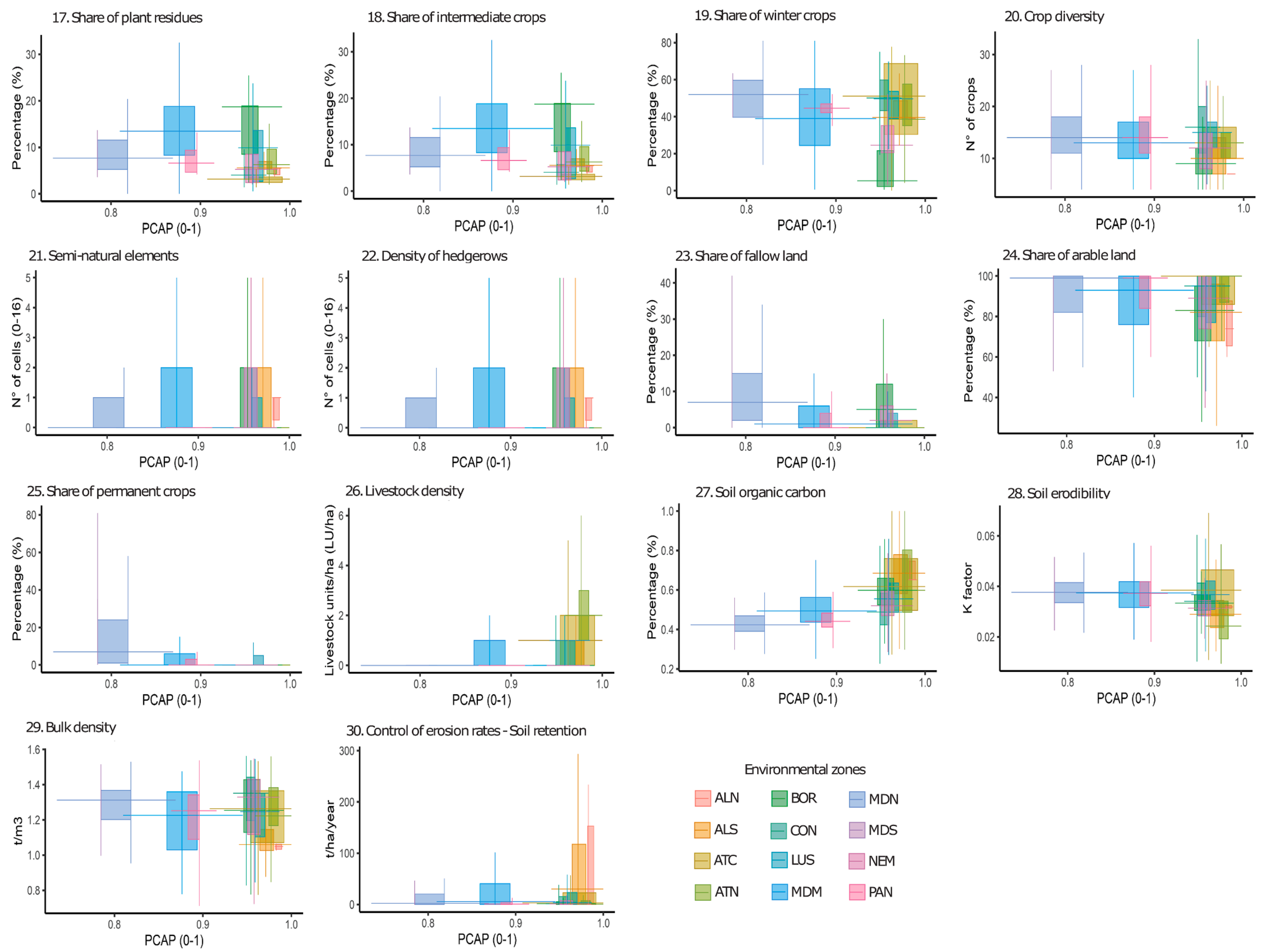


**Fig. 4.** (*continued*).

When looking at the influence of tillage practices on the capacity of agroecosystems to control soil erosion, we identified positive and negative correlations. *Provision capacity* was low in areas with high shares of *conventional* and *zero tillage* and higher in zones with high *conservation tillage* ($p < 0.05$, Figs. 4-13 to 4-15). Similarly, management practices with different soil cover over winter had positive and negative correlations with provision capacity. The share of *intermediate crops* was high in areas with high *provision capacity*, while shares of *winter crops*, *plant residues* and *bare soil* were high in zones with lower capacity ($p <$ 0.05, Figs. 4-16 to 4-19).

The indicators related to structural agroecosystem attributes had a negative correlation with provision capacity, except for the *density of hedgerows*. Zones with higher *crop diversity* and *livestock density*, higher shares of *fallow* and *arable land* and *permanent crops* had lower provision capacity ($p < 0.05$, Figs. 4-20, 4-23 to 4-26). On the other hand, capacity was high in areas with a high *density of hedgerows* ($p < 0.05$, Figs. 4-22). The share of *semi-natural vegetation* was not significantly correlated ($p =$ 0.85, Fig. 4-21).

The indicators of structural soil attributes and control of erosion rates had positive and negative correlations with provision capacity. *Soil organic carbon* was high in areas with high capacity, while *soil erodibility* and *bulk density* were higher in areas with lower capacity ($p < 0.05$, Fig. 4-27 and 4-28). Bulk density and *soil retention* were not significantly correlated with provision capacity ($r_s$ = -0.07 and $r_s = 0.03$, Fig. 4-29 and 4-30).

According to the PCA (see Supporting Information S3), the climate indicators related to temperature and growing season length, permanent crops and provision capacity had the highest significance in the whole study area. When looking at the *EZs* separately (Fig. 5 and Table SI4), the indicators contributed to the variance prediction differently. The *share of bare soil* had a meaningful contribution in all *EZs*. The *average temperature* had the most meaningful contribution to the variance prediction in Northern Europe, and *conventional tillage* in Central Europe. In the Mediterranean zones, indicators of *precipitation intensity* and *bulk density* had the highest contribution. Fig. 5 shows the most significant indicators in each *EZ* based on the factor loadings (>0.5 or $< -0.5$) from the PCA (see the values per indicator in Table SI4).

## 4. Discussion

This study aimed to test the indicators for pressures, ecosystem condition and control of erosion rates in agroecosystems at the European scale. These indicators offer valuable information about the areas with high pressures, limiting conditions and a high soil erosion risk, in which it is necessary to implement measures to improve ecosystem condition and prevent or mitigate soil loss. In this section, we discuss the relationships between the indicators at the EU and *EZ* levels. We then focus on the differences in control of erosion rates per *EZ* and reflect on the limitations of the assessment. The paper concludes with a discussion of the potential applications of our approach in policy and decision-making.

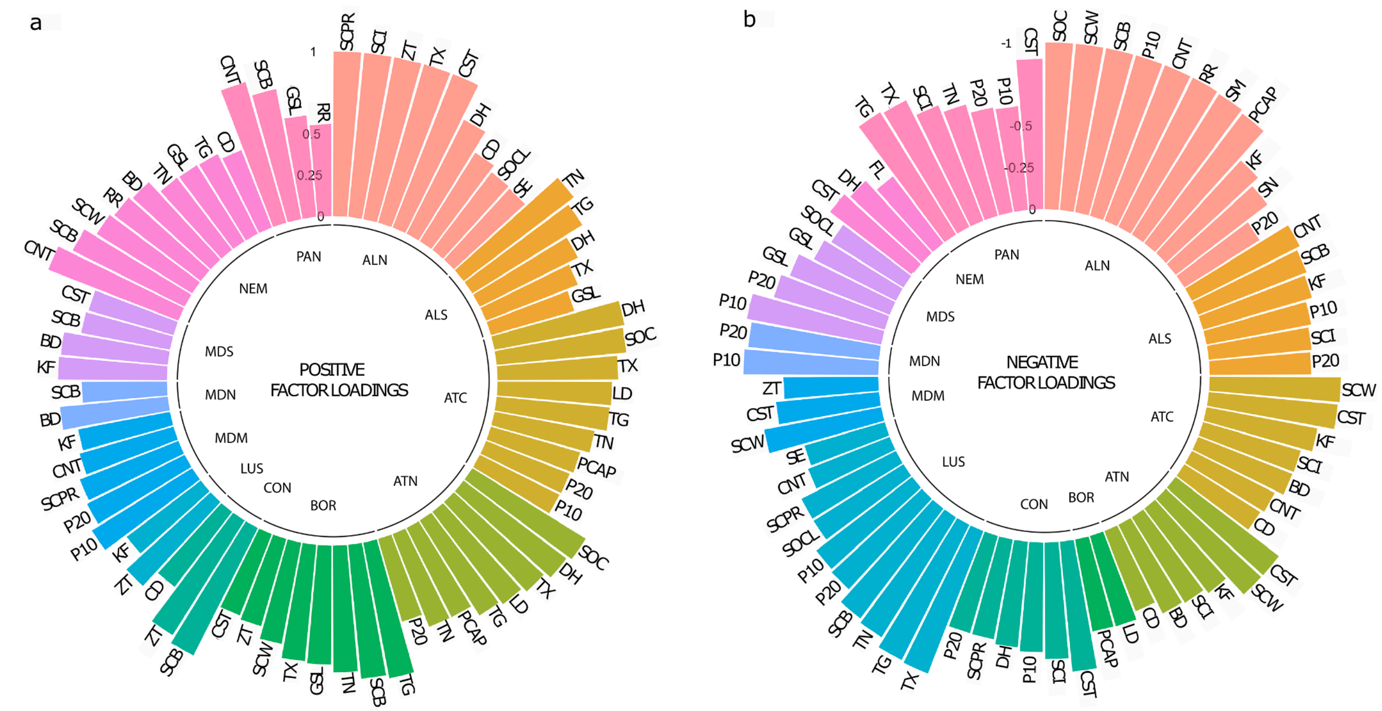


**Fig. 5.** High significance of indicators based on the PCA per Environmental Zone (*EZ*): (a) significant positive factor loadings (>0.5), (b) significant negative factor loadings (<-0.5). ALN: Alpine North, ALS: Alpine South, ATC: Atlantic Central, ATN: Atlantic North, BOR: Boreal, CON: Continental, LUS: Lusitanian, MDM: Mediterranean Mountainous, MDN: Mediterranean North, MDS: Mediterranean South, NEM: Nemoral, PAN: Pannonian.

### *4.1. Relationships between pressures, ecosystem condition and control of erosion rates*

#### *4.1.1. At the European Union level*

We did not find significant correlations between the indicators of pressures, ecosystem condition and *control of erosion rates* at the continental level (see Table SI5). The main reason for this might be the marked differences between the landscape characteristics, climate and cropping patterns and variations in soil erosion processes throughout Europe. Additionally, the resolution used in the study (1 km × 1 km) might play a role in this non-significant correlation, as soil erosion processes occur at local scales, which may not always be visible at this resolution. Furthermore, some indicators do not provide enough information about the pressures or condition of agroecosystems. Principally due to data availability, but also because of the way they are formulated.

The indicator *change in ecosystem extent* needs a more refined analysis to evaluate the relationships with ecosystem services, including the *control of erosion rates*. This analysis should cover aspects such as (i) changes in agricultural management systems, (ii) identification of the nature of the changes (i.e., within agricultural land use/land cover, and from or to other land uses), and (iii) the differences between *EZ*s. With such a refined analysis, we would better understand the effects of change in ecosystem extent and condition on the provision of this service.

Appropriate site-specific management of crops is essential to enhance soil quality and prevent its degradation. The benefits of soil conservation practices are substantial in reducing erosion (Borrelli et al., 2017). Soil protection with vegetation can compensate for the effects of erosive rain because plants intercept rainfall and allow water infiltration, changing the topsoil structure. In this sense, the use of cover crops and the application of reduced tillage can enhance fertility and control runoff (Panagos et al., 2016). Our results, however, show little correlation between *provision capacity* and *zero tillage* but a higher correlation with *conservation tillage* at the continental level. Management practices involving soil cover with *plant residues* in the winter season or *winter crops* did not correlate with provision capacity. The correlation was positive with *intermediate crops* and negative with *bare soil*, indicating the high relevance of soil cover in winter to reduce soil erosion.

*Soil retention* did not correlate with almost any of the indicators we assessed, including provision capacity both at the continental and *EZ* level. This finding aligns with the framework we used in this study proposed by Guerra et al. (2014) that demonstrates a significant difference between soil retention and provision capacity. Our results show that areas with high provision capacity do not necessarily have high soil retention (the actual ecosystem service provision). Therefore, it is essential to communicate these differences to avoid misinformation and erroneous interpretations of the results. Additionally, this differentiation helps to identify vulnerable areas and to target mitigation or restoration measures, e.g. implementing buffer strips or hedges, reducing livestock density and applying conservation tillage.

We are aware that there might be autocorrelation within the datasets since some pressure and condition indicators are also part of the soil erosion modelling. However, we did not address this issue in this study since the aim was to identify the relationships between the pressures and condition indicators proposed by MAES and the ecosystem *service control of erosion rates* calculated based on the USLE model.

#### *4.1.2. At the environmental zones (EZ) level*

We identified considerable differences in the correlations between indicators and their relevance within the *EZs* (Fig. 5). For example, our results align with theoretical expectations for *semi-natural vegetation* and *density of hedgerows*: most regions with high *land use intensity* had a low abundance of *semi-natural vegetation*. The opposite occurred in *EZs* such as *Alpine South* that features low-intensity agriculture. Previous research has shown that a high share of semi-natural landscapes correlates positively with the supply of multiple ecosystem services since these landscapes are not optimised to provide a single ecosystem service (García-Feced et al., 2015). However, this was only true for semi-natural vegetation in Northern Europe and the Alps and hedgerows in Central Europe. Probably, this is due to the multiple factors affecting soil erosion control, including climate, topography and management.

As expected, the *shares of arable land* and *permanent crops* were negatively correlated with *erosion control* in most of the study area. We observed high percentages of *arable land* and high *provision capacity* only in Central Europe. This positive correlation might be associated with the topography, as arable lands are mostly present in flat or gently sloping terrain (Maes et al., 2020a). Regions with high *shares of permanent crops*, such as the Mediterranean, had high *soil erosion rates* and low *provision capacity*, probably due to the olives crops and vineyards in hilly areas.

The Cover management factor (C-factor) in the USLE model is sensitive to reduced ground cover provoked by high cattle density but less sensitive to soil compaction that is also an effect of high cattle density. Accordingly, Guerra et al. (2014), who conducted an assessment at a local scale, emphasise that areas with high cattle breeding intensity often have lower control of erosion rates due to the grazing pressure on vegetation, whereas low-density areas have higher provision. Moreover, Rendon et al. (2020) also looked at the relationships between livestock density and erosion control in agricultural areas on a regional scale and found a negative correlation. However, when we compared *livestock density* with *provision capacity*, we found a positive correlation in the Alps and Central Europe. This difference might be related to the scale of the assessments (local and regional *vs* continental) and the different environmental conditions in the study areas (Mediterranean and Northern Germany *vs EZ*). Since our study covers a larger area, many regional features become somewhat fuzzy. Additionally, we used the mean values of the *EZ*s to evaluate the relationships between the indicators. It means that the cells with high livestock density do not necessarily overlap with the cells with high provision capacity.

*SOC* has a significant effect on *soil erodibility* and hence on *soil erosion control*. High values of *SOC* contribute to low *soil erodibility* (K factor of the USLE / RUSLE) values (Panagos et al., 2014a). We observed this relation in Southern and Western parts of the continent, whereas the Mediterranean zones and Central Europe had lower *SOC* and high *soil erodibility*. As expected, high erodibility in the Mediterranean zones and *Pannonian* contributed to low *soil retention* and *provision capacity*. We also found that in half of the *EZ*s, the *provision capacity* correlated negatively with *bulk density*. This correlation was principally evident in the Mediterranean zones, *Pannonian, Alpine South* and *North*.

We identified different threats for agricultural areas with low *soil retention* and low *provision capacity*, particularly in the Mediterranean zones and *Pannonian*. These threats relate to intensive *land use*, high *soil erosion*, a high *percentage of arable land*, low *soil organic carbon* and high *soil erodibility*. Additionally, these *EZ*s are subjected mainly to *conventional tillage* and less to sustainable practices such as *conservation* or *zero tillage* and *soil cover*. In this context, it is worth noting that the *control of erosion rates* in the Mediterranean will likely decrease even more in the coming decades. This predicted decline is due to climate change, land abandonment, urbanization and overgrazing (Guerra et al., 2014).

### 4.2. Differences in control of erosion rates per EZ

The EU covers a wide range of climatic and natural landscape conditions. Therefore, merging and analysing environmental data for the whole EU combines very different existing situations. It also creates a relatively blurred image of current ecosystem conditions and the potential supply of ecosystem services. For these reasons, we integrated the *EZ*s to analysed data on pressures, ecosystem condition and the indicators for the ecosystem service *control of erosion rates*. Only this kind of stratification allows comparing areas with similar environmental characteristics and can create relevant outcomes that can be used in regionally adapted policies and management decisions.

Precipitation, soil type, topography, land use, and land management are the main factors affecting soil loss rates (Panagos et al., 2015e). These factors vary considerably within the EU and the Member States, resulting in significant variations in soil erosion processes and rates. Boardman and Poesen (2007), for instance, name snowmelt as an important erosion triggering process in Scandinavia and mountainous regions. On the other hand, rainfall-driven erosion occurs principally in arable land in Northern and Central Europe and is especially important in the erosion-prone loess belt. Whereas in the Mediterranean, high-intensity storms lead to extreme erosion events.

The results of Table SI3 directly show the variations between the environmental conditions in the *EZ*s and the typical management practices. There are broad differences in the climatic conditions, growing seasons, agricultural practices and crops and crop rotations within the *EZ*s. The rainfall distribution within the year (Mediterranean: winter rain, Central and Northern Europe: summer rain) and the magnitude of heavy rainfalls show marked contrasts between the *EZ*s (Panagos et al., 2015a).

It is also worth noting that the values within the *EZ*s show an extensive range. The order of the median and mean values in Table SI3 indicates the right-skewed distribution of *soil erosion*, *soil retention* and *provision capacity* within the *EZ*s. These distributions are related to large areas with low values in opposition to small areas with high loss rates and their control, which is also evident in the analysed pressure and condition indicators. The hotspots within the *EZ*s can only be indirectly addressed in this Pan-European study and must be targeted in more detailed regional studies. Our results highlight that soil erosion prevention measures must be developed in a tiered approach. That necessitates overall policy targets and guidelines on European and Member State level, regional adapted soil conservation frameworks and locally implemented practices.

### 4.3. Main limitations of the indicators

#### 4.3.1. Pressure indicators

Our results show a small change in the extent of agroecosystems, which was expected due to the comparably short assessment period (6 years). Additionally, we only considered the changes from or to different land cover classes without accounting for the “internal” conversions within the agricultural areas. These kinds of changes reflect the ways society, industry and agriculture respond to economic and social conditions and therefore present large differences between regions (European Environment Agency, 2006).

For the assessment of the climatic variables, we used E-OBS data from Copernicus Climate Change Service (Copernicus, 2019). It is important to note that these data were not corrected to improve their homogeneity and that the number of stations varied over time. Another aspect is the discrepancies between maximum, minimum and mean temperatures that were gridded independently (Cornes et al., 2018). As a result, caution is required when interpreting the maps and outcomes of these indicators. Additional data sources are needed to assess trends and identify the impacts of climate change on the condition of agroecosystems. Lengthier growing seasons and more suitable crop conditions (higher temperatures and milder winters) would have some positive effects in Northern Europe (Ciscar et al., 2011), whereas the number of extreme events negatively affecting agriculture is projected to increase (Maes et al., 2020b).

There are significant differences when assessing soil moisture at local and continental scales. At a local scale, changes in soil moisture relate to changes in land cover altering the regional hydrological cycle. At a continental scale, soil water content varies in space and time due to the variability of precipitation and temperature in short and long periods associated with large-scale atmospheric circulation (Kurnik et al., 2015). These differences are relevant when designing and implementing measures to improve agroecosystems and their services at different spatial and temporal scales.

There are multiple ways to assess land use intensity, for example, by quantifying the nitrogen applications (Temme and Verburg, 2011) or by analysing mowing and grazing rates (Felipe-Lucia et al., 2020). We calculated this indicator based on the results from Pérez-Soba et al. (2015), who considered various sources of energy inputs, such as seed development, delivery and planting, soil preparation, pest control and harvest. The selection of different input data influences the outcome of the assessment greatly. Additionally, various sources of input have significant spatial differences. For example, in Central Europe, energy for cultivation and fertilizers are the largest sources, whereas, in Mediterranean zones, irrigation plays a significant role. These differences need to be taken into consideration when assessing and managing land use intensity and its effect on agroecosystems condition.

Similar to land use intensity, there are many ways to calculate soil erosion by water in the EU. One is the Pan European Soil Erosion Risk Assessment (PESERA) model (Kirkby et al., 2008), which combines the effect of topography, climate, and soil into an integrated forecast of runoff and soil erosion (Kirkby et al., 2004). Another approach uses data from the European Environment Information and Observation Network for soil (EIONET – SOIL) and applies the Universal Soil Loss Equation (USLE) or the revised version (RUSLE) model. In this study, we used the data from Panagos et al. (2015e) who implements a modified version of the RUSLE model (RUSLE2015). Some significant differences in the methods and outcomes of the mentioned approaches to calculating soil erosion relate to the mapping procedures, the influence of slopes and vegetation, the input data, and the scale (Panagos et al., 2014b).

Once again, the selection of input data has a considerable effect on the results of the assessment. Furthermore, apart from the well-known limitations of the USLE related to the sources of erosion and the interactions between variables e.g., underestimated impacts of thalwegs and gully erosion (Boardman and Poesen, 2007), and the neglected seasonality of erodibility and its interactions with climate (Auerswald et al., 2014), there are other sources of uncertainty. One is associated with the impossibility to identify regional features in a continental-scale study. The other source of uncertainty is the assessment of soil erosion by water without accounting for wind or tillage erosion. Wind erosion is a common problem in northern Germany, The Netherlands, the Iberian Peninsula, France, Denmark and parts of England (European Commission, 2017). And tillage erosion is a relevant erosion process in agricultural areas (Van Oost et al., 2006). Additionally, Panagos et al. (2015e) calculated soil erosion rates per grid cell without accounting for the amount of soil transferred or received from one pixel to another. These aspects indicate that the actual soil erosion is higher than what we presented here.

We observed the highest SOC losses in mountainous areas in the Mediterranean zones and the Alps, especially in Spain and Italy, in which the rates of soil erosion are high. However, according to Lugato et al. (2016), when looking at the SOC losses in the land cover types within agricultural areas, there are some differences that we did not consider in this study. Orchards and grasslands had high losses compared with croplands, probably because orchards are present in hilly areas. On the other hand, SOC stocks are higher in the superficial soil layer of grasslands than in croplands, and this layer sustains SOC losses even when erosion is low. It is worth noting that Lugato et al. (2016) estimated SOC losses per grid cell similarly to Panagos et al. (2015e) for soil erosion. Thus, these results do not take into account the C received or transferred from one pixel to another. Besides, the estimation only included agricultural areas without taking potential C input from other land uses into account.

We calculated the management indicators related to tillage and soil cover in winter based on official statistical data from EUROSTAT at the Nomenclature of Territorial Units for Statistics 2 (NUTS2) level. We downscaled and disaggregated the data to a 1 km grid level to compare them against other indicators so that some generalizations might not be accurate. Our results highlight the need for harmonized data with higher resolution and with standards that ensure comparability and that can be validated at the European scale.

#### 4.3.2. Ecosystem condition indicators

Due to data availability, we estimated the crop diversity as the number of crops in a 5 arc minutes raster cell, which differs from the units proposed by Maes et al. (2018) of the number of crops per 10 km cell. However, our results provide a general picture of the regions with the highest number of crops. A limitation of this indicator is that it does not describe the types of crops present in a specific area but only the number. This information is relevant to understand the vulnerability of agricultural soils to erosion because some crop types are more sensitive than others.

To calculate the share of semi-natural vegetation, we used a dataset developed by Rega et al. (2018). However, these data do not contain smaller semi-natural features occurring in agricultural landscapes. To overcome this limitation, we used another dataset to calculate the density of hedgerows. Nevertheless, our results show that these two indicators are not closely related, presumably due to differences in the methodologies to calculate them and the data formats. These differences highlight the need to develop frameworks to integrate datasets that describe similar aspects and use high-resolution satellite images.

We used official statistical data to calculate the share of fallow land and arable land, permanent crops and livestock density. Consequently, these indicators have the same limitations as the management indicators mentioned in Section 4.3.1. Additionally, due to limited data availability, we could not assess relevant factors such as the duration and the management of the fallow land and the differences between livestock management, which can influence agroecosystem condition in diverse ways.

The content of soil organic carbon is highly variable in agricultural and forest areas. It can be affected by natural factors such as climate, soil parent material, vegetation and topography, but it can also be affected by anthropogenic factors such as land management (European Commission, 2017). We analysed the SOC content in agricultural areas without looking at the differences between the sub-levels (arable land, permanent crops, pastures and heterogeneous agricultural areas) that belong to this classification. This could lead to the omission of SOC content and loss variations that depend on cultivation practices, crop or plant cover and drainage status. Besides, we looked at the average content for one year but did not consider the short or long-term changes that occur when there are conversions in land cover.

Similar to SOC, bulk density is highly variable since it depends on the soil type and the land cover. For this reason, bulk density has the same limitations as SOC, as we did not consider the sub-levels within the land cover agricultural areas. Additionally, the model had a low performance when predicting the values in mountainous and hilly areas, possibly related to the high diversity in terrains, land covers and substrates in such regions (Ballabio et al., 2016).

#### 4.3.3. Ecosystem service indicators

There are various uncertainties associated with the calculation of the ecosystem service *control of erosion rates*. Some of these uncertainties derive from the modelling of land use and land cover changes based on the Land Use-based Integrated Sustainability Assessment (LUISA) modelling platform (Commission, 2016). This platform requires spatially explicit and statistical data, which are not always available (Maes et al., 2015). Another source of uncertainty is the USLE/RUSLE model. As it has many factors, which individually bring uncertainty to the outputs. The rainfall erosivity factor, for instance, does not have the required temporal and spatial resolution to represent the impact of heavy rainfall. On the other hand, the crop and management factor, modelled from the LUISA and other modelled spatial data, could increase the degree of uncertainty.

A more dynamic modelling framework developed at a continental scale would contribute to overcoming these uncertainties. This model would improve the annual soil erosion estimations by incorporating high temporal and spatial resolution data on vegetation change, detailed databases on crop types, soil characteristics and soil loss information, and numerical models to regularly estimate rainfall erosivity (Panagos et al., 2020).

### 4.4. Potential for policy implementation

We used a methodological approach to test the framework and indicators proposed by MAES to assess agroecosystem condition at the EU level. However, we only selected indicators with direct implications on the ecosystem service *control of erosion rates*. This selection could seem biased and probably does not show a complete picture of the agroecosystem condition in the entire EU. Nonetheless, these indicators can be used to assess changes over time, depending on data availability. This is because most of the estimation of pressures, condition and soil erosion regulation are based on models that could also be used in the development of future scenarios related to land use and climate changes.

Showing the link between good ecosystem condition and a higher provision of ecosystem services supports site-specific sustainable land management and conservation and restoration efforts. We found correlations within the *EZ*s, which indicates that agroecosystems in better condition have a higher capacity to prevent soil erosion. It is worth noting that the assessment of other ecosystem types and services would, of course, require different or additional indicators and would certainly provide different outcomes. Studies of bundles of ecosystem services enable assessments of trade-offs and synergies between diverse land use options and related ecosystem services. Such studies deliver highly relevant information for improved management of multifunctional landscapes.

The heterogeneous territories and agricultural practices in the EU pose a significant challenge to the common implementation of environmental and agricultural policies (Recanati et al., 2019). We identified significant differences between the *EZ*s that could support the definition of targets and possible measures to improve agroecosystem condition. For example, in *EZ*s with high pressures, limiting conditions and low provision capacity, such as the Mediterranean and *Pannonian*, the focus should be on monitoring precipitation and implementing the mitigating soil cover and tillage practices. In Mountainous areas with high soil erosion rates and SOC loss, targets should focus on the diversification of crops, the maintenance of permanent grassland and the covered soils during winter. However, more detailed transdisciplinary assessments on different spatial and temporal scales are required to inform policy and decision-makers about interactions between diverse sectors such as economy, public health and the environment. Such evidence-based decision making will help improve agroecosystem condition and the delivery of multiple ecosystem services in the long term. It would also promote new market opportunities and improve the needs of consumers and farmers.

## 5. Conclusions

Analysing the relationships between agroecosystem condition and ecosystem services such as *control of erosion rates* contributes to understanding the importance of site-specific sustainable agriculture in supporting good environmental conditions and human well-being. It also demonstrates how environmental and anthropogenic pressures can affect the capacity of ecosystems to provide multiple ecosystem services. However, our results on European and regional levels highlight that these effects vary depending on regional characteristics such as climate, landscape structure and cropping patterns.

To our knowledge, this is the first integrated mapping and assessment of agroecosystems and their capacity to control soil erosion at the European level. Our results emphasise that patterns in the complex interactions between this ecosystem service and ecosystem condition indicators should be analysed at a sub-European scale to address variations in landscapes, climate and therefore also erosion processes and rates. On the level of *EZ*s, we found that the control of erosion rates is correlated positively with multiple condition indicators and negatively with pressure indicators. Our results also help identify *EZ*s where actions should be taken to mitigate the environmental and anthropogenic pressures on agroecosystems and improve their condition. Although our results are limited to one exemplarily chosen ecosystem service, they indicate that a good ecosystem condition is necessary for the capacity of ecosystems to provide services.

Transdisciplinary research on additional ecosystem services provided by agroecosystems, human health and economic aspects is needed for awareness-raising and evidence-based sustainable decision making. More comprehensive assessments would support management practices and policies beneficial for farmers, consumers and the environment on various spatial and temporal scales.

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## Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2021.101387.

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