Relationships between ecosystem condition and ecosystem services at different spatial scales

Von der Naturwissenschaftlichen Fakultät der Gottfried Wilhelm Leibniz
Universität Hannover

zur Erlangung des Grades
Doktorin der Naturwissenschaften (Dr. rer. nat.)

genehmigte Dissertation
von
Paula Mercedes Rendon Cardona, M. Sc.

2022

Referent:	Prof. Dr rer. nat. Benjamin Burkhard Leibniz Universität Hannover
Korreferentin:	Prof. Dr rer. nat. Emily Poppenborg Martin Leibniz Universität Hannover
Korreferent:	Associate Prof. Dr Rudolf de Groot Wageningen University
Tag der Promotion	05.11.2021

Summary

Land degradation is one of the main drivers of decline in biodiversity, ecosystem condition and services. Agricultural expansion is the main form of land-use change and causes adverse effects on forests, wetlands, grasslands and their capacity to provide ecosystem services. Some of the most significant impacts of land degradation related to agricultural expansion occur in the soil. It causes the deterioration of soils' physical, chemical and biological characteristics and reduces the vegetation cover in the long term. Soil erosion is a direct consequence of land degradation, and it reduces nutrients and agricultural productivity and causes sedimentation and flooding in areas where the washed-out soil accumulates.

Therefore, monitoring and measuring ecosystem condition are relevant to recognise early signs of land degradation. At the same time, the analysis of the relationship between ecosystem condition and the provision of ecosystem services helps identify priority areas where specific parameters need improvement to guarantee the sustainable provision of ecosystem services. Thus, the objective of this thesis is to improve the knowledge about ecosystem condition assessment and the relationship with ecosystem services. The aims are: (1) to identify the indicators used in Europe to assess ecosystem condition; (2) to test the indicators proposed by MAES to assess ecosystem condition; and (3) to analyse the relationships between ecosystem condition and services at a regional and continental scale using as an example the ecosystem service *control of erosion rates*.

First, the thesis provides some background information about ecosystem condition and services, emphasising on agroecosystems and *control of erosion rates*. Afterwards, a description of the methods used for the assessment is presented, followed by the research questions that guided the subsequent chapters.

Second, chapter 2 presents the literature review results on ecosystem condition mapping and assessment in Europe. The aim is to provide an overview of the state of research and highlight some limitations when evaluating ecosystem condition and services. The results show some gaps in ecosystem condition mapping, principally related to the methods used and the coverage of ecosystems. The outcomes indicate the need to explore the relationships between ecosystem condition and the capacity of ecosystems to provide services. The findings also highlight the need to improve the applicability of condition indicators in policy and decision making.

Third, based on the research gaps identified in the literature review, chapters 3 and 4 test the MAES indicators and analyse the relationship between agroecosystem condition and the ecosystem service *control of erosion rates* in Northern Germany and Europe, respectively. The studies follow an operational framework for the integrated mapping and assessment of ecosystems and their services. The results show an uneven distribution of the condition indicators across the study and positive, negative, and no significant correlations between the different pressures and conditions and the *control of erosion rates* with considerable regional differences.

This thesis concludes with a discussion of the main findings and the conclusions based on the most relevant results. Finally, the report ends with some practical recommendations and an outlook on future research.

Keywords: ecosystem condition, ecosystem status, ecosystem services, mapping, assessment, control of erosion rates, soil erosion.

Zusammenfassung

Die Verschlechterung der Bodenqualität ist eine der Hauptursachen für den Rückgang der biologischen Vielfalt, des Zustands der Ökosysteme und ihrer Leistungen. Die Ausdehnung der Landwirtschaft ist die wichtigste Form der Landnutzungsänderung und hat nachteilige Auswirkungen auf Wälder, Feuchtgebiete und Grasland sowie deren Fähigkeit, Ökosystemdienstleistungen zu erbringen. Einige der wichtigsten Auswirkungen der Bodendegradation im Zusammenhang mit der Ausweitung der Landwirtschaft betreffen den Boden. Sie führt zu einer Verschlechterung der physikalischen, chemischen und biologischen Eigenschaften der Böden und verringert langfristig die Vegetationsdecke. Die Bodenerosion ist eine unmittelbare Folge der Bodendegradation und führt zu einem Rückgang der Nährstoffe und der landwirtschaftlichen Produktivität sowie zu Sedimentation und Überschwemmungen in Gebieten, in denen sich der ausgewaschene Boden ansammelt.

Daher ist die Überwachung und Messung des Zustands der Ökosysteme wichtig, um Anzeichen für eine Verschlechterung der Bodenqualität frühzeitig zu erkennen. Gleichzeitig hilft die Analyse des Verhältnisses zwischen dem Zustand der Ökosysteme und der Bereitstellung von Ökosystemdienstleistungen dabei, vorrangige Gebiete zu ermitteln, in denen bestimmte Parameter verbessert werden müssen, um die nachhaltige Bereitstellung von Ökosystemdienstleistungen zu gewährleisten. Ziel dieser Arbeit ist es daher, das Wissen über die Bewertung des Zustands von Ökosystemen und den Zusammenhang mit Ökosystemdienstleistungen zu verbessern. Die Ziele sind: (1) die Indikatoren zu identifizieren, die in Europa zur Bewertung des Zustands von Ökosystemen verwendet werden; (2) die von MAES vorgeschlagenen Indikatoren zur Bewertung des Zustands von Ökosystemen zu testen; und (3) die Beziehungen zwischen dem Zustand von Ökosystemen und ihren Leistungen auf regionaler und kontinentaler Ebene zu analysieren, wobei als Beispiel die Kontrolle der Erosionsraten als Ökosystemdienstleistung dient.

Zunächst werden einige Hintergrundinformationen über den Zustand und die Leistungen von Ökosystemen gegeben, wobei der Schwerpunkt auf Agrarökosystemen und der Kontrolle von Erosionsraten liegt. Danach wird eine Beschreibung der für die Bewertung verwendeten Methoden gegeben, gefolgt von den Forschungsfragen, die den nachfolgenden Kapiteln zugrunde liegen.

Zweitens werden in Kapitel 2 die Ergebnisse der Literaturrecherche zur Kartierung und Bewertung des Zustands von Ökosystemen in Europa vorgestellt. Ziel ist es, einen Überblick über den Stand der Forschung zu geben und einige Einschränkungen bei der Bewertung des Zustands und der Leistungen von Ökosystemen aufzuzeigen. Die Ergebnisse zeigen einige Lücken in der Kartierung des Ökosystemzustands auf, vor allem in Bezug auf die verwendeten Methoden und die Abdeckung der Ökosysteme. Die Ergebnisse weisen auf die Notwendigkeit hin, die Beziehungen zwischen dem Zustand der Ökosysteme und ihrer Fähigkeit, Leistungen zu erbringen, zu untersuchen. Die Ergebnisse verdeutlichen auch die Notwendigkeit, die Anwendbarkeit von Zustandsindikatoren in Politik und Entscheidungsfindung zu verbessern.

Drittens werden auf der Grundlage der in der Literaturübersicht ermittelten Forschungslücken in den Kapiteln 3 und 4 die MAES-Indikatoren getestet und die Beziehung zwischen dem Zustand der Agrarökosysteme und der Kontrolle der Erosionsraten durch die Ökosystemdienstleistungen in Norddeutschland bzw. Europa analysiert. Die Studien folgen einem operativen Rahmen für die integrierte Kartierung und Bewertung von Ökosystemen und ihren Leistungen. Die Ergebnisse zeigen eine ungleichmäßige Verteilung der Zustandsindikatoren innerhalb der Studie sowie positive, negative und keine signifikanten Korrelationen zwischen den verschiedenen Belastungen und Bedingungen und der Kontrolle von Erosionsraten mit erheblichen regionalen Unterschieden.

Diese Arbeit schließt mit einer Diskussion der wichtigsten Erkenntnisse und den Schlussfolgerungen, die auf den wichtigsten Ergebnissen basieren. Der Bericht endet mit einigen praktischen Empfehlungen und einem Ausblick auf künftige Forschungsarbeiten.

Schlagworte: Zustand des Ökosystems, Ökosystemdienstleistungen, Kartierung, Bewertung, Kontrolle der Erosionsraten, Bodenerosion.

Contents

List of figures

List of abbreviations

AIC	Akaike Information Criterion
BD	Birds Directive
BISE	Biodiversity Information System for Europe
BKG	Bundesamt für Kartographie und Geodäsie (Federal Agency for Cartography and Geodesy)
CICES	Common International Classification of Ecosystem Services
CLC	CORINE Land Cover
CORINE	Coordination of Information on the Environment
DPSIR	Drivers, Pressures, State, Impact and Responses
DWD	Deutscher Wetterdienst
EAP	Environmental Action Programme
ECMWF	European Centre for Medium-Range Weather Forecasts
EEA	European Environment Agency
EQR	Ecological Quality Ratio
ESDAC	European Soil Data Centre
EU	European Union
EUROSTAT	European Statistical Office
FAO	Food and Agriculture Organization of the United Nations
GHG	Greenhouse gas
GIS	Geographic Information Systems
HD	Habitats Directive
I DM	De Martonne Aridity Index
IPBES	Intergovernmental Panel on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
JRC	Joint Research Centre
LU	Livestock Units
MA	Millennium Ecosystem Assessment
MAES	Mapping and Assessment of Ecosystems and their Services
MMU	Minimum Mapping Unit
MSFD	Marine Strategy Framework Directive
NEA	National Ecosystem Assessment

REDD+	Reducing emissions from deforestation and forest degradation in developing countries, and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks
SDG	Sustainable Development Goals
SEEA -EA	System of Environmental Economic Accounting – Ecosystem Accounting
SOC	Soil Organic Carbon
TEEB	The Economics of Ecosystems and Biodiversity
UAA	Utilized Agricultural Area
UN	United Nations
USLE	Universal Soil Loss Equation
WFD	Water Framework Directive

1

Introduction

1. Introduction

1.1. Motivation and objectives

Land degradation is one of the main drivers of declining biodiversity, ecosystems condition and their capacity to provide services. Land use change, unsustainable land management, pests and diseases, among others, induce land degradation (Mirzabaev et al., 2016). One of the principal forms of land use change is agricultural expansion, linked to the growing human population and consumption. This expansion has significant impacts on forests, wetlands and grasslands and the services they provide. At the same time, the degradation of agricultural lands reduces the biodiversity of cultivated and wild crops and domesticated breeds, which suggests that agroecosystems are less resilient to the impacts of climate change, pests and diseases (Lin, 2011).

Other relevant impacts of land degradation include soil degradation, deterioration of the soil biological, physical and chemical properties, and the long-term reduction of vegetation (Le et al., 2016). Soil degradation involves the decline in soil quality which reduces its functions and services (Lal, 2015). Soil erosion is a predominant consequence of soil degradation. It causes nutrients and agricultural productivity loss, flooding, sedimentation and pollution in areas where the washed-out soil accumulates (Lal, 2014; Rickson, 2014). Monitoring and measuring ecosystem condition are therefore relevant to identify early signs of land degradation, and to take the actions needed to reduce or mitigate its impacts. Furthermore, the analysis of the relationships between ecosystem condition and services helps identify priority areas where the improvement of condition parameters is needed to guarantee the sustainable provision of ecosystem services.

The overall objective of this study is to improve the knowledge on the assessment of ecosystem condition and the relationships with ecosystems services. The aims are, first, to identify the methods and indicators used to assess ecosystem condition in Europe. Second, test the framework and indicators proposed by the Mapping and Assessment of Ecosystems and their Services (MAES) group of the European Commission to assess agroecosystem condition (Maes et al., 2018). Third, analyse the relationships between the condition indicators and one exemplary selected ecosystem service (*control of erosion rates*), both at a regional and continental scale.

This study follows an operational framework for integrated mapping and assessment of ecosystems and their services (Burkhard et al., 2018). This framework consists of a series of steps to guide the ecosystem assessment and includes: mapping of ecosystems, the definition of the condition of ecosystems, quantification and mapping of services and condition indicators, and integration of results. The aim is to improve the methodology and applicability of the MAES framework and indicators for the assessment of agroecosystem condition at different spatial scales.

The introductory part provides background knowledge about ecosystem condition, the terms to describe it, and the different frameworks and perspectives to assess it. Afterwards, the condition of agroecosystems and the dual role of agriculture related to land degradation is discussed. Later, the ecosystem services provided from agroecosystems are mentioned with an emphasis on *control of erosion rates*. Subsequently, a description of the methods used for the assessment is presented, followed by the research questions that guided the next chapters.

1.2. Ecosystem condition

The extent and condition of ecosystems, the composition of communities, and species populations have declined over the last decades (IPBES, 2019a). The acceleration of social and economic activity that posed enormous pressures on the environment is one of the principal causes of this decline. These pressures are expected to increase even more due to the growth of the global human population by almost a third by 2050, the doubled resource use by 2060, and the increase in water and energy demand (European Environment Agency, 2019a). Policies have been proposed and implemented in the European Union (EU) to protect the environment and safeguard the health and well-being of its citizens. These efforts, however, have been insufficient. A possible cause for this is the complexity of the environmental systems that suggest a time lag between the reduction of pressures and the improvement of the condition of the environment (*Ibid.*). Therefore, a better understanding of these complex systems and the interlinkages with social systems would improve the definition of goals and the actions needed to achieve them.

Ecosystem condition has been a subject of increasing interest, together with ecosystem services, since the release of the Millennium Ecosystem Assessment(MA) (MA Board, 2005). The links between ecosystem condition and human well-being are one of the main aspects of the MA approach (DeFries and Pagiola, 2005). Since then, many definitions have been proposed for this concept. Condition is considered as *"a scientific description of the ecological state of an environmental asset, measured through indicators that describe an asset's vigour (level of productivity), organisation (structure and interactions) and resilience (ability to rebound from a* *shock)"* (Wentworth Group of Concerned Scientists, 2016, p. 27). The SEEA EA (System of Environmental Economic Accounting – Ecosystem Accounting) defines ecosystem condition as *the "overall quality of an ecosystem in terms of its biotic and abiotic characteristics"* (United Nations, 2021, p. 335). Another definition includes the quality measures and the biophysical state measures that reflect the functioning and integrity of the ecosystem (Bordt, 2015). The MAES working group has defined ecosystem condition as the *"physical, chemical, and biological condition or quality of an ecosystem at a particular point in time"* (Maes et al., 2018, p. 5). Additionally, it has been used as a synonym of *state* to avoid confusion with the term *status* that describes legal aspects such as the ecosystems' protection under different environmental directives (Erhard et al., 2016). (Chapter 2 describes other synonyms and descriptors of ecosystem condition).

According to Keith et al. (2020), there are some divergences in the definition of ecosystem condition derived from the different perspectives about the purpose of assessing condition. One purpose can be the analysis of the intrinsic values of the ecosystem, while the other can be the representation of the instrumental values of ecosystems. In the first purpose, condition implies the integrity of the ecosystem and its status compared against a reference condition. Whereas in the second purpose, condition is the capacity to supply ecosystem services (Figure 1.). This research used the definition from Czúcz and Condé (2017, p. 13) that combines the SEEA-EA and OpenNESS (Potschin-Young et al., 2016) definitions. The authors propose that ecosystem condition is *"the overall quality of an ecosystem unit, in terms of its main characteristics underpinning its capacity to generate ecosystem services"*. Based on the framework proposed by Keith et al. (2020), this definition fits in the bottom-right quadrant of Figure 1 (instrumental and anthropocentric).

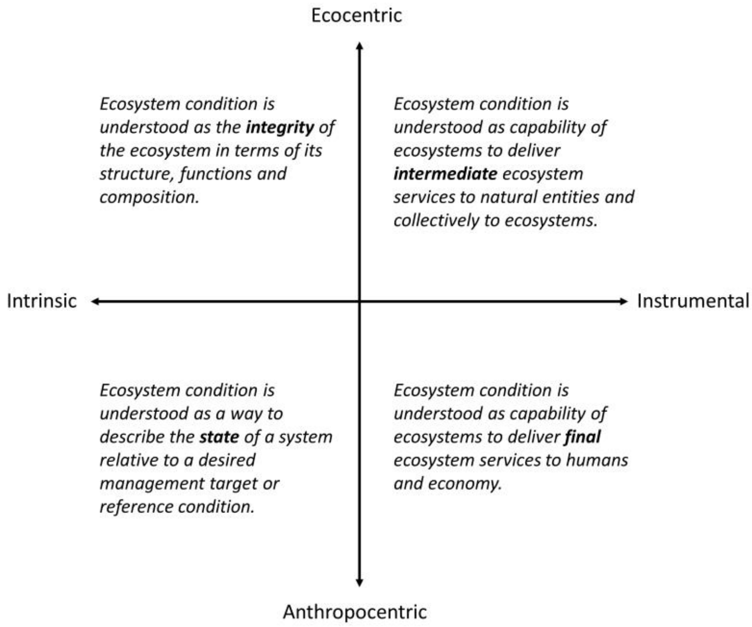

Figure 1. Framework representing the range from intrinsic to instrumental values and from ecocentric to anthropocentric world views (Keith et al., 2020).

When assessing ecosystem condition, it is relevant to take into consideration the reference condition and reference levels. Reference condition is the state of an ecosystem used for setting reference levels. It is a condition against which the past, present or future condition can be evaluated (Maes et al., 2020). Reference condition is used to assess the impacts of human activity on ecosystems. However, its definition is highly dependent on the type and purpose of the assessment. Therefore, all reference conditions must be stated explicitly in relation to them by using concepts such as minimally-disturbed condition, historic condition, least-disturbed condition, best-attainable condition (Stoddard et al., 2006). These concepts are predominantly applicable to systems that have been affected by human disturbances. For undisturbed ecosystems, the reference condition is its ecological integrity or naturalness.

Reference levels are the values of an ecosystem condition variable used to rescale the variable to an indicator. They are decisive to derive appropriate ecosystem condition indicators or indexes (Keith et al., 2020). Concepts such as target and threshold levels are slightly similar to reference levels. However, target levels are desired values for planning and policy that take into consideration societal values (Levin et al., 2015). Whereas threshold levels are estimated values at which point the functioning of an ecosystem changes irreversibly, often with undesirable consequences (Vergano and Nunes, 2007). These values should not be used as reference values but as ancillary information to assess the risks of change or the degree of resilience of an ecosystem.

The European Environment Agency (EEA) monitors and assesses the condition of ecosystems in the EU and provides the information necessary to establish policy objectives and targets. In the latest report, the EEA summarises the past trends, outlooks and prospects of meeting the policy objectives and targets in 2020, 2030 and 2050 (European Environment Agency, 2019a) The results are discouraging, especially concerning the protection, conservation and enhancement of natural capital. Aspects such as common species (birds and butterflies), urbanisation and land use by agriculture and forestry, soil condition and climate change, and impacts on ecosystems have had deteriorating trends in the past, and the prospects of meeting the selected policy targets are not on track. Biodiversity loss, resource extraction and hazardous emissions continue due to agriculture, fisheries, transport, industry and energy production (*Ibid.*).

1.3. Condition of agroecosystems

Agricultural production has more than tripled since the 1960s. This expansion is partly due to the Green Revolution that prompted high crop research and infrastructure investments (Pingali, 2012). This increase in agricultural production also relates to the expansion in land, water and natural resources use (FAO, 2017). Livestock production is one of the sectors of agriculture with the fastest growth, principally in developing countries. The consumption of milk, dairy products and meat per capita has almost doubled since the 1970s (Alexandratos and Bruinsma, 2012). However, this increase came with adverse effects on the natural resources needed for agricultural development. These effects include land degradation, salinisation of soils, over-extraction of groundwater, soil erosion and reduction in biodiversity (FAO, 2017). Other negative impacts are evident in the broader environment through deforestation, water pollution and greenhouse gas (GHG) emissions, which are discussed in more detail in the next section.

Drivers affecting agroecosystems and impacts of agriculture

Various key drivers affect the capacity of agricultural areas to provide food, feed and other ecosystem services necessary for human well-being. These drivers include climate change, land use change, soil degradation, pollution, and invasive alien species. At the same time, agriculture is one of the economic activities with the highest impacts on ecosystems and society. Mechanized harvesting and tillage, land clearing and the use of pesticides and fertilizers impact soil, water and air quality and biodiversity. This section describes the main drivers affecting agroecosystems and the impacts of agriculture.

Climate change

The Intergovernmental Panel on Climate Change (IPCC) predicts that by 2050 agricultural production will decrease by 2% every decade due to climate change, while demand will increase by 14% (IPCC, 2014). Climate change will exacerbate soil erosion rates and sediment transport, causing negative impacts on the agroecosystems (Bussi et al., 2016). Changes in the amount, intensity and spatiotemporal distribution of rainfall will directly impact soil erosion (Li and Fang, 2016). On the other hand, increased temperatures will have indirect impacts due to the reduced soil moisture and vegetation cover (*Ibid.)*.

The expansion and intensification of agriculture are also major contributors to climate change. Around 25% to 35% of the global GHG emissions that cause climate change come from land clearing, crop production and fertilization (Foley et al., 2011). Land clearing is associated with increased GHG emissions as the crops that replaced trees hold less carbon per unit area than forests (Longobardi et al., 2016). Methane, nitrous oxide, nitric oxides and ammonia emissions come principally from ruminants, rice cultivation, biomass burning, livestock and application of animal manure (Burney et al., 2010; Power, 2010).

Land-use change

The expansion of agriculture is the most widespread form of land use-change worldwide. One-third of the globe's surface is used for cultivating crops and animal husbandry (FAO and ITPS, 2015). Agricultural expansion, urbanisation and increased infrastructure had come at the cost of losing forests, wetlands and grasslands (IPBES, 2019b). In addition, land use change is associated with air, water and soil pollution and regulating and cultural ecosystem services decrease.

Apart from the increase in the extension of agricultural areas, agricultural intensification has risen in the last decades with mainly adverse effects on the environment (Rasmussen et al., 2018). Although the intensification of agriculture increases food and timber production, it also increases the consumption of fertilizers and pesticides and expands the area under irrigation (Felipe-Lucia et al., 2020; Schiefer et al., 2016). These negative aspects will further impact the fertility of soils, the quality of surface and groundwater and will aggravate the loss of biodiversity and the simplification of ecosystems (Schiefer et al., 2016).

Soil degradation

Soil degradation has negative impacts on agriculture and, at the same time, is exacerbated by inadequate agricultural practices. According to the IPBES, land degradation has reduced global agricultural productivity by 23% and affects 3.2 million people (IPBES, 2018). Soil degradation is caused by poor management practices that lead to increases in soil erosion, compaction, contamination and sealing and a decrease in soil organic carbon and biodiversity (Lal, 2015).

Degradation caused by soil erosion reduces crop productivity due to the decline in soil resources and nutrients (Lal, 2014; Panagos et al., 2018). Soil erosion also reduces crop production by decreasing topsoil thickness and organic matter content. It also induces soil compaction and crusting and degrades the physical and chemical properties of soils (Blanco-Canqui and Lal, 2010). Other negative impacts of erosion are the modification of the carbon, nitrogen and phosphorus biochemical cycles (Quinton et al., 2010) and off-site effects such as the pollution of water, the sedimentation of reservoirs and floods (Rickson, 2014).

Soil degradation also involves the contamination, acidification and salinisation of soils. The use of fertilizers and pesticides in addition to atmospheric depositions increase the contamination and concentrations of heavy metals in soils (Quinton and Catt, 2007). Acidification is associated with the atmospheric deposition of acid substances resulting from sulphur dioxide and nitrogen oxides emissions (Greaver et al., 2012). Salinisation, on the other hand, is caused by inadequate irrigation practices (including brackish water use), poor drainage and the intrusion of seawater in coastal areas (FAO and ITPS, 2015).

Water withdrawal and pollution

Around 70 – 90% of the water withdrawals from rivers, lakes and aquifers, come from agricultural activities. Principally for the production of crops and livestock (Foley et al., 2005). Additionally, the use of pesticides affects non-target ecosystems such as surface waters, threatening freshwater integrity. The use of pesticides is a significant driver for biodiversity loss in aquatic ecosystems impacted by agriculture. It causes marked reductions in macroinvertebrate richness in rivers (Stehle and Schulz, 2015) and acute and long-term effects on fish, invertebrates and algae (Malaj et al., 2014).

Pollution also occurs in agricultural areas caused by fertilizers and pesticides used in crops. These products affect the soil quality, farmland biodiversity, including pollinators and other beneficial organisms (Dudley et al., 2017). Global consumption of fertilizers is growing worldwide at an annual average rate of 1.4%. While in contrast, the consumption of pesticides has remained relatively stable, with an average of 2.6 kg per ha of cropland globally (FAO, 2021, 2019).

Invasive alien species

Alien species are introduced deliberately or accidentally through their transport and introduction to areas beyond their typical biogeographical barriers (Bellard et al., 2016). These species can become invasive and displace native species, disturb habitats and community structure (Charles and Dukes, 2008). Invasive alien species pose high pressures on the new environment. They cause adverse effects on biodiversity, ecosystems and their services (Barbet-Massin et al., 2020). These pressures on ecosystems will grow in the future due to the disturbance of habitats and the increasing trade between regions with similar climatic and environmental conditions (Seebens et al., 2015).

The economic impact of invasive species is enormous, with estimated losses from invasive insects of almost US$70 billion per year globally (Bradshaw et al., 2016). In agricultural areas, the increase in transboundary pests and diseases has particularly adverse effects on productivity. The Food and Agriculture Organization of the United Nations (FAO) estimates that the global crop losses associated with pests are between 20 and 40% of the production (FAO, 2017). However, the threat from invasive species varies among countries, as some have a higher dependency on agriculture, which makes them more vulnerable (Paini et al., 2016).

1.4. Ecosystem services from agriculture

According to Burkhard et al. (2012), *"ecosystem services are the contributions of ecosystem structure and functions – in combination with other inputs – to human well-being"*. This definition is especially relevant for agroecosystems as they are managed ecosystems that provide and consume multiple services, and they comprise both natural and anthropogenic inputs (e.g., water, fertilizers, energy, technology, labour and knowledge) (Burkhard et al., 2014).

This combination of natural ecosystem conditions and human inputs enable ecosystem services provision from agroecosystems (Power, 2016). However, the capacity of agroecosystems to provide services varies according to climatic, geological, geographical and anthropogenic factors. Climatic conditions such as radiation and temperature, soil characteristics and quality, and crop features are determinant for agroecosystems to generate crop biomass. Additionally, factors such as water and nutrients availability or the presence of pests, and diseases, are generally compensated by anthropogenic inputs or management practices (Bethwell et al., 2021).

The main objective of agroecosystems are food, fibres, timber, market products, fuels, pharmaceutical and energy crops production (Power, 2010). Agroecosystems also provide regulating services such as climate and water regulation (Balmford et al., 2011), and cultural ecosystem services such as landscape aesthetics and knowledge systems (Bernués et al., 2014; Burkhard et al., 2014). At the same time, agroecosystems depend on the supply of other ecosystem services for their functioning. These services include pollination, water, nutrients and climate regulation, soil erosion regulation, pest and disease control, and genetic diversity (Schulte et al., 2014; Zhang et al., 2007). However, the provision of these services is threatened by the pressures mentioned in section 1.3.1. In Europe, especially, soil degradation caused by erosion is a major threat (Panagos et al., 2018).

Soil erosion occurs when water, wind or tillage remove the topsoil from the land surface (FAO and ITPS, 2015). Water erosion occurs due to erosive raindrops that detach the soil particles and runoff that accumulates and removes the soil from narrow channels (Li and Fang, 2016). Wind erosion occurs by the action of strong winds that mobilize fine, loose and dry soil (Borrelli et al., 2016). Tillage erosion, conversely, is the movement of soil downslope by tillage implements (Van Oost et al., 2006). Piping erosion is another erosion type that results from the detachment of soil entrained by subsurface water flow (Poesen, 2018). As mentioned in section 1.3, soil erosion can cause severe on-site and off-site problems damaging properties, affecting livelihoods and services, and inducing economic disruption and ecological deterioration.

Agroecosystems rely on the functioning of soils, their properties and their management to provide many ecosystem services. The physical, chemical and biological characteristics of soils and their management are essential elements for the sustainability of agricultural systems (Paul et al., 2020). *Control of erosion rates*, chosen as an exemplary service in this study, is a regulating ecosystem service that mitigates the potential soil loss occurring in the absence of vegetation, which means that vegetation covering the soil is the principal provider of the service (Guerra et al., 2014). The provision of this service is relevant not only to prevent soil loss but also to prevent the loss of other soil-related ecosystem services (Steinhoff-Knopp et al., 2021). Agroecosystems provide this service through the retention of sediments and soil (Terrado et al., 2014). However, properties related to soil organic carbon, soil structure and water capacity, together with management practices regarding soil cover, crop rotation and tillage, are fundamental (Adhikari and Hartemink, 2016; Stavi et al., 2016; Steinhoff-Knopp and Burkhard, 2018).

Chapters 3 and 4 provide a detailed description of the motivation and framework used to calculate the ecosystem service *control of erosion rates* in agricultural areas in each study area.

1.5. Concept and methods

1.5.1. Conceptual framework

The general framework guiding this research proposed by the MAES working group (Maes et al., 2013) shows the link between ecosystems and socio-economic systems (Figure 2). This link is bidirectional as ecosystems provide services but are also affected by drives of change that alter their state (or condition). On the left side, the framework highlights the role of biodiversity in the functioning of ecosystems that determine their capacity to provide ecosystem services. The argument is that an ecosystem in good condition has a maximum potential for ecosystem functions and services. On the right side, the flow of ecosystem services that benefit people and affect human well-being is essential to manage the socio-economic system. Here the authors include other values for the benefits to people, such as the health, social and conservation values as not all the benefits can be measured in monetary terms. Another key element in the framework is the response from institutions, stakeholders and users of services that affect ecosystems through drivers of change.

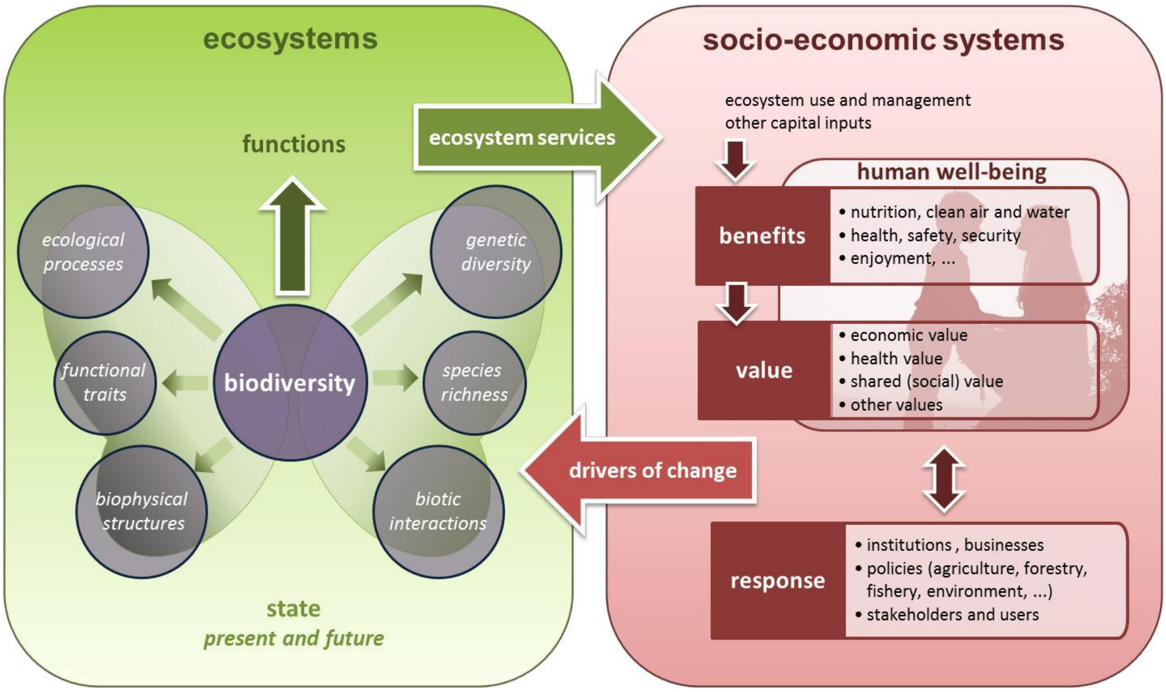

Figure 2. Conceptual framework for EU wide ecosystem assessments (Maes et al., 2013).

The DPSIR (Drivers, Pressures, State, Impact and Responses) model was another guiding framework used in this study (Figure 3). The model was proposed by Smeets and Weterings (1999) and used by the EEA to report their activities. The *Drivers* comprise the socioeconomic and demographic processes that lead to climate change, land use and land cover change. These changes are *Pressures* that affect the state of the ecosystems. They can be direct or indirect, and their strength, persistence and change are determinant for ecosystem condition (*State*). Therefore, it is necessary to analyse the connections between pressures and condition in the various ecosystem types and at different spatial units (Erhard et al., 2017) (see chapters 3 and 4). Additionally, if there is enough available data to analyse trends in pressures, they provide valuable information about the expected changes in the future. This information is significant for decision-making as *Response* to reduce *Impacts* and mitigate and adapt to the effects.

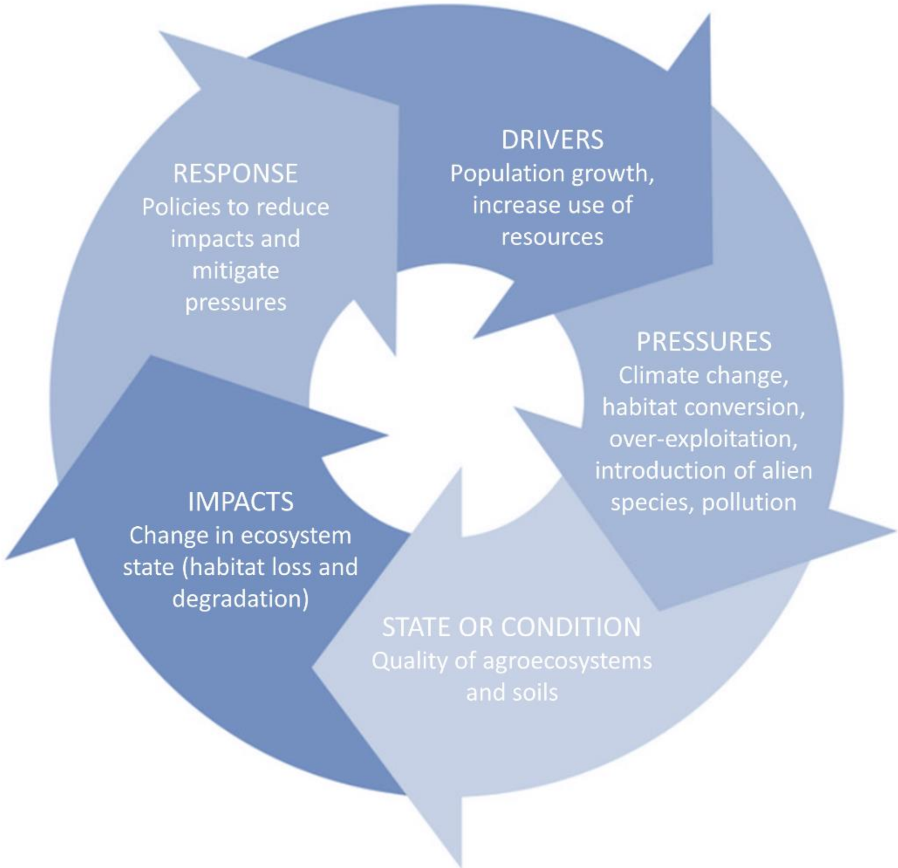

Figure 3. DPSIR framework for the assessment of agroecosystem condition.

The conceptual frameworks presented in this section are addressed in the different chapters of this study. Chapter 2, for instance, demonstrates that information on ecosystem condition is often insufficient for proper mapping and assessment. The chapter also highlights knowledge gaps regarding the relationships between pressures, ecosystem condition and services. Chapters 3 and 4, on the other hand, test the applicability of pressure and condition indicators in agroecosystems at different spatial scales. Figure 3 in chapter 3 shows a synthesis of the relationships between pressures, condition, *control of erosion rates,* and the policy objectives (understood as responses in the DPSIR model) in agroecosystems based on Maes et al. (2018). Additionally, chapters 3 and 4 follow a methodological approach to assess the relationships between these indicators and provide recommendations for policy implementation.

1.5.2. Quantification and mapping of indicators

Indicators are quantitative metrics that reflect a specific phenomenon (Potschin-Young et al., 2016). They are helpful to the set goals and to evaluate their fulfilment within an established period. Most of the indicators used in this study come from the work developed by the MAES working group and their different ecosystem pilots. In their 5th report (Maes et al., 2018), MAES proposed a set of indicators to assess pressures and condition per ecosystem type. They include urban ecosystems, forests and woodlands, wetlands, rivers and lakes, crops and grasslands, heathland, shrub and sparsely vegetated land, marine inlets and transitional waters, coastal, shelf and open ocean. As this research focuses on agroecosystems, various indicators proposed for croplands and grasslands were used. These indicators provide information about pressures such as habitat conversion, climate change, overexploitation, invasive species, pollution, and information on the biotic and abiotic quality of the ecosystems.

Other indicators included in the assessment are associated with agricultural land management. They inform principally about tillage and soil cover practices and are not part of the MAES framework. However, these indicators are quantified and mapped in the study due to their impacts on vegetation cover, soil quality and the ecosystem service *control of erosion rates*. Additionally, indicators related to this service are a significant part of the study as the aim was to compare them with pressures and condition indicators.

Mapping ecosystem services has gained increasing attention in the last years. This increase corresponds to advances in computing, modelling, methodological approaches, and consensus about the role of ecosystem services maps in providing a direct connection between ecosystem services and landscapes (Palomo et al., 2017). Maps are effective in communicating and organizing data. However, their utility depends on the availability of data and the purpose of the mapping and assessment (Dick et al., 2014). In line with the instrumental definition of ecosystem condition mentioned in section 1.2., information about the current condition of the ecosystem is necessary to map and assess the capacity of ecosystems to provide services. This information about the condition should include maps and assessments accompanied by information about the pressures that affect the condition directly or indirectly (Erhard et al., 2017).

In this study, the indicators for pressures, ecosystem condition and the ecosystem service *control of erosion rates* were quantified and spatially represented in the form of maps (see chapters 3 and 4). The selection of this ecosystem service has to do with its importance in maintaining soil quality and enabling additional provisioning, regulating and cultural services supply. Protecting soil from erosion is needed to sustain the capacity of ecosystems to provide services such as food production, nutrient regulation and water purification (Steinhoff-Knopp et al., 2021; Steinhoff-Knopp and Burkhard, 2018). All the selected indicators were imported into

ArcGIS 10.7 by ESRI for their analysis and mapping. Chapters 3 and 4 and the supplementary information associated with them provide a detailed description of the methods to quantify and map the indicators.

1.5.3. Statistical analyses

Statistics are valuable to make sense of the available data. Statistical analysis includes the design of the study and the selection and measuring of variables. They also guide the sampling, cleaning and selection of methodologies to analyse the data. Statistical analyses were carried out in this study, first, to identify the general characteristics of the mapping and assessment of ecosystem condition in Lower Saxony and Europe. Second, statistical analyses helped identify the correlations between the indicators of pressures, ecosystem condition and *control of erosion rates,* both in the regional and continental scales.

In the regional assessment, the use of Box-whisker-plots, the Akaike Information Criteria and the Kruskal-Wallis and Jonkheere -Tersptra tests allowed for the identification of the main characteristics of the data and the relationships between the different indicators. Additionally, the normalization of indicators contributed to creating multivariate maps that highlight priority areas. On the continental scale, the use of 2-dimensional boxplots, the Principal Component Analysis, and the Spearman correlation helped to recognize differences between the environmental zones and the variable importance of the indicators within them. All the statistical analyses were done using the software RStudio (version 1.2.1335) (RStudio Team, 2015) Chapters 3 and 4 provide a detailed description of the statistical methods.

1.5.4. Data acquisition

Pressures, ecosystem condition and the ecosystem service *control of erosion rates* are the main study subjects of this assessment. They represent the status of the agroecosystems in the regional and continental scales and their capacity to control soil erosion. Therefore, the search and acquisition of appropriate data constituted a significant part of the research and were necessary to map and quantify the different indicators.

A valuable data source was the CORINE Land Cover (CLC) dataset developed by the EEA and produced by most countries by visual representation of high-resolution satellite imagery. This dataset is updated every six years since the year 2000, and it contains an inventory of land cover in 44 classes presented in Minimum Mapping Units (MMU) of 25 ha for areal phenomena, 100 m for linear phenomena and 5 ha to identify the changes (European Environment Agency, 2021). The availability of data for different years and countries allows for the spatial and temporal comparison of land covers. The CLC dataset was used to delimitate the study areas and identify the changes in the extent of the agroecosystems in this study.

Additional data at the regional level were obtained from official sources in Germany and the State of Lower Saxony. Climate data, for instance, were provided by the Climate Data Center of the German Weather Service (of DWD, its initials in German). These data include measure parameters from the DWD stations, derived parameters at the station locations, grids for Germany and Europe, among others (Deutscher Wetterdienst, 2018). The Federal Research Institute for Rural Areas, Forestry and Fisheries (Thünen Institute) provided data on agricultural production and distribution of cropping areas. The Federal Institute for Geosciences and Natural Resources and the State Office for Mining, Energy and Geology provided soil data. Other soil data were obtained from the European Soil Data Centre (ESDAC) from the Joint Research Centre (JRC) due to the limited availability at the regional scale. All the data were re-scaled at the level of municipalities to allow for comparisons and spatial representation in multivariate maps.

For the continental scale, several sources were used to obtain the data and calculate the indicators. The Copernicus Climate Change Service implemented by the European Centre for Medium-Range Weather Forecasts (ECMWF) on behalf of the European Commission (Copernicus, 2019) provided climate data. Soil moisture data were available from the EEA and soil-related data from ESDAC upon request. EUROSTAT (European Statistical Office) provided statistical data about agricultural areas and practices. Additionally, different staff members of the Joint Research Centre provided data to calculate land-use intensity, the density of seminatural areas and control of soil erosion during a 3-month visit in 2019. However, most of these data are also available on the JRC data catalogue (https://data.jrc.ec.europa.eu/) or upon request. The data had different temporal and spatial scales and, when needed, they were rescaled or disaggregated to a 1 km grid level to allow for comparisons between the indicators. Chapters 3 and 4 provide a detailed description of the data, their sources and spatial and temporal scales.

1.5.5. Case studies

Quantification and mapping of pressure, condition and ecosystem services indicators were done at different spatial scales. The first case study chosen to analyse the relationships between agroecosystem condition and the provision of the ecosystem service *control of erosion rates* was the State of Lower Saxony in Northern Germany (see Figure 4). This federal state is a suitable case study as it is one of the most important locations for agriculture in Europe. However, it is also affected by increasing changes in climate, land use and land cover.

The geographical location, the climate and the soil quality are some relevant factors that play a role in the agricultural production in Northern Germany (Lower Saxony Ministry of Food Agriculture and Consumer Protection, 2019). Lower Saxony has the largest acreage of potatoes and sugar beet in Germany, and large amounts of the wheat, rye and barley consumed in the country come from this region. Animal breeding and husbandry are other significant factors in the agriculture and food industry in this State. With high shares of eggs production, meat and pork processing, milk and dairy products (*Ibid.*).

However, the climatic variations in the last years have had negative consequences on the state of agriculture. These changes affect some of the main crops that suffer from weather stress. In Lower Saxony, for instance, grains are impacted by wet autumns. Especially in 2017 when the sowing conditions were limited, reducing the areas dedicated to growing winter crops. On the other hand, in 2018, the dry conditions in the north and east reduced the hectare payments for grains by almost 18% and the tonnes of grain harvested compared to 2017 (Landwirtschaftskammer Niedersachsen, 2018). Additionally, there is a high risk of soil erosion in potatoes, sugar beet, corn and winter wheat crops. These high levels of soil loss are due to steep slopes, unsustainable tillage and farming practices and soil compaction (Federal Environment Agency, 2016).

Agriculture in Lower Saxony has a dual role related to climate change. As mentioned before, it is a victim of climate change, but also it is a cause of GHG emissions. Agriculture emits about 28 million tonnes of CO_2 equivalent, approximately 28% of the GHG emissions per year. Around 9.4 million tonnes of CO_2 equivalent correspond to the agricultural use of peatland and other carbon-rich soils (Lower Saxony Ministry of Food Agriculture and Consumer Protection, 2020). Another significant environmental impact from agriculture in Lower Saxony and, in general, in Germany is groundwater pollution. According to the German Federal Environment Agency, almost 88% of the nitrate in the groundwater in Germany comes from agricultural areas (below the root zone) (Federal Environment Agency, 2020).

Chapter 4 applies the indicators assessed previously in the regional case study at the European level (see Figure 4). The MAES working group proposed these indicators for EU-wide analyses.

Therefore, the agroecosystems in Europe (including cropland and grassland) were a good case study to test and apply the framework and indicators to assess pressures, ecosystem condition and control of erosion rates. Agriculture is the predominant soil cover type in Europe, accounting for around 40 % of the land surface (European Environment Agency, 2019b). Cereals are the most common crops, followed by oilseeds, protein crops, olives, fruits, and vineyards. Another important segment of the farming industry is livestock represented principally by sheep, goats, poultry cattle and pigs (EUROSTAT, 2018). Similar to Lower Saxony, agroecosystems in Europe have been suffering from increasing pressures associated with climate change, land use change and land degradation.

Climatic effects in Europe are evident due to the sensitivity of crops to prevailing weather conditions. However, changes in temperature and precipitation will have different impacts across the territory. For example, climate change will reduce crop productivity in southern Europe, but it will improve the conditions for growing crops in the northern parts due to the longer growing seasons (European Environment Agency, 2019b). Climate change will also have different impacts depending on the time of the year. For example, heavy spring frost can destroy blossoms and reduce the growth of cereals. Summer droughts can cause the decay of crops, whereas strong winds and heavy rain affect the structure of the plants and soil (EUROSTAT, 2018). Another impact will be the reduced water availability needed for irrigation, livestock watering, processing, storage and transport of agricultural products (European Environment Agency, 2019b).

Land take and soil sealing are the biggest threats to agriculture ecosystems and their productivity in Europe. Although the rate of land take had stalled, the pressures of fragmentation on rural landscapes remain (European Environment Agency, 2019a). Additional pressures relate to land use practices associated with the need for food, water, transport and settlements for the growing human population that drive these changes (Lambin and Meyfroidt, 2011). In several parts of Europe, agricultural land abandonment and land intensification happen simultaneously, and both cause adverse impacts on habitats and farmland species. This process occurs due to the abandonment of grasslands and the intensification of arable land. Or conversely, it takes place due to increasing pressures on land that lead to lower productivity and later abandonment (Temme and Verburg, 2011).

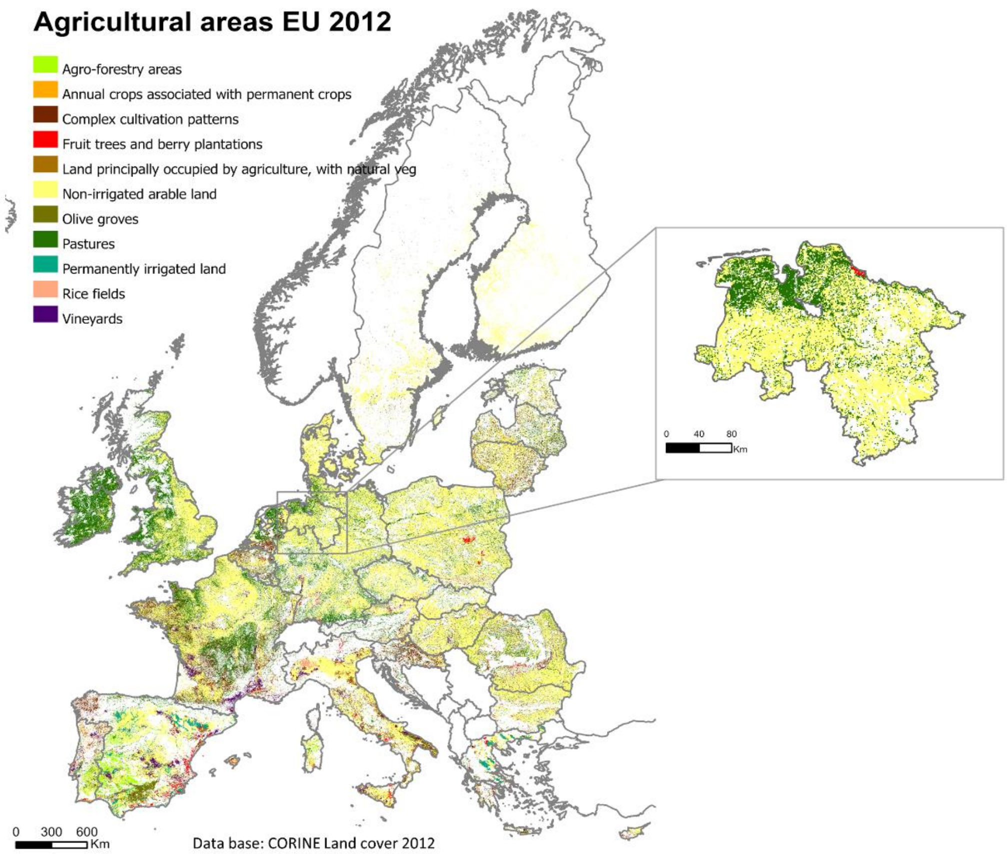

Figure 4. Map with the study areas. Agroecosystems in the EU 27 and the UK (left) and Lower Saxony (right).

1.6. Research questions and outline of the thesis

Each chapter of this study focuses on different aspects of the assessment of ecosystem condition and its relationship with ecosystem services and corresponds to an article published in a peer-reviewed international journal. The following research questions were proposed to achieve the overall objectives described in section 1.1.

- How has ecosystem condition been assessed in different European countries?
- What indicators can be used to better define the condition of an ecosystem?
- How can the proposed indicators contribute to making policy decisions for managing ecosystems?
- To what extent does the condition of an ecosystem determine its capacity to supply ES?

Figure 5 shows the structure of this study, the content of each chapter and their relationships and linkages. **Chapter one** provides the context and general framework that served as input for the design and development of the study. **Chapter two** analyses the trends in mapping and assessment of ecosystem condition in Europe. It provides an overview of the regions and ecosystems evaluated, the methods and indicators used to assess ecosystem condition, and the ecosystem services involved. The review highlights the methodological gaps principally associated with the relatively low spatial representation of indicators and the unequal distribution of assessments throughout the continent. It also identifies knowledge gaps, especially in the relationships between pressures, condition and ecosystem services, and the absence of holistic approaches to assess ecosystem condition.

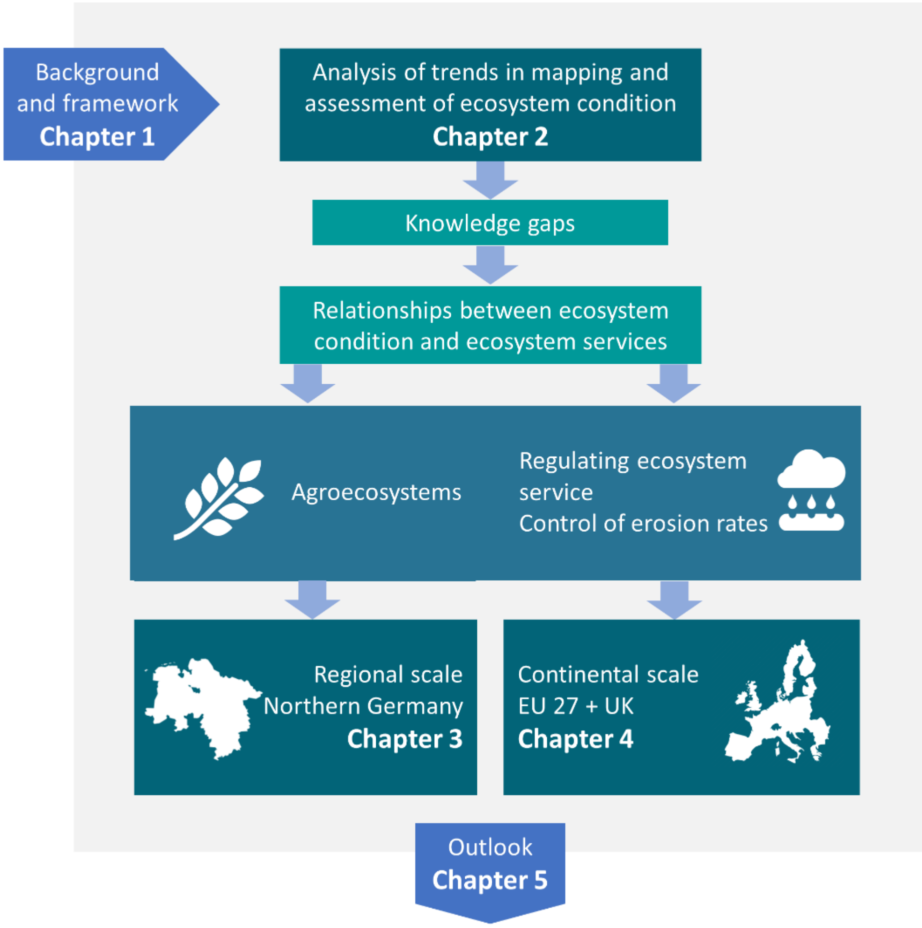

Figure 5. Structure of the study.

Based on the knowledge gaps identified in Chapter two, **Chapter three** aims to (1) test the indicators proposed by MAES to assess ecosystem condition and (2) improve the knowledge about the links between condition and services. It takes one exemplary ecosystem (agroecosystems) and one regulating ecosystem service (*control of erosion rates*) at a regional scale (Northern Germany). An operational framework for the integrated mapping and assessment of ecosystems and their services is used (Burkhard et al., 2018). This framework describes a stepwise approach that starts with the theme or policy question identification. Later, it covers the ecosystem type selection, the definition of ecosystem condition and services, and the quantification and mapping of indicators. It finalizes with the integration and dissemination of results. The assessment identifies some correlations between pressures and condition indicators with *control of erosion rates*. However, it also points out some limitations regarding the availability of spatially and temporally explicit data.

The same framework followed in Chapter three is used in **Chapter four** for the integrated assessment of agroecosystems and the ecosystem service *control of erosion rates*, but this time applied in the EU 27 and the UK. However, due to the size of the study area, the environmental stratification of Europe proposed by Metzger et al. (2005) was used to analyse the variations between regions with similar climatic and topographic characteristics. This study also identifies some correlations between the indicators but highlights considerable differences between the environmental zones. Furthermore, the chapter discusses the implications of the results and provides recommendations for future research and policy applications.

Chapter five provides a summary of the main results and a discussion of the study. It gives short answers to the research questions presented in Chapter one and draws the principal conclusions and the outlook towards future research.

References

Adhikari, K., Hartemink, A.E., 2016. Linking soils to ecosystem services - A global review. Geoderma 262, 101–111. https://doi.org/10.1016/j.geoderma.2015.08.009

Alexandratos, N., Bruinsma, J., 2012. World agriculture towards 2030/2015. The 2012 Revision, ESA Working paper. Rome.

Balmford, A., Fisher, B., Green, R.E., Naidoo, R., Strassburg, B., Turner, R.K., Rodrigues, A.S.L., 2011. Bringing Ecosystem Services into the Real World : An Operational Framework for Assessing the Economic Consequences of Losing Wild Nature. Environ. Resour. Econ. 48, 161–175. https://doi.org/10.1007/s10640-010-9413-2

Barbet-Massin, M., Salles, J.M., Courchamp, F., 2020. The economic cost of control of the invasive yellow-legged Asian hornet. NeoBiota 55, 11–25. https://doi.org/10.3897/NEOBIOTA.55.38550

Bellard, C., Cassey, P., Blackburn, T.M., 2016. Alien species as a driver of recent extinctions. Biol. Lett. 12. https://doi.org/10.1098/rsbl.2015.0623

Bernués, A., Rodríguez-Ortega, T., Ripoll-Bosch, R., Alfnes, F., 2014. Socio-Cultural and Economic Valuation of Ecosystem Services Provided by Mediterranean Mountain Agroecosystems. PLoS One 9, e102479. https://doi.org/10.1371/journal.pone.0102479

Bethwell, C., Burkhard, B., Daedlow, K., Sattler, C., Reckling, M., Zander, P., Bethwell, C., Daedlow, K., Reckling, M., Sattler, C., Zander, P., 2021. Towards an enhanced indication of provisioning ecosystem services in agroecosystems. Environ. Monit. Assess. 193, 1–37. https://doi.org/https://doi.org/10.1007/s10661-020-08816-y

Blanco-Canqui, H., Lal, R., 2010. Soil erosion and food security, in: Principles of Soil Conservation and Management. Springer Science+Business Media, Dordrecht Heidelberg London New York, pp. 493–512.

Bordt, M., 2015. Land and Ecosystem Condition and Capacity, SEEA Experimental Ecosystem Accounting Technical Guidance. New York.

Borrelli, P., Panagos, P., Ballabio, C., Lugato, E., Weynants, M., Montanarella, L., 2016. Towards a Pan-European Assessment of Land Susceptibility to Wind Erosion. L. Degrad. Dev. 27, 1093–1105. https://doi.org/10.1002/ldr.2318

Bradshaw, C.J.A., Leroy, B., Bellard, C., Roiz, D., Albert, C., Fournier, A., Barbet-Massin, M., Salles, J.M., Simard, F., Courchamp, F., 2016. Massive yet grossly underestimated global costs of invasive insects. Nat. Commun. 7. https://doi.org/10.1038/ncomms12986

Burkhard, B., de Groot, R.S., Costanza, R., Seppelt, R., Jørgensen, S.E., Potschin-Young, M., 2012. Solutions for sustaining natural capital and ecosystem services. Ecol. Indic. 21, 1–6. https://doi.org/10.1016/j.ecolind.2012.03.008

Burkhard, B., Kandziora, M., Hou, Y., Müller, F., 2014. Ecosystem service potentials, flows and demands - Concepts for spatial localisation, indication and quantification. Landsc. Online

34, 1–32. https://doi.org/10.3097/LO.201434

Burkhard, B., Santos-Martín, F., Nedkov, S., Maes, J., 2018. An operational framework for integrated Mapping and Assessment of Ecosystems and their Services (MAES). One Ecosyst. 3, e22831. https://doi.org/10.3897/oneeco.3.e22831

Burney, J.A., Davis, S.J., Lobell, D.B., 2010. Greenhouse gas mitigation by agricultural intensification. Proc. Natl. Acad. Sci. U. S. A. 107, 12052–12057. https://doi.org/10.1073/pnas.0914216107

Bussi, G., Dadson, S.J., Prudhomme, C., Whitehead, P.G., 2016. Modelling the future impacts of climate and land-use change on suspended sediment transport in the River Thames (UK). J. Hydrol. 542, 357–372. https://doi.org/10.1016/j.jhydrol.2016.09.010

Charles, H., Dukes, J., 2008. Impacts of Invasive Species on Ecosystem Services, in: Nentwig, W. (Ed.), Biological Invasions. Springer, Berlin, pp. 217–237. https://doi.org/https://doi.org/10.1007/978-3-540-36920-2_13

Copernicus, 2019. Copernicus Climate Change Service [WWW Document]. URL https://climate.copernicus.eu/index.php/ (accessed 7.31.19).

Czúcz, B., Condé, S., 2017. Note on definitions related to ecosystem conditions and their services based on different glossaries.

DeFries, R., Pagiola, S., 2005. Analytical Approaches for Assessing Ecosystem Condition and Human Well-being, in: Ecosytems and Human Well-Being: Current State and Trends, by Millenium Ecosystem Assessment. Washington: World Resources Institute.

Deutscher Wetterdienst, 2018. Climate Data Center (CDC OpenData) [WWW Document]. URL https://opendata.dwd.de/climate_environment/CDC/ (accessed 10.8.18).

Dick, J., Maes, J., Smith, R.I., Paracchini, M.L., Zulian, G., 2014. Cross-scale analysis of ecosystem services identified and assessed at local and European level. Ecol. Indic. 38, 20–30. https://doi.org/10.1016/j.ecolind.2013.10.023

Dudley, N., Attwood, S.J., Goulson, D., Jarvis, D., Bharucha, Z.P., Pretty, J., 2017. How should conservationists respond to pesticides as a driver of biodiversity loss in agroecosystems? Biol. Conserv. 209, 449–453. https://doi.org/10.1016/j.biocon.2017.03.012

Erhard, M., Banko, G., Abdul Malak, D., Santos-Martín, F., 2017. 3.5 Mapping ecosystem types and conditions, in: Burkhard, B., Maes, J. (Eds.), Mapping Ecosystem Services. Pensoft Publishers, so, pp. 77–82.

Erhard, M., Teller, A., Maes, J., Meiner, A., Berry, P.M., Smith, A., Eales, R., Papadopoulou, L., Bastrup-Birk, A., Ivits, É., Royo Gelabert, E., Digel, G., Petersen, J.-E., Reker, J., Cugny-Seguin, M., Kristensen, P., Uhel, R., Estreguil, C., Fritz, M., Murphy, P., Banfield, N., Ostermann, O., Abdul Malak, D., Marín, A., Schröder, C., Condé, S., Garcia-Feced, C., Evans, D., Delbaere, B., Naumann, S., McKenna, D., Gerdes, H., Graf, A., Boon, A., Stoker, B., Mizgajski, A., Santos-Martín, F., Jol, A., Lükewille, A., Werner, B., Romao, C., Desaulty, D., Wugt Larsen, F., Louwagie, G., Zal, N., Gawronska, S., Christiansen, T., 2016. Mapping and Assessment of Ecosystems and their Services: Mapping and assessing the condition of

Europe's ecosystems: Progress and challenges. https://doi.org/10.2779/351581

European Environment Agency, 2021. CORINE Land Cover [WWW Document]. URL https://land.copernicus.eu/pan-european/corine-land-cover

European Environment Agency, 2019a. The European environment-state and outlook 2020. Knowledge for transition to a sustainable Europe. https://doi.org/10.2800/96749

European Environment Agency, 2019b. Climate change adaptation in the agriculture sector in Europe. EEA Rep. 112.

EUROSTAT, 2018. Agriculture, forestry and fishery statistics. 2018 Edition, Eurostat. Luxembourg.

FAO, 2021. FAOSTAT - Comparison of Pesticide indicators [WWW Document]. URL http://www.fao.org/faostat/en/#compare (accessed 5.5.21).

FAO, 2019. World fertilizer outlook and trends to 2022.

FAO, 2017. The future of food and agriculture: trends and challenges. Rome.

FAO and ITPS, 2015. Status of the World's Soil Resources (SWSR) - Main report. Rome.

Federal Environment Agency, 2020. Reactive nitrogen flows in Germany 2010 -2014 (DESTINO Report 2). Final report. Dessau-Roßlau.

Federal Environment Agency, 2016. Erosion [WWW Document]. URL https://www.umweltbundesamt.de/en/topics/soil-agriculture/land-a-precious-resource/erosion#soil-erosion-induced-by-water-an-underestimated-danger-

Felipe-Lucia, M.R., Soliveres, S., Penone, C., Fischer, M., Ammer, C., Boch, S., Boeddinghaus, R.S., Bonkowski, M., Buscot, F., Fiore-Donno, A.M., Frank, K., Goldmann, K., Gossner, M.M., Hölzel, N., Jochum, M., Kandeler, E., Klaus, V.H., Kleinebecker, T., Leimer, S., Manning, P., Oelmann, Y., Saiz, H., Schall, P., Schloter, M., Schöning, I., Schrumpf, M., Solly, E.F., Stempfhuber, B., Weisser, W.W., Wilcke, W., Wubet, T., Allan, E., 2020. Land-use intensity alters networks between biodiversity, ecosystem functions, and services. Proc. Natl. Acad. Sci. U. S. A. 117, 28140–28149. https://doi.org/10.1073/pnas.2016210117

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science (80-.). 309, 570–574. https://doi.org/10.1126/science.1111772

Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337–342. https://doi.org/10.1038/nature10452

Greaver, T.L., Sullivan, T.J., Herrick, J.D., Barber, M.C., Baron, J.S., Cosby, B.J., Deerhake, M.E., Dennis, R.L., Dubois, J.J.B., Goodale, C.L., Herlihy, A.T., Lawrence, G.B., Liu, L., Lynch, J.A., Novak, K.J., 2012. Ecological effects of nitrogen and sulfur air pollution in the US: What do we know? Front. Ecol. Environ. 10, 365–372. https://doi.org/10.1890/110049

Guerra, C., Pinto-Correia, T., Metzger, M.J., 2014. Mapping Soil Erosion Prevention Using an Ecosystem Service Modeling Framework for Integrated Land Management and Policy. Ecosystems 17, 878–889. https://doi.org/10.1007/s10021-014-9766-4

IPBES, 2019a. The IPBES Global Assessment on Biodiversity and Ecosystem Services. Chapter 2.3. Status and Trends - Nature's Contributions to People (NCP). pp. 1–79.

IPBES, 2019b. Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Bonn. https://doi.org/10.1111/padr.12283

IPBES, 2018. The assessment report on degradation and restoration. Summary for policymakers.

IPCC, 2014. Climate Change 2014: Synthesis Report. Geneva. https://doi.org/10.1017/CBO9781139177245.003

Keith, H., Czúcz, B., Jackson, B., Driver, A., Nicholson, E., Maes, J., 2020. A conceptual framework and practical structure for implementing ecosystem condition accounts. One Ecosyst. 5, 1–54. https://doi.org/10.3897/oneeco.5.e58216

Lal, R., 2015. Restoring soil quality to mitigate soil degradation. Sustain. 7, 5875–5895. https://doi.org/10.3390/su7055875

Lal, R., 2014. Soil conservation and ecosystem services. Int. Soil Water Conserv. Res. 2, 36–47. https://doi.org/10.1016/S2095-6339(15)30021-6

Lambin, E.F., Meyfroidt, P., 2011. Global land use change, economic globalization, and the looming land scarcity. Proc. Natl. Acad. Sci. U. S. A. 108, 3465–3472. https://doi.org/10.1073/pnas.1100480108

Landwirtschaftskammer Niedersachsen, 2018. Jahresbericht 2018 - Landwirtschaftskammer Niedersachsen, Jahresbericht 2018.

Le, Q.B., Nkonya, E., Mirzabaev, A., 2016. Biomass Productivity-Based Mapping of Global Land Degradation Hotspots, in: Nkonya, E., Mirzabaev, A., von Braun, J. (Eds.), Economics of Land Degradation and Improvement - A Global Assessment for Sustainable Development. Springer, pp. 55–84. https://doi.org/https://doi.org/10.1007/978-3-319-19168-3_7

Levin, P.S., Williams, G.D., Rehr, A., Norman, K.C., Harvey, C.J., 2015. Developing conservation targets in social-ecological systems. Ecol. Soc. 20. https://doi.org/10.5751/ES-07866-200406

Li, Z., Fang, H., 2016. Impacts of climate change on water erosion: A review. Earth-Science Rev. 163, 94–117. https://doi.org/10.1016/j.earscirev.2016.10.004

Lin, B.B., 2011. Resilience in agriculture through crop diversification: Adaptive management for environmental change. Bioscience 61, 183–193. https://doi.org/10.1525/bio.2011.61.3.4

Longobardi, P., Montenegro, A., Beltrami, H., Eby, M., 2016. Deforestation Induced Climate Change : Effects of Spatial Scale. PLoS One 11, e0153357. https://doi.org/10.1371/journal.pone.0153357

Lower Saxony Ministry of Food Agriculture and Consumer Protection, 2020. Climate protection [WWW Document]. URL https://www.ml.niedersachsen.de/startseite/themen/landwirtschaft/nachhaltigkeit_und_klimaschutz/der-klimawandel---folgen-und-aufgaben-fuer-die-niedersaechsische-land--und-forstwirtschaft-145563.html (accessed 5.12.21).

Lower Saxony Ministry of Food Agriculture and Consumer Protection, 2019. The Agriculture and Food Industry in Lower Saxony.

MA Board, 2005. MA, Millennium Ecosystem Assessment. Washington, DC.

Maes, J., Driver, A., Czúcz, B., Keith, H., Jackson, B., Nicholson, E., Dasoo, M., 2020. A review of ecosystem condition accounts: lessons learned and options for further development. One Ecosyst. 5, e53485. https://doi.org/10.3897/oneeco.5.e53485

Maes, J., Teller, A., Erhard, M., Grizzetti, B., Barredo, J.I., Paracchini, M.L., Condé, S., Somma, F., Orgiazzi, A., Jones, A., Zulian, G., Petersen, J.-E., Marquardt, D., Kovacevic, V., Abdul Malak, D., Marin, A., Czúcz, B., Mauri, A., Loffer, P., Bastrup-Birk, A., Biala, K., Christiansen, T., Werner, B., 2018. Mapping and Assessment of Ecosystems and their Services: An analytical framework for ecosystem condition. Luxembourg. https://doi.org/10.2779/12398

Maes, J., Teller, A., Erhard, M., Liquete, C., Braat, L., Egoh, B., Puydarrieux, P., Fiorina, C., Santos-Martín, F., Paracchini, M.L., Keune, H., Wittmer, H., Hauck, J., Fiala, I., Verburg, P.H., Condé, S., Schägner, J.P., San-Miguel-Ayanz, J., Estreguil, C., Ostermann, O., Barredo, J.I., Pereira, H.M., Stott, A., Laporte, V., Meiner, A., Olah, B., Royo Gelabert, E., Spyropoulou, R., Petersen, J.-E., Maguire, C., Zal, N., Achilleos, E., Rubin, A., Ledoux, L., Brown, C., Raes, C., Jacobs, S., Vandewalle, M., Connor, D., Bidoglio, G., 2013. Mapping and Assessment of Ecosystems and their Services: An analytical framework for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020.

Malaj, E., Von Der Ohe, P.C., Grote, M., Kühne, R., Mondy, C.P., Usseglio-Polatera, P., Brack, W., Schäfer, R.B., 2014. Organic chemicals jeopardize the health of freshwater ecosystems on the continental scale. Proc. Natl. Acad. Sci. U. S. A. 111, 9549–9554. https://doi.org/10.1073/pnas.1321082111

Metzger, M.J., Bunce, R.G.H., Jongman, R.H.G., Mücher, C.A., Watkins, J.W., 2005. A climatic stratification of the environment of Europe. Glob. Ecol. Biogeogr. 14, 549–563. https://doi.org/10.1111/j.1466-822X.2005.00190.x

Mirzabaev, A., Nkonya, E., J, G., T, J., W, A., 2016. Global Drivers of Land Degradation and Improvement, in: Nkonya, E., Mirzabaev, A., von Braun, J. (Eds.), Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development. Springer, pp. 167–195. https://doi.org/https://doi.org/10.1007/978-3-319-19168-3_7

Paini, D.R., Sheppard, A.W., Cook, D.C., De Barro, P.J., Worner, S.P., Thomas, M.B., 2016. Global threat to agriculture from invasive species. Proc. Natl. Acad. Sci. U. S. A. 113, 7575–7579.

https://doi.org/10.1073/pnas.1602205113

Palomo, I., Bagstad, K., Nedkov, S., Klug, H., Adamescu, M., Cazacu, C., 2017. 3.4. Toold for mapping ecosystem services, in: Burkhard, B., Maes, J. (Eds.), Mapping Ecosystem Services. Pensoft Publishers, Sofia, Bulgaria, pp. 72–75.

Panagos, P., Standardi, G., Borrelli, P., Lugato, E., Montanarella, L., Bosello, F., 2018. Cost of agricultural productivity loss due to soil erosion in the European Union: From direct cost evaluation approaches to the use of macroeconomic models. L. Degrad. Dev. 29, 471–484. https://doi.org/10.1002/ldr.2879

Paul, C., Kuhn, K., Steinhoff-Knopp, B., Weißhuhn, P., Helming, K., 2020. Towards a standardization of soil-related ecosystem service assessments. Eur. J. Soil Sci. 1–16. https://doi.org/10.1111/ejss.13022

Pingali, P.L., 2012. Green revolution: Impacts, limits, andthe path ahead. Proc. Natl. Acad. Sci. U. S. A. 109, 12302–12308. https://doi.org/10.1073/pnas.0912953109

Poesen, J., 2018. Soil erosion in the Anthropocene: Research needs. Earth Surf. Process. Landforms 43, 64–84. https://doi.org/10.1002/esp.4250

Potschin-Young, M., Haines-Young, R., Heink, U., Jax, K., 2016. OpenNESS Glossary [WWW Document]. URL http://www.openness-project.eu/glossary (accessed 2.17.21).

Power, A.G., 2016. Can Ecosystem Services Contribute to Food Security?, in: Potschin-Young, M., Haines-Young, R., Fish, R., Turner, R.K. (Eds.), Routledge Handbook of Ecosystem Services. Routledge, London and New York, pp. 491–500.

Power, A.G., 2010. Ecosystem services and agriculture: Tradeoffs and synergies. Philos. Trans. R. Soc. B Biol. Sci. 365, 2959–2971. https://doi.org/10.1098/rstb.2010.0143

Quinton, J.N., Catt, J.A., 2007. Enrichment of heavy metals in sediment resulting from soil erosion on agricultural fields. Environ. Sci. Technol. 41, 3495–3500. https://doi.org/10.1021/es062147h

Quinton, J.N., Govers, G., Van Oost, K., Bardgett, R.D., 2010. The impact of agricultural soil erosion on biogeochemical cycling. Nat. Geosci. 3, 311–314. https://doi.org/10.1038/ngeo838

Rasmussen, L.V., Coolsaet, B., Martin, A., Mertz, O., Pascual, U., Corbera, E., Dawson, N., Fisher, J.A., Franks, P., Ryan, C.M., 2018. Social-ecological outcomes of agricultural intensification. Nat. Sustain. 1, 275–282. https://doi.org/10.1038/s41893-018-0070-8

Rickson, R.J., 2014. Can control of soil erosion mitigate water pollution by sediments? Sci. Total Environ. 468–469, 1187–1197. https://doi.org/10.1016/j.scitotenv.2013.05.057

RStudio Team, 2015. RStudio: Integrated Development for R. RStudio, Inc.

Schiefer, J., Lair, G.J., Blum, W.E.., 2016. Potential and limits of land and soil for sustainable intensification of European agriculture. Agric. Ecosyst. Environ. 230, 283–293. https://doi.org/10.1016/j.agee.2016.06.021

Schulte, R.P.., Creamer, R., Donnellan, T., Farrelly, N., Fealy, R., O'Donoghue, C., O'hUallachain, D., 2014. Functional land management: A framework for managing soil-based ecosystem services for the sustainable intensification of agriculture. Environ. Sci. Policy 38, 45–58. https://doi.org/10.1016/j.envsci.2013.10.002

Seebens, H., Essl, F., Dawson, W., Fuentes, N., Moser, D., Pergl, J., Pyšek, P., van Kleunen, M., Weber, E., Winter, M., Blasius, B., 2015. Global trade will accelerate plant invasions in emerging economies under climate change. Glob. Chang. Biol. 21, 4128–4140. https://doi.org/10.1111/gcb.13021

Smeets, E., Weterings, R., 1999. Environmental indicators : Typology and overview, European Environment Agency.

Stavi, I., Bel, G., Zaady, E., 2016. Soil functions and ecosystem services in conventional, conservation, and integrated agricultural systems. A review. Agron. Sustain. Dev. 36. https://doi.org/10.1007/s13593-016-0368-8

Stehle, S., Schulz, R., 2015. Agricultural insecticides threaten surface waters at the global scale. Proc. Natl. Acad. Sci. U. S. A. 112, 5750–5755. https://doi.org/10.1073/pnas.1500232112

Steinhoff-Knopp, B., Burkhard, B., 2018. Mapping Control of Erosion Rates: Comparing Model and Monitoring Data for Croplands in Northern Germany. One Ecosyst. 3, e26382. https://doi.org/10.3897/oneeco.3.e26382

Steinhoff-Knopp, B., Kuhn, T.K., Burkhard, B., 2021. The impact of soil erosion on soil-related ecosystem services: development and testing a scenario-based assessment approach. Environ. Monit. Assess. 193. https://doi.org/10.1007/s10661-020-08814-0

Stoddard, J.L., Larsen, D.P., Hawkins, C.P., Johnson, R.K., Norris, R.H., 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecol. Appl. 16, 1267–1276.

Temme, A.J.A.M., Verburg, P.H., 2011. Mapping and modelling of changes in agricultural intensity in Europe. Agric. Ecosyst. Environ. 140, 46–56. https://doi.org/10.1016/j.agee.2010.11.010

Terrado, M., Acuña, V., Ennaanay, D., Tallis, H., Sabater, S., 2014. Impact of climate extremes on hydrological ecosystem services in a heavily humanized Mediterranean basin. Ecol. Indic. 37, 199–209. https://doi.org/10.1016/j.ecolind.2013.01.016

United Nations, 2021. System of Environmental-Economic Accounting—Ecosystem Accounting: Final Draft.

Van Oost, K., Govers, G., de Alba, S., Quine, T.A., 2006. Tillage erosion: A review of controlling factors and implications for soil quality. Prog. Phys. Geogr. 30, 443–466. https://doi.org/10.1191/0309133306pp487ra

Vergano, L., Nunes, P.A.L.D., 2007. Analysis and evaluation of ecosystem resilience: An economic perspective with an application to the Venice lagoon. Biodivers. Conserv. 3385–3408. https://doi.org/10.1007/s10531-006-9085-y

Wentworth Group of Concerned Scientists, 2016. Accounting for Nature: A scientific method for constructing environmental asset condition accounts.

Zhang, W., Ricketts, T.H., Kremen, C., Carney, K., Swinton, S.M., 2007. Ecosystem services and dis-services to agriculture. Ecol. Econ. 64, 253–260. https://doi.org/10.1016/j.ecolecon.2007.02.024

2

Analysis of trends in mapping and assessment of ecosystem condition in Europe

Rendon, P., Erhard, M., Maes, J., & Burkhard, B.

Ecosystems and people (2019), 15(1), 156–172

ECOSYSTEMS AND PEOPLE
2019, VOL. 15, NO. 1, 156–172
https://doi.org/10.1080/26395916.2019.1609581

REVIEW

Analysis of trends in mapping and assessment of ecosystem condition in Europe

Paula Rendon[a], Markus Erhard[b], Joachim Maes[c] and Benjamin Burkhard[a,d]

[a]Institute of Physical Geography & Landscape Ecology, Leibniz Universität Hannover, Hannover, Germany; [b]Biodiversity and Nature, European Environmental Agency, Copenhagen, Denmark; [c]Land Resources Unit D.3, Joint Research Centre (JRC) European Commission, Ispra, Italy; [d]Leibniz Centre for Agricultural Landscape Research (ZALF) Müncheberg, Germany

ABSTRACT
Ecosystem condition is the overall quality of an ecosystem unit, in terms of its biological, physical and chemical characteristics underpinning its capacity to generate ecosystem services. Changes in ecosystem condition affect the delivery of services and therefore human well-being. Despite increasing research in this field, the relations between biodiversity, ecosystem condition and services are still not well understood. This study examined scientific articles and reports to analyse the development of ecosystem condition mapping and assessments in Europe since the year 2000. The aim was to provide an overview of the current state of research and to highlight some challenges for ecosystem condition and ecosystem services research. The review analysed the ecosystems under study, scales, methods, indicators, and the ecosystem services assessed. Based on this review, some gaps were identified, especially in the methods used for condition assessment, the coverage of ecosystems, and the applicability of indicators in policy. It is necessary to develop integrative methods to determine ecosystems condition and its influence on the ecosystem service provision, in order to produce robust information. The results of this review can be harnessed by people who need an overview about existing ecosystem condition studies, such as scientists, land managers or decision makers.

ARTICLE HISTORY
Received 15 May 2018
Accepted 15 April 2019

EDITED BY
Graciela Rusch

KEYWORDS
Review; ecosystem state; ecosystem services; mapping; assessment; ecosystem condition indicators

1. Introduction

Ecosystem condition is the overall quality of an ecosystem unit[1] in terms of its main characteristics underpinning its capacity to generate ecosystem services (Potschin-Young et al. 2016). This concept is commonly used as a synonym for '*ecosystem state*' which is the physical, chemical and biological condition of an ecosystem at a specific point in time which can also be referred to as its *quality* (Maes et al. 2014, 2018). In the EU, ecosystem condition includes the legal concept of *status* measured over time and compared to agreed targets, such as the European Union (EU) environmental directives (Water Framework Directive – WFD, Marine Strategy Framework Directive – MSFD, Birds, and Habitats Directives – BD and HD) (Maes et al. 2014).

Ecosystem condition also comprises descriptors related to the state, pressures and biodiversity of ecosystems that are suitable to analyse state and dynamics of complex social-ecological systems, a dimension that is not further investigated in this paper. These descriptors include *ecosystem health* that reflects the 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 (Costanza et al. 1992; HELCOM 2010; O'Brien et al. 2016). Hence, ecosystem health integrates environmental conditions with the impacts of anthropogenic activities (Burkhard et al. 2008). Another descriptor is *ecosystem integrity* defined as the structure, composition, function and degree of self-organization of an ecosystem operating within a natural range of variability that exhibits little or no human influence (Young and Sanzone 2002; Potschin-Young et al. 2018). *Ecosystem functioning* is a descriptor that involves the biogeochemical and physical processes that take place within an ecosystem which contribute to the overall performance of the system (Pinto et al. 2014; Potschin-Young et al. 2018).

The concept of ecosystem condition and how to measure it is still under debate. There are different definitions of condition and sometimes there is not a clear distinction between the condition and the potential/capacity of ecosystems to provide services (Potschin-Young et al. 2017). In addition, assessment of ecosystem condition is often difficult due to the lack of appropriate information and limited knowledge on the combined effects of pressures on ecosystem structures and processes (Erhard et al. 2017). However, the evaluation of the different descriptors mentioned above could provide a better picture of the condition

CONTACT Paula Rendon rendon@phygeo.uni-hannover.de
Supplementary data for this article can be accessed here.

of ecosystems and could contribute to understanding its role in the delivery of ecosystem services.

According to the working group on Mapping and Assessment of Ecosystems and their Services (MAES[2]) of the European Commission, the condition of ecosystems as well as their spatial accessibility determines the ability of ecosystems to deliver services that support human well-being (Maes et al. 2014, 2018). This implies the assumption that an ecosystem in good condition ensures the long-term, high-quality and sustainable delivery of ecosystem services. However, changes in ecosystem condition caused by drivers and pressures such as land use change climate change, or pollution and nutrient load can impair the ability of ecosystems to deliver these services in sufficient quantity and quality (Erhard et al. 2016). Biodiversity loss, as an effect of drivers and pressures, particularly, has a great impact on the delivery of ecosystem services due to the important role of biodiversity in the regulation of ecosystems processes and functioning (Maseyk et al. 2017).

In order to halt the loss of biodiversity and ecosystem services, the EU has adopted the Biodiversity Strategy to 2020 (European Commission 2011). Action 5 under Target 2 of this strategy states that all Member States of the EU should map and assess the state of ecosystems and their services in their territory, assess the economic value of such services and integrate these values into accounting and reporting systems at the national and EU level. However, the degree of implementation of Action 5 varies across the Member States of the EU. Some countries have undertaken national ecosystem assessments like the United Kingdom (UK National Ecosystem Assessment 2011)[3] or Spain (Spanish National Ecosystem Assessment 2014),[4] while others have undertaken regional or case study assessments like Belgium (Stevens et al. 2015), Germany (Lautenbach et al. 2011) and Italy (Rova et al. 2015). From these assessments, it was possible to identify the need to improve some aspects related to the framework, methods, and indicators used to map and assess ecosystem services across the different countries (Maes et al. 2014).

There is an increasing amount of literature assessing the links between biodiversity, natural capital, the world's overall stock of living and non-living resources and ecosystem services. As mentioned before, biodiversity has an important role in regulating the processes and functioning of ecosystems and consequently in their capacity to provide services. Furthermore, there is strong evidence on the positive influence of natural capital attributes (including biodiversity) on ecosystem services (Smith et al. 2017). According to Maseyk et al. (2017), many publications recognize the importance of natural capital and biodiversity in underpinning ecosystem services and benefits to humans. Smith et al. (2017) for example, demonstrated the ways in which natural capital influences the provision of ecosystem services based on a systematic review and the development of a typology to guide the application of the ecosystem approach. Harrison et al. (2014) used a network analysis to visualize the relationships between ecosystem services providers, their biodiversity attributes and the influence of abiotic factors. However, a major challenge that remains in the research of ecosystems, their condition and their services is that the interdependencies between biodiversity, ecosystem components, processes, functioning, and ecosystem services are still not well understood and many uncertainties remain due to the complexity of those relationships (Bastian 2013; Schneiders et al. 2012; Balvanera et al. 2014).

This review analyses the trends in the development of mapping and assessment of ecosystem condition in Europe since the year 2000 using a non-statistical meta-analysis of scientific articles, and national and international reports. The aim is to (1) provide an overview of the past and current state of research on ecosystem condition, (2) to highlight knowledge gaps and (3) to identify research needs when incorporating ecosystem condition in mapping and assessment of ecosystem services. In the following section, the methods used in the analysis of the literature are described. Section 3 provides the results of the literature review, focusing on patterns of ecosystem condition publications by country of origin, characteristics of the assessments, indicators, and ecosystem services. These results are discussed in Section 4, followed by the conclusions with a special emphasis on the applicability of findings and indicators in policy making and research needs in Section 5.

2. Methods

A non-statistical meta-analysis was conducted by assessing scientific journal articles and national and international reports of European case studies published in English from 2000 to 2017. The objective of this analysis was to identify the main characteristics and trends of ecosystem condition research. The recommendations for systematic reviews of the PRISMA statement, which were originally suggested for the review of medical studies, but are also applicable to other fields (Moher et al. 2009), were taken into account (see Suppl. Material 2 for the complete PRISMA checklist and its application in this study). Various combinations of the terms 'ecosystem', 'ecological', 'environment', 'environmental', 'biological' and descriptors of ecosystem 'condition', 'state', 'status', 'health', 'integrity', 'functioning' and 'quality' (e.g. 'ecosystem condition' AND 'assessment') specifically for European studies were used in an initial screening that took place from July to November 2017. The articles were sourced from the science databases Web of Science, Science Direct, Scopus, as well as Google Scholar. The national and international reports were sourced mainly from Google and Google Scholar.

The review was divided into four phases (see Suppl. Material 3). First, 15,313 articles and reports were identified from the different databases and with the different combinations of the search terms. Second, duplicated entries, indices, and retracted publications were removed and the titles and abstracts of the remaining 2036 publications were screened. This resulted in the rejection of 1531 publications that did not include at least one case of ecosystem condition assessment in Europe. Third, we read through the full texts of 505 publications to examine whether they were eligible for further analysis and another 105 publications were excluded. Fourth, 401 publications, including reports (see Suppl. Material 4), were analysed using seven criteria for comparison (Table 1). The information about each article was recorded in a database.

The analysis covered the different methodologies and the type of information used in the case studies. The classes of methods and data were selected based on a previous screening of the literature before identifying the papers to be analysed. The spatial scale was identified as the total extent of the area assessed or mapped, except in the cases where more than one area was studied. In those cases, the scale was local or regional depending on the study. The ecosystem types were classified using the ecosystem typology proposed by the MAES working group (Maes et al. 2013) with terrestrial, freshwater and marine environment in level 1 and their corresponding sublevels (see Table 1).

The papers and reports were also analysed in terms of whether they mention the ecosystem service concept or not, and if they assess ecosystem services. For the analysis of the articles that did assess ecosystem services, the Common International Classification of Ecosystem Services (CICES[5]) version 5.1 was used, including the sections of provisioning, regulating and maintenance, and cultural ecosystem services, their divisions, groups and classes (Haines-Young and Potschin-Young 2018). The ecosystem services classified using other systems such as the Millennium Ecosystem Assessment (MA 2005) or the TEEB classification (TEEB 2010) were translated into the CICES system to facilitate the comparison and analysis using the online BBN ecosystem service classification tool.[6] This tool tabulates the classes of CICES against the services listed in the MA, TEEB, UK NEA and the Belgian classification and expresses the probabilities of finding the number of categories in the other system that could correspond to the one that is being assessed. In those cases where a single service of the MA or TEEB appeared in multiple CICES classes, the service with the highest percentage was selected. In the cases with equal percentages, the ecosystem services were linked to the CICES classification, based on the information provided in the study.

Information about the indicators used in the assessments was also recorded in the database. These indicators were grouped into the classes: pressure, state[7] and biodiversity indicators and related subclasses. The classes and subclasses were built based on a preliminary system that was proposed on a MAES workshop on mapping and assessment of ecosystem condition that took place in June 2017 (European Commission, European Environment Agency, Joint Research Centre, European Topic Centre for Biodiversity, European Topic Centre Urban Land Use Systems 2017). We are aware that the class definitions and the assignment of individual indicators from the reviewed studies to respective classes may in some cases have been somewhat arbitrary. There is, however, up to today no respective categorisation system for ecosystem condition indicators available and the used system proved to be pragmatic.

Pressure indicators are an important part of the assessment of ecosystem condition because they contribute to determine the reasons why an ecosystem is in a certain state (Maes et al. 2018). State and biodiversity indicators provide a general picture of the quality of environmental compartments and biological elements of an ecosystem. Additionally, information regarding variables, measures, factors and properties such as physical features (area, altitude, depth, etc.), socioeconomic and climate information, among others, were described for each case study and recorded in a database. Table 1 gives an overview of all criteria, classes and subclasses that were used to classify the reviewed articles.

3. Results

3.1. Number of publications on ecosystem condition assessments

There has been an exponential growth in the number of studies assessing ecosystem condition in Europe, from 2 in 2002 to more than 15 per year since 2007 (Figure 1). Additionally, the number of case studies that mention the ecosystem services concept together with ecosystem condition also increased in that period, from 3 in 2005 to more than 10 per year since 2010. The journals *Ecological Indicators, Marine Pollution Bulletin, Science of the Total Environment* and *Estuarine, Coastal and Shelf Science* are the leading journals publishing ecosystem condition-related articles. Other journals that publish ecosystem condition assessments are mainly in the disciplines of freshwater and marine biology, ecology, environmental management, biodiversity and conservation. National and international reports have focused

Table 1. Criteria used to compare the ecosystem condition assessments.

Criteria	Classes and subclasses	Rationale
Country/region		Country or region where the study takes place
Purpose of the study		
	Mapping	Creation of maps of ecosystem condition
	Assessment	Analysis of amount, value or quality of ecosystems
Methodology/data		
	GIS	Geographic information systems used to analyse and present spatial of geographic data of ecosystems
	Remote sensing	Satellite or sensor-based information used to analyse ecosystems
	Modelling	Models used to represent ecosystems, their structures and processes, and possible changes under different scenarios
	Scenarios	Scenarios to describe the future conditions of an ecosystem under different driving forces and changes
	Physical analysis	Measurements of physical properties of an ecosystem
	Chemical analysis	Measurements of chemical properties of an ecosystem
	Biological analysis	Measurements of biological properties of an ecosystem
	Surveys	Measurements of opinions of a group of people about aspects of the study area
	Interviews/Workshops/ expert opinion	Conversations with interested parties about aspects of the study area
	Review other sources	Studies that assess literature about an ecosystem or a region
Spatial scale		
	Local	Specific geographic position such as farms, villages, small administration units, cities
	Regional	Administrative unit or area with similar characteristics e.g. Flanders, Baltic region
	National	Countries or states
	European	Continent of Europe or Member States of the European Union + additional countries in Europe
	Global	Worldwide
Ecosystem Type (Maes et al., 2013)		
	Terrestrial	
	Urban	Areas where most human population lives. Includes urban, industrial, commercial, and transport areas, urban green areas, mines, dumping and construction sites
	Cropland	Food production area including cultivated agricultural, horticultural, and domestic habitats and agro-ecosystems with significant coverage of natural vegetation
	Grassland	Areas of grassy vegetation, includes managed pastures and natural or seminatural grasslands
	Woodland and forest	Areas of woody vegetation of various age or they have succession climax vegetation types on most of the area
	Heathland and shrub	Areas with vegetation dominated by shrubs or dwarf shrubs, includes moors, heathland and sclerophyllous vegetation
	Sparsely vegetated land	Unvegetated or sparsely vegetated habitats (naturallyunvegetated areas), includes rocks, glaciers, dunes, beaches and sand plains
	Wetlands	Areas of water-logged specific plant and animal communities, includes natural or modified mires, bogs, fens and peat extraction sites
	Freshwater	
	Rivers and lakes	Permanent freshwater inland surface waters, include water courses and water bodies
	Marine	
	Marine inlets and transitional waters	Ecosystems on the land-water interface under the influence of tides and with salinity higher than 0.5 %. Include coastal wetlands, lagoons, estuaries and other transitional waters, fjords and sea lochs as well as embayments
	Coastal	Coastal, shallow, marine systems that experience significant land-based influences. Depth is between 50 and 70 m
	Shelf	Marine systems away from coastal influence, down to the shelf break. They are usually about 200 m deep.
	Open ocean	Marine systems beyond the shelf break. Depth is beyond 200 m.
Types of ES (CICES Version 5.1)		
	Provisioning	Nutritional, non-nutritional material and energetic outputs from living systems as well as abiotic outputs (including water).
	Regulation and maintenance	Ways in which living organisms can mediate or moderate the ambient environment that affects human health, safety or comfort, together with abiotic equivalents
	Cultural	Non-material, and normally non-rival and non-consumptive, outputs of ecosystems (biotic and abiotic) that affect physical and mental states of people.
Types of indicators (European Commission et al., 2017)		
	Pressure Indicators	Indicators of human activities that exert pressures on the environment, as a result of production or consumption processes.
	Human disturbance	Human activities that generate changes in ecosystems, includes fishing, water consumption, human population, etc.
	Mining	Extraction of minerals or geological materials from the earth
	Climate change	Changes in climate patterns in a global or regional level associated to high concentration of CO_2 in the atmosphere
	Natural system modifications	Activities that convert or degrade habitats largely as a result of human management, include hydromorphological changes, eutrophication, dredging, etc.
	Agriculture	Production of food through the cultivation of land or breeding of animals
	Sylviculture	Production of wood and fibres through the cultivation and growing of trees
	Urbanisation	Increase in the proportion of population living in cities, caused by the movement of people from rural to urban areas, includes indicators of urban surface areas, housing, etc.
	Invasive Alien Species	Live specimen of a species, subspecies or lower taxon of animals, plants, fungi or microorganisms introduced outside its natural range
	Pollution	Introduction of harmful substances into the environment, include pollutants concentration, urban solid waste, atmospheric deposition, etc.

(*Continued*)

Table 1. (Continued).

Criteria	Classes and subclasses	Rationale
	Fragmentation	Division of a landscape into smaller and often disconnected pieces
	State Indicators	Indicators of the quality of environmental compartments
	Land Use	Describe the use of land by humans
	Environmental state	Indicators of air, water, soil and ecosystem quality
	Red List conservation status	Description of the conservation status of biological species compared against the criteria used in the IUCN Red List of Threatened Species
	Conservation status	Description of the conservation status of habitats and species according to the article 17 of the EU Habitats directive
	Biodiversity Indicators	Description of the state of biological quality elements
	Species diversity	Description of the number of species, includes richness, abundance, distribution, evenness, etc.

mainly on the assessment of the environmental state of the country or region as a whole or in a specific ecosystem like marine, freshwater, forest and wetlands in a larger spatial scale.

This review identified that the term *status* in combination with the words 'environment', 'environmental', 'ecological' and 'biological' was mentioned most often (27% of the publications) to refer to ecosystem condition. *Quality* is next with 20%, followed by *condition* and *functioning* (13% each), *state* (11%), *integrity* (9%) and health (7%). The use of these terms has varied in recent years. *Status* and *functioning* were regularly used since 2007 and 2008, and during the last years, their use has increased. A similar situation is evident with the terms *condition* and *state* with an upward trend since 2012. On the contrary, *health* was used more often in 2010 and *quality* and *integrity* were mentioned since 2012, but in recent years their use has decreased.

3.2. Origin of ecosystem condition assessment publications

The assessment of ecosystem condition is unequally distributed over the European territory, and the number of publications shows significant differences among European countries. Spain leads the number of publications on ecosystem condition assessment with 61 case studies, followed by Italy with 43 studies, whereas there are many countries with no publications at all,[8] and some countries (Hungary and Slovakia) have only one publication. Thirty-nine studies were conducted on a European scale and 33 studies on a transnational scale which covered different spatial scales (local, regional, European) simultaneously (Figure 2).

3.3. Characteristics of the ecosystem condition assessments

Only 84 out of the 401 analysed publications included mapping of ecosystem condition. Maps on the condition of woodlands and forests as well as grasslands were most commonly found in the literature, whereas marine and coastal ecosystems were mapped in just a few studies. The spatial resolution used or presented in the maps varied across the studies, ranging from very high resolutions of 1–5 m to much lower spatial resolutions of 10–16 km.

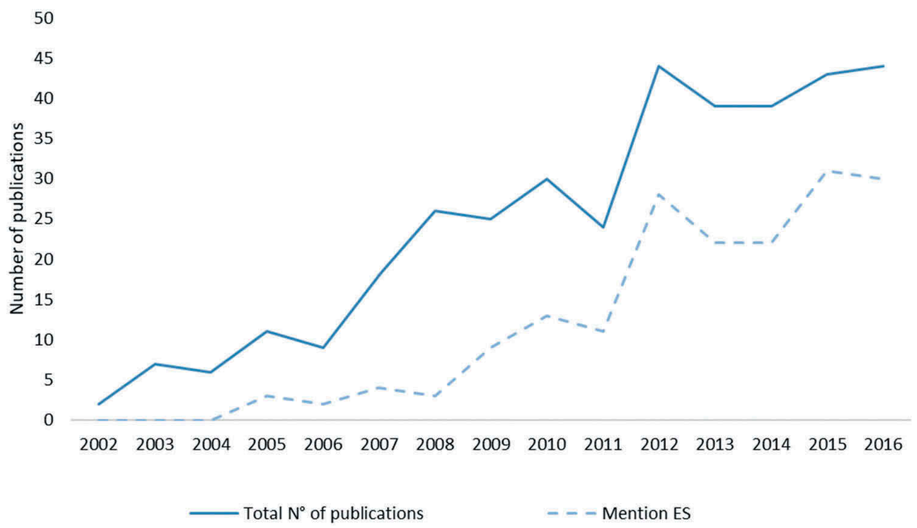

Figure 1. Number of studies related to ecosystem condition published between 2002 and 2016.

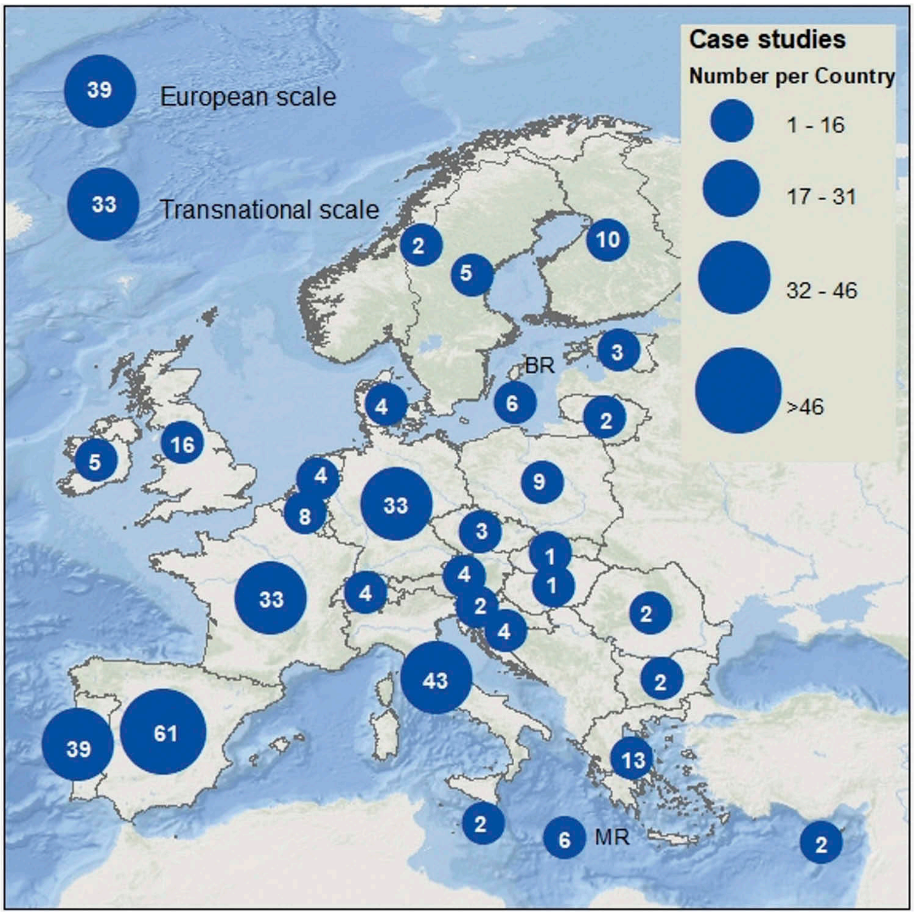

Figure 2. Geographical location of studies mapping and assessing ecosystem condition in Europe. BR: Baltic Region, MR: Mediterranean Region.

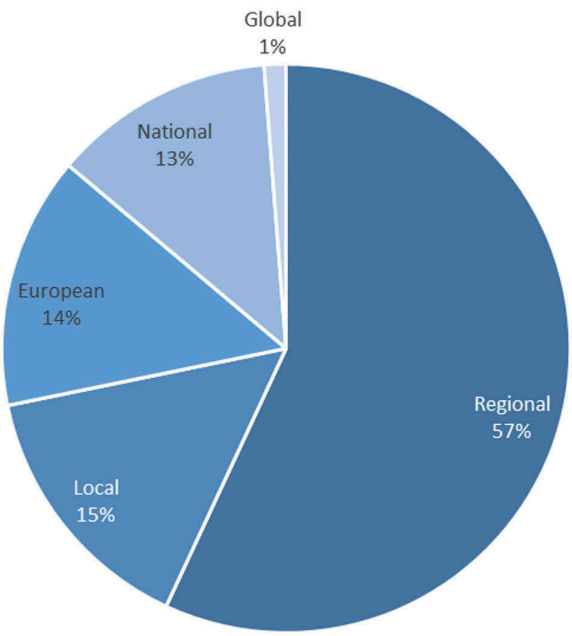

Figure 3. Spatial scale of publications on ecosystem condition assessments.

The spatial scales used to describe ecosystem condition were also diverse. More than half of the publications (236 studies, 58%) were conducted on regional scales, mostly within countries, followed by local assessments with 60 publications (15%). European and national scales were next with 52 and 51 cases, respectively (13% each). Only 5 publications assessed ecosystem condition on a global scale (1%) (Figure 3).

Figure 4 shows the scale of the study sites for each MAES ecosystem type (Maes et al. 2013). Studies on a local scale were conducted mainly in marine inlets and transitional waters, rivers and lakes and woodlands and forests, followed by grasslands and croplands. This reflects the characteristic design of the studies, which tend to be conducted in a particular site or ecosystem such as a lake or a forest. For rivers and lakes, marine inlets and transitional waters and coastal ecosystems, there were more studies at the regional scale, as many of these studies were carried out at the level of a catchment. There are not many studies conducted on larger scales, possibly because of the difficulty to measure the condition of ecosystems in larger areas.

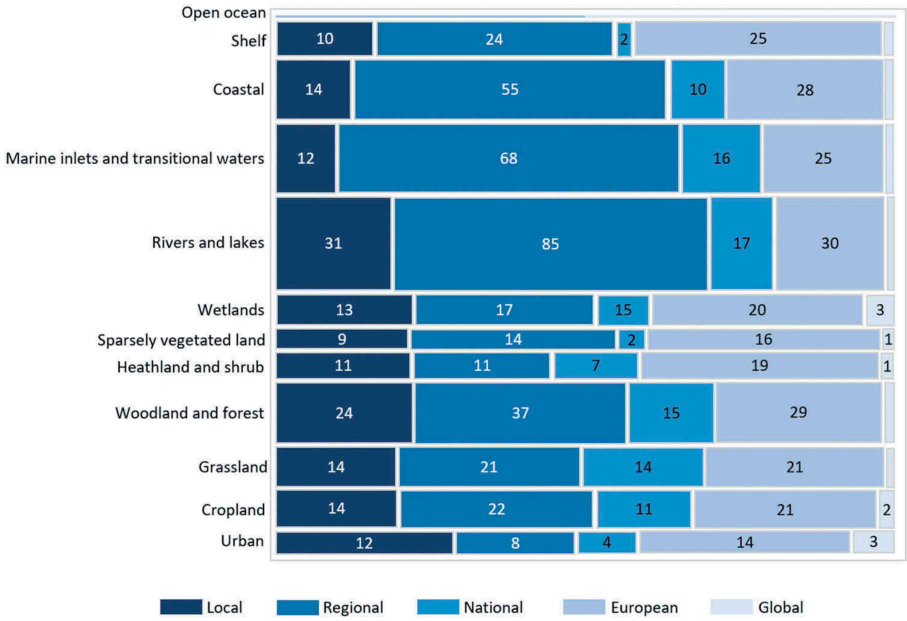

Figure 4. Number of studies on the different scales by ecosystem types (there can be more than one ecosystem type per study).

Most of the global studies are reviews, although there are some analyses of global datasets like a study on the relationships between the condition of ecosystems and the nursery function (Liquete et al. 2016a). The reduced amount of global studies in this review shows that estimations of global ecosystem conditions are still quite undeveloped. However, the lack of global studies could also be related to the fact that this review was mainly focused on original studies, so many review-type publications were not assessed.

Within the 401 publications, 889 ecosystem types were described. That means that on average two ecosystem types were mapped and/or assessed per publication. The most frequently assessed ecosystem types were rivers and lakes as well as marine inlets and transitional waters (164 and 123 times) (Figure 4).

The methods used in the studies and the sources of data varied within the publications: 58% of the studies used biological, physical and chemical analyses, direct measurements and monitoring data of different ecosystems and their components. Thirteen percent of the studies used modelling approaches such as food-web modelling to estimate carbon flux (e.g. Tecchio et al. 2015), to quantify the health status of the natural system (Piroddi et al. 2016) or to predict the effects of toxicity and ecological interactions on a river ecosystem (Grechi et al. 2016). Scenarios, remote sensing, surveys and interviews or group activities were less commonly used to assess ecosystem condition.

3.4. Ecosystem condition and ecosystem services

One of the objectives of this review was to identify research needs when incorporating ecosystem condition in ecosystem services mapping and assessment. For this purpose, the ecosystem condition studies identified before were reviewed in order to find out whether ecosystem services were mentioned and assessed. One-hundred-ninety-three studies mentioned the ecosystem services concept, but only 36% of these studies actually assessed them. The Millennium Ecosystem Assessment classification was most commonly used (86% of the studies), followed by CICES (11%) and TEEB (3%).

All three ecosystem services categories included in the CICES classification have received some attention according to this review. Of the studies that assess ecosystem services, the regulation and maintenance services received the greatest attention with 68 studies, followed by provisioning services with 60 studies and cultural services with 43 studies.

3.4.1. Regulating and maintenance ecosystem services

Regulating and maintenance services were the most commonly found services in this review. Almost 40% of the studies assessed services in this category, in particular within the CICES group of *life-cycle maintenance, habitat and gene pool protection* (CICES code 2.2.2) (42 studies), which includes *pollination* (2.2.2.1), *seed dispersal* (2.2.2.2), and *maintenance of nursery populations and habitats* (2.2.2.3). Another relevant group is the *regulation of baseline flows and extreme events* (2.2.1) (37 studies) in which *hydrological cycle and water flow regulation* (2.2.1.3) and *control of erosion rates* (2.2.1.1) are the services most assessed. The group

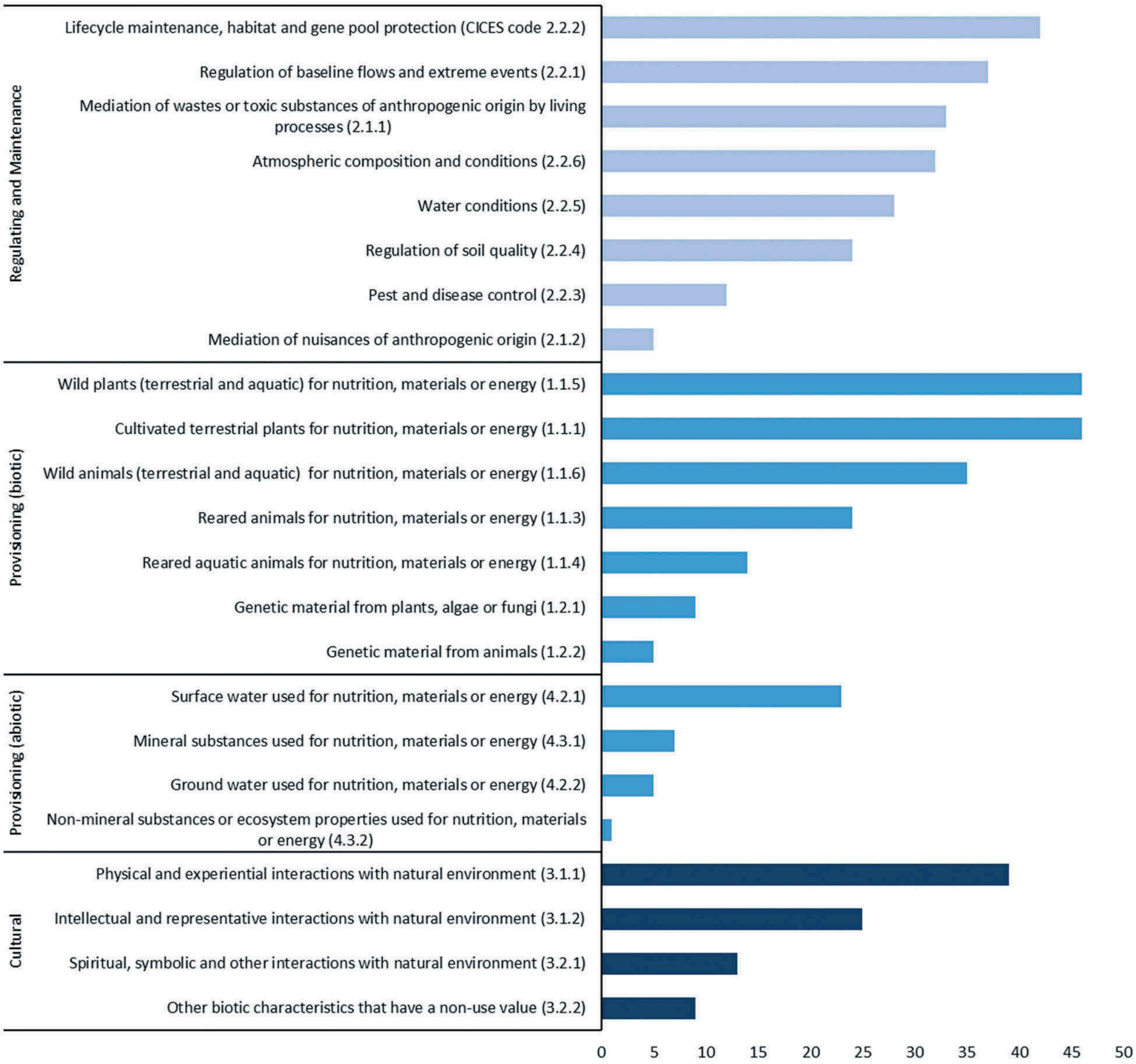

Figure 5. Number of publications per ecosystem service (on CICES group level).

mediation of nuisances of anthropogenic origin (2.1.2) received the least attention (Figure 5). *Mediation of waste, toxics and other nuisances by non-living processes* (5.1.1) and *maintenance of physical, chemical, biological conditions* (5.2.2) were not assessed in the studies reviewed.

3.4.2. Provisioning ecosystem services

Provisioning services (biotic and abiotic) were the second most common ecosystem services found in this review (35%). Among the studies that assessed biotic provisioning services, *wild plants (terrestrial and aquatic)* (1.1.5) and *cultivated terrestrial plants used for nutrition, materials or energy* (1.1.1) received the most attention (46 studies each group). These cover mostly *cultivated terrestrial plants (including fungi, algae) grown for nutritional purposes* (1.1.1.1) and *wild plants (terrestrial and aquatic, including fungi, algae) used for nutrition* (1.1.5.1) respectively. The provisioning services *wild animals (terrestrial and aquatic) for nutrition, materials or energy* (1.1.6) were also broadly covered in this section. In the abiotic provisioning services, 23 studies assessed the provision *of surface water for nutrition, materials or energy* (4.2.1), including *water for drinking* (4.2.1.1) and *for not drinking purposes* (4.2.1.2). *Non-mineral substances or ecosystem properties used for nutrition, materials or energy* (4.3.2), specifically *wind energy* (4.3.2.3) received the least attention among the provisioning services (Figure 5). *Cultivated aquatic plants for nutrition, materials or energy* (1.1.2) and *mineral* (4.3.1) and *non-mineral substances or ecosystem properties used for nutrition, materials or energy* (4.3.2) were not assessed in the publications.

3.4.3. Cultural ecosystem services

Cultural services were the least assessed services (25%). Among the studies that assessed cultural services, *physical and experiential interactions with natural environment* (3.1.1) which includes *characteristics of living* *systems that enable activities promoting health, recuperation or enjoyment through active or immersive interactions* (3.1.1.1) and *passive or observational interactions* (3.1.1.2) received the most attention with 39 studies. Followed by *intellectual and representative interactions with natural environment* (3.1.2) with 25 studies. *Other biotic characteristics that have a non-use value* (3.2.2) received the least attention in the cultural ecosystem services section (Figure 5). Assessments of abiotic cultural services were not identified in the literature.

3.5. Indicators used in ecosystem condition assessments

The 5th MAES report on ecosystem condition proposes a series of indicators for mapping and assessment of condition per ecosystem type, as well as for thematic assessments across different ecosystems (Maes et al. 2018). Some of the main indicators presented in the MAES report were also confirmed by the reviewed literature, which means that the indicators used in the assessment of ecosystem condition are in line with those proposed by MAES. These indicators, together with information about ecosystem extent and services, are important inputs for integrated ecosystem assessments (Burkhard et al. 2018). The frequency of the indicators used for ecosystem condition assessment in the reviewed publications is presented in Supplementary material 1.

Three-hundred-sixty-two studies used state indicators for assessing ecosystem condition and were focused mainly on environmental state, 270 used pressure indicators and looked mostly at human disturbance, pollution and natural system modifications, and 216 studies used biodiversity indicators (Figure 6).

3.5.1. State, status or condition indicators

State indicators reflect the condition of ecosystems based, for instance, on data on water and soil quality. Status indicators describe the condition of ecosystems in a legally defined framework such as the distribution and conservation status of species and habitats (Erhard et al. 2016). The state indicators most commonly found in the reviewed literature were those that provide information on the *environmental status* of the ecosystems (Figure 6) as, reported under the EU Birds Directive, Habitats Directive, WFD and MSFD.

Environmental condition indicators covered by the WFD and MSFD that determine the chemical and ecological status of water bodies and marine ecosystems based on physical, chemical and biological analyses were often found in the literature. Such indicators provide information on oxygenation conditions (e.g. dissolved oxygen), nutrient conditions (e.g. nitrates, phosphates and ammonium concentrations), salinity and acidification status (Pascoal et al. 2003; Ioannou et al. 2009; Stein et al. 2010; Roig et al. 2015 among others). Additionally, some indicators describe the *environmental state* of fresh and marine waters based on the analysis of biological elements or features. The

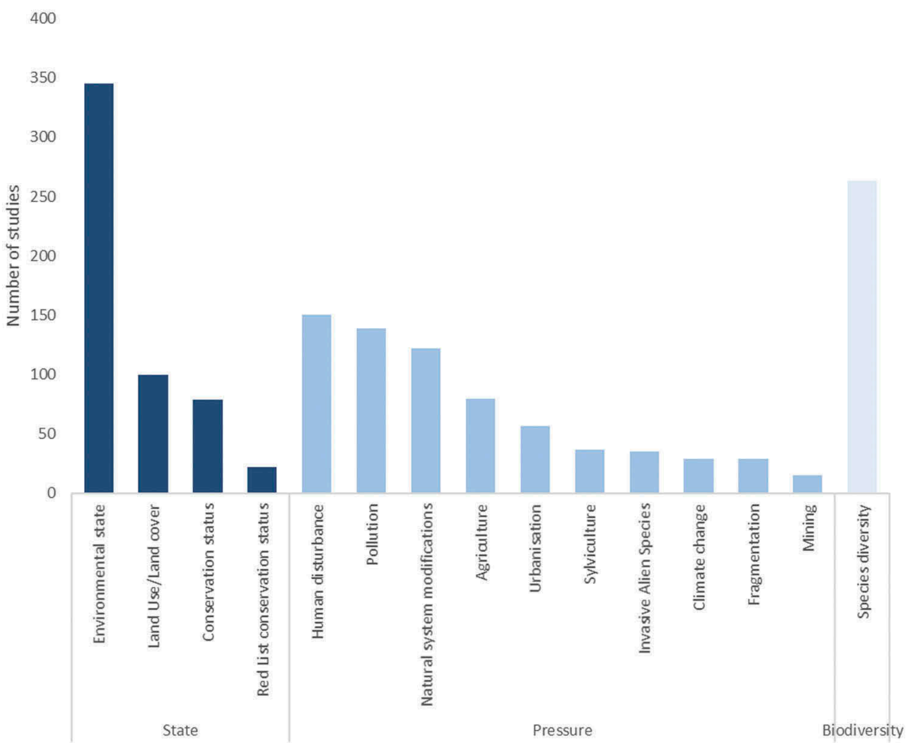

Figure 6. Types of indicators used in ecosystem condition assessments (there can be more than one type of indicator per study).

Ecological Quality Ratio (EQR) for instance, was often calculated based on indicators and reference values of macrophytes and macroinvertebrates (Sutela et al. 2013), seaweeds (Juanes et al. 2008), marine food webs, and concentration of contaminants, among others (Borja et al. 2011).

Similarly, soil characteristics and processes were also covered in some of the studies as indicators of *environmental state*. The most common indicators related to soil condition were soil nitrogen and carbon, carbon-to-nitrogen ratio (Kahmen et al. 2005), soil organic matter, phosphorus compounds (Mulder et al. 2011), evaporation/transpiration, nitrogen mineralization, and nitrogen balance (Müller et al. 2006). Additionally, indicators that include parameters related to land use, land cover (European Environment Agency 2010), land characteristics and landscape functions such as waste treatment and nutrient regulation (see Kienast et al. 2009 for more detailed information) were also taken into account to determine the condition of ecosystems.

Conservation status indicators that are included in the Nature Directives (Birds and Habitats Directives) were used in the reviewed studies, for example, estimates of population size or habitat area (Putkuri et al. 2013), and conservation status of habitats and species (Herrero and Castañeda 2009; Müller et al. 2010; European Environment Agency 2015). *Red list conservation status indicators* were mostly shown as the percentage distribution of red list categories (Aarts and Nienhuis 2003; Stevens et al. 2015).

3.5.2. Pressure indicators

Pressures refer to those actions that have an impact on ecosystem condition or state (Erhard et al. 2016). Pressures affect ecosystem condition depending on their strength, persistence and change over time (Erhard et al. 2017). In this review, indicators related to *human disturbance* were most commonly found in the group of pressure indicators with parameters that include fishing (Kenny et al. 2009), water consumption (Brkić et al. 2010), shipping (Korpinen et al. 2013) high human population density (Garnier and Billen 2007) and tourism (Gobert et al. 2009).

The second type of pressure indicator was *pollution*, mainly in freshwater and estuarine ecosystems, including concentrations of pollutants in water bodies (von der Ohe et al. 2009), in fish tissues (Corsi et al. 2003), plants and sediments. The main sources of pollution were agriculture and waste from urban areas. Another type of pressure indicators refers to *natural system modifications* such as hydromorphological pressures (Ippolito et al. 2010; Borja et al. 2013; Maceda-Veiga et al. 2014), anthropogenically affected shoreline due to settlement areas, intensity of shore use and farmland areas (Brämick et al. 2008).

3.5.3. Biodiversity indicators

Biodiversity indicators are measurable characteristics that provide information about a changing element of biodiversity (Harrington et al. 2010). The indicators covered in this class are those that assess biological quality elements such as composition, abundance and biomass of flora and fauna. Although subclasses such as ‘Conservation Status’ and ‘Invasive Alien Species’ could also be considered as biodiversity indicators, they were included in the classes state and pressure, respectively, as they provide information about ecological status and anthropogenic pressures. Most of the publications identify common biodiversity indicators applicable across different ecosystems, while some develop or apply indices that are specific for an ecosystem, region or country (e.g. Pascoal et al. 2003; Breine et al. 2007). Diversity descriptors such as species richness, abundance, composition, density and frequency, as well as evenness, and population size were commonly measured in the studies. Other aspects that were assessed to a lesser degree were living and feeding habits and mobility (Törnroos et al. 2015), trophic levels (Jayasinghe et al. 2015) and adult life habitat (Marchini et al. 2008).

4. Discussion

This literature review shows that the assessment of ecosystem condition has gained more importance in Europe in recent years. The results show the current state, trends and gaps in the application of the concept of ecosystem condition. Although this analysis was limited to scientific publications and national and international reports of studies conducted in Europe, the results show which ecosystem types are currently being evaluated with the different methods that are available. This analysis provides an overview of the main characteristics of ecosystem condition assessments and highlights that the term is being used to portray the general state of an ecosystem based on general descriptors (including biodiversity, environmental pressures and states). However, these assessments do not analyse the functional characteristics determined by the physical, chemical and biological quality of the ecosystems that underpins particular ecosystem services. In other words, these assessments do not analyse ecosystem *capacity*,[9] but are based on the potential capacity usually estimated by ecosystem extent only. This shows that there is an important knowledge gap regarding the understanding of the relationship between the condition of ecosystems (including biodiversity) and their capacity to deliver ecosystem services. Bridging this gap could help to assess to what extent ecosystem services can be maintained and potentially be enhanced.

4.1. Categorization system for ecosystem condition components

The lack of an appropriate categorisation system for the individual ecosystem condition components, for example, a system that would be comparable to comprehensive ecosystem services classification systems such as CICES, was a relevant issue. Such a system could, for instance, build on existing concepts like the essential biodiversity variables or ecological integrity indicators (Haase et al. 2018). For this review, preliminary classes suggested by the EU MAES Working Group (European Commission, European Environment Agency, Joint Research Centre, European Topic Centre for Biodiversity, European Topic Centre Urban Land Use Systems 2017) were used because suitable criteria to group the reviewed studies and used indicators were needed. The chosen criteria, classes and subclasses proved to be functional, although overlaps were obvious in some of the defined classes. For instance, the pressure indicator subclass 'Human disturbance' is rather general and would include almost all other subclasses, such as 'Mining', 'Natural system modifications' and 'Fragmentation'. Besides EU MAES, also the SEEA EEA[10] (United Nations et al. 2014) has set up a process to revise its technical recommendations for developing ecosystem accounts. The revision of ecosystem condition accounts will include a proposal for a classification of ecosystem condition indicators. It was, however, not the aim of this review study to come up with a comprehensive ecosystem condition categorisation system.

4.2. Ecosystems and geographical coverage

Results show that ecosystem condition assessments have been well-implemented on regional and local scales, supporting the compliance of the European countries with the actions proposed in the EU Biodiversity Strategy to 2020 and other environmental strategies and directives. The reviewed studies show that ecosystem condition has been most commonly assessed in rivers and lakes since 2003 and marine inlets and transitional waters since 2008. This indicates that most European Union member states have been working on the implementation of the WFD since it was adopted in 2000 (European Parliament & Council of the European Union 2000) and later on the MSFD which was adopted in 2008 (European Parliament & Council of the European Union 2008). However, case studies on other ecosystem types such as woodland and forest have gained more importance since 2011, probably after the adoption of the Birds Directive in 2009 (European Parliament & Council of the European Union 2010) and the Biodiversity Strategy in 2011 (European Commission 2011). Most of the studies on the three evaluated types of ecosystem categories (terrestrial, freshwater and marine) used the terms ecological, environmental or conservation status to refer to their condition, which is also in line with the terminology used in the aforementioned directives.

There seems to be a significant tendency in the geographical coverage of regions and ecosystems, with 61 out of the 401 studies identified in the review being located in Spain. Followed by Italy (43), Portugal and Europe as a whole (39), France and Germany (33). Hungary and Slovakia have only one study each, and some countries do not have any study as mentioned in Section 3.2. The relatively high amount of studies in Spain might be associated with the increasing research on the environmental condition of marine and freshwater ecosystems conducted in this country where water is a limited resource of great social relevance and importance. This tendency can also be seen when looking at the distribution of ecosystem types assessed across the countries. Thirteen percent of the rivers and lakes and 14% of marine inlets and transitional waters, and coasts of Europe identified in the literature are located in Spain. In addition, the large number of case studies assessing rivers and lakes can be related to the typical format of the studies, which tend to be experimental analyses at particular sites, like sampling locations along waterbodies. In this sense, it is important to highlight that freshwater ecosystems were most commonly assessed on a regional scale, where many of the studies were conducted at the level of a catchment, similar to marine inlets and transitional waters, and woodlands and forests. This tendency in the geographical coverage of regions and ecosystems shows the need for collaboration between countries to develop more assessments and maps of the condition and services provided by the various European ecosystems.

4.3. Mapping ecosystem condition

Only around 20% of the reviewed case studies included ecosystem condition mapping. This relatively low number can probably be explained by the fact that most of the studies were not intended to present the results in the form of maps, as it is not requested in the environmental directives mentioned before or mapping is not within the scope of the disciplines involved in the studies. Another reason can be the insufficiency and/or inadequacy of data or lack of technical capacity. Additionally, even though the pressures on an ecosystem may be known, the combined effects on its functioning and condition are often not well understood. This poses some difficulties when mapping ecosystem condition, especially in the selection of suitable spatially explicit indicators that reflect the actual condition of an ecosystem (Erhard et al. 2017). Despite the great progress that has been made in mapping ecosystem services (Maes and Burkhard 2017), which is often based on the geographical distribution of ecosystems, more research on the relationship between ecosystem structures, processes and pressures is still needed to produce robust and reliable maps on ecosystem condition.

4.4. Methods used to assess ecosystem condition

There has been an increase in the variety and frequency of methods to assess ecosystem condition since 2005, which correspond to the adoption of the environmental directives in the EU and the increase in scientific publications. The most traditional methods such as biological, physical and chemical approaches have been broadly used since 2002, which are in line with the requirements of these directives. However, in recent years, GIS methods, models and scenarios have been gaining more importance in the estimation of ecosystem condition, based on information about the spatial distribution and heterogeneity of pressures and current and future state of ecosystems. Schröder et al. (2015), for instance, developed a spatial explicit methodology for evaluating the integrity of forests by comparing current, future and reference states. Another example is the study of Tecchio et al. (2016) that assessed the pressures of the extension of a harbour on an estuarine ecosystem based on food web models. On the contrary, participatory methods, such as surveys, interviews or workshops have not been broadly used in ecosystem condition research, probably because these methods are more frequently used to assess ecosystem services than condition. This review shows that mostly monodisciplinary approaches are used, focusing on the biophysical characteristics of ecosystems. Yet it is necessary to develop interdisciplinary approaches that look at the dynamics of social-ecological systems, the multiple ecosystem service provision, service bundles, synergies and trade-offs to more thoroughly assess ecosystem condition and its links with ecosystem services and human well-being.

4.5. Assessment of ecosystem services

The literature assessing ecosystem condition, functioning, structure and services together is limited (less than 18% of the studies assess ecosystem services), which can be related to the fact that the analysis of the relationship between ecosystem condition and ecosystem services is a rather complex endeavour and a process still in an initial phase (Maes et al. 2018). The number of publications mentioning or alluding to the term ecosystem services, increased after 2005 presumably due to the release of major reports such as the Millennium Ecosystem Assessment (2005), the TEEB (2010) and CICES (since 2009) that have helped to draw more attention to this topic. As well as other publications such as from De Groot et al. (2010) that proposes some criteria and indicators to describe the interactions between ecological processes and ecosystem services. These findings also coincide with the findings of McDonough et al. (2017) who reported that the number of studies on ecosystem services have increased from less than 500 in 2005 to approximately 3000 in 2016. Less than half of the reviewed publications that mention ecosystem services also assess them, and the majority of these assessments were conducted after the release of the EU Biodiversity Strategy to 2020 in 2011. The MA classification of ecosystem services was the most commonly used in these studies, probably because it was the first to be published and the best known classification. However, different classifications of ecosystem services are being used depending on the interests and priorities of the countries. The CICES classification, for example, is becoming more popular because it is linked to the System of Environmental Economic Accounts (SEEA) of the UN and facilitates the integration into accounting and reporting systems.

The focus on different groups or types of ecosystem services has also been associated with research needs that support the development of specific policies. One example is the assessment of climate regulation-related ecosystem services. Most of the studies that assess ecosystem services such as *regulation of chemical composition of the atmosphere and oceans* (2.2.6.1) as well as *regulation of temperature and humidity, including ventilation and transpiration* (2.2.6.2) were carried out after 2010. This can be linked to the publication of the fourth assessment report of the Intergovernmental Panel on Climate Change (IPCC[11]) and the development of associated policies such as the UN-REDD Programme.[12] Such initiatives have resulted in an increase of research in climate change which has become a priority for many governments and international bodies.

4.6. The use of ecosystem attributes as indicators of ecosystem condition

There is a diverse set of attributes of ecosystems, including biotic, abiotic and socioeconomic variables that are used as indicators of ecosystem condition. Although these attributes are not often linked with the ecosystem services that depend on them, some links do exist. Various publications that assess ecosystem services describe the relationships between these attributes and the provision of specific services. Some examples are species diversity, habitat area, nutrient cycling, and age structure and/or diameter distribution of forests. These attributes tend to be associated with the provision of biomass in the form of cultivated and wild plants, and fibres. Other aspects such as primary production, nutrients uptake, native and alien species abundance and distribution are mostly related to the regulation of physical, chemical and biological conditions. Braun et al. (2017) for instance, assessed the relationship between gross primary production and the provision of food and the regulation of carbon. Potts et al. (2014) analysed the relative importance of habitats and species diversity in the provision of ecosystem services such as regulation of water and sediment quality in marine areas. Nutrients uptake through primary production and burial of organic matter work as proxies of the freshwater purification service because they constitute efficient nutrient removal processes (Liquete et al. 2016b). Attributes such as attractive landscape features, variety in landscapes with recreational uses and variety in natural features with cultural values are mostly related to the provision of cultural ecosystem services (Kienast et al. 2009). Other attributes include accessibility to suitable recreation areas, and spiritual and inspirational properties, but sometimes there are trade-offs between them.

Although most of the indicators identified in the literature provide a description of ecosystem condition, some of them are based on a specific parameter or attribute, which are sometimes insufficient to support decisions. These indicators result from the great amount of methods to assess the condition of a particular component of an ecosystem. However, methods that holistically assess multiple components and physical, chemical and biological aspects of an ecosystem are still lacking, as well as the definition of adequate reference conditions (Borja 2014). There are only a few methods, especially for the assessment of marine ecosystems, that integrate the principal processes and structural characteristics of the social-ecological system and the pressures and responses of the system to such pressures (Borja et al. 2016). The development of such holistic methods would contribute to formulating a more limited number of indicators that are easier to update and to portraying the most general aspects of the current and future ecosystem condition. These indicators, accompanied by additional background information about the drivers and pressures that affect ecosystem condition, would facilitate a better understanding of the social-ecological systems. Furthermore, the analysis of synergies and trade-offs between services and the required conditions for their supply provides quantitative tools to make informed decisions at different scales.

5. Conclusions

The number of literature on mapping and assessment of ecosystem condition in Europe is increasing. This review focused on assessments in published scientific papers and national and international reports in English, excluding unpublished documents or publications in other languages or outside of Europe. Only a few studies identified and described links between ecosystem condition and ecosystem services. Despite this weak link, ecosystem services were most commonly related to ecosystem condition in studies that assess ecosystems for which there is more knowledge on the services they provide such as rivers and lakes or woodlands and forests. The main indicators identified in the literature for the assessment of ecosystem condition have been confirmed in the 5th MAES report (Maes et al. 2018) and constitute a good starting point for integrated ecosystem assessments. However, in order to be policy-relevant, some of these indicators require additional contextual information. For example, awareness of synergies and trade-offs between ecosystem services and a better understanding of the functional relationships between ecosystem condition and service delivery, including biodiversity, are highly relevant for informed decision-making. Contextual information also covers the characteristics of the ecosystems, the causes of pressures on the ecosystem, and beneficiaries of ecosystems services. This information helps to understand the drivers and pressures that affect the condition of the systems of interest and to make better decision for the optimization of long-term ecosystem service delivery.

Most of the current methods for assessing ecosystem condition are monodisciplinary and focus solely on one environmental attribute. Based on this review, the authors suggest that more multidisciplinary and holistic approaches should become available together with a comprehensive categorisation system to determine the condition of an ecosystem and its influence on the provision of ecosystem services. These approaches, combined with assessments of the effects of socioeconomic factors and land/sea use decisions, could provide more robust information that helps enhance the implementation of more adequate policy measures to protect our environment and guarantee the sustainable provision of ecosystem services from functioning and diverse ecosystems. This could lead to a switch from management approaches that degrade nature and biodiversity in order to maximise one ecosystem service at the expense of others, towards approaches that create multifunctional, healthy and sustainable landscapes.

5.1. Applications for ongoing policy-making processes

In the EU but also globally, the assessment of ecosystem condition is gaining increasing attention from biodiversity policy. The quantification and assessment of ecosystem condition is an essential component of the ecosystem accounts of the UN statistical division's SEEA framework. The current System of Environmental-Economic Accounting (United Nations, European Commission, Food and Agricultural Organization of the United Nations, International Monetary Fund, Organization for Economic Co-operation and Development, The World Bank 2014) is now under revision with a view to develop a statistical standard. In these accounts, ecosystem condition takes a central position between ecosystem extent accounts and ecosystem service accounts and will help understand whether and how ecosystems are being degraded. Such knowledge is crucially important, for instance, to support an ecosystem restoration agenda, which is required under the Convention of Biological Diversity. So the mapping and assessment of ecosystem condition for different ecosystem types at different spatial scales can deliver essential information of where to restore ecosystems and help set priorities for restoration financing and activities at multiple levels of governance. The results of this review study provide an overview of existing ecosystem condition studies and can help to identify research gaps and support priority setting of future research efforts.

Notes

1. Ecosystem unit is an ecosystem type within a basic spatial unit; see Czúcz and Condé (2017) for more detailed definition.
2. Mapping and Assessment of Ecosystems and their Services. Available in: http://biodiversity.europa.eu/maes Accessed on 14–12-2018.
3. UK National Ecosystem Assessment. Available in: http://uknea.unep-wcmc.org/ Accessed on 10–08-2017.
4. Ecosystem for Human Well-being. Evaluación de ecosistemas del Milenio de España. Available in http://www.ecomilenio.es/Accessed on 10.08.2017.
5. CICES Towards a common classification of ecosystem services. Available in: www.cices.eu Accessed on 14.12.2018.
6. Classifying Ecosystem Services. The BBN Ecosystem Service Classification Tool. Available in: http://openness.hugin.com/example/cices Accessed on 14.12.2018.
7. Based on the DPSIR model promoted by the EU. Available in: https://www.eea.europa.eu/publications/TEC25 Accessed on 14.12.2018.
8. No studies were found for Albania, Andorra, Armenia, Azerbaijan, Belarus, Bosnia and Herzegovina, Georgia, Iceland, Kazakhstan, Kosovo, Latvia, Liechtenstein, Luxembourg, Macedonia, Moldova, Monaco, Montenegro, Russia, San Marino, Serbia, Turkey, Ukraine and Vatican City.
9. For respective definitions see Potschin-Young et al. (2018).
10. SEEA EEA: System of Environmental-Economic Accounting Experimental Ecosystem Accounting. Available in: https://unstats.un.org/unsd/envaccounting/eea_project/default.asp Accessed on 14.03.2019.
11. IPCC – Intergovernmental Panel on Climate Change. Available in: http://www.ipcc.ch/ Accessed on 18–01-2018.
12. UN-REDD Programme. Available in: http://www.un-redd.org/ Accessed on 18.01.2018.

Acknowledgments

The authors would like to thank the anonymous reviewers for their feedback which helped to improve the quality of the paper. We thank Angie Faust for double-checking the English language.

Disclosure statement

No potential conflict of interest was reported by the authors.

Funding

This work was supported by the ESMERALDA - European Union Horizon 2020 programme [Grant Agreement Number: 642007].

References

Aarts BGW, Nienhuis PH. 2003. Fish zonations and guilds as the basis for assessment of ecological integrity of large rivers. Hydrobiologia. 500:157–178. doi:10.1023/A:1024638726162.

Balvanera P, Siddique I, Dee L, Paquette A, Isbell F, Gonzalez A, Byrnes J, O'Connor MI, Hungate BA, Griffin JN. 2014. Linking biodiversity and ecosystem services: current uncertainties and the necessary next steps. Bioscience. 64:49–57. doi:10.1093/biosci/bit003.

Bastian O. 2013. The role of biodiversity in supporting ecosystem services in Natura 2000 sites. Ecol Indic. 24:12–22. doi:10.1016/j.ecolind.2012.05.016.

Borja Á. 2014. Grand challenges in marine ecosystems ecology. Front Mar Sci. 1:1–6. doi:10.3389/fmars.2014.00001.

Borja Á, Chust G, Del Campo A, González M, Hernández C. 2013. Setting the maximum ecological potential of benthic communities, to assess ecological status, in heavily morphologically-modified estuarine water bodies. Mar Pollut Bull. 71:199–2018. doi:10.1016/j.marpolbul.2013.03.014.

Borja Á, Elliott M, Andersen JH, Berg T, Carstensen J, Halpern BS, Heiskanen A-S KS. 2016. Overview of integrative assessment of marine systems : the ecosystem approach in practice. Front Mar Sci. 3:1–20. doi:10.3389/fmars.2016.00020.

Borja Á, Galparsoro I, Irigoien X, Iriondo A, Menchaca I, Muxika I, Pascual M, Quincoces I, Revilla M, Rodríguez JG, et al. 2011. Implementation of the European marine strategy framework directive: A methodological approach for the assessment of environmental status, from the Basque Country (Bay of Biscay). Mar Pollut Bull. 62:889–904. doi:10.1016/j.marpolbul.2010.11.026.

Brämick U, Diekmann M, Lemcke R, Mehner T. 2008. Assessing shifts in fish assemblages of German large lakes by literature data and commercial catch statistics. Fundam Appl Limnol/Arch Für Hydrobiol. 171:87–103. doi:10.1127/1863-9135/2008/0171-0087.

Braun D, Damm A, Paul-Limoges E, Revill A, Buchmann N, Petchey OL, Hein L, Schaepman ME. 2017. From instantaneous to continuous: Using imaging spectroscopy and in situ data to map two productivity-related ecosystem services. Ecol Indic. 82:409–419. doi:10.1016/j.ecolind.2017.06.045

Breine J, Maes J, Quataert P, van Den Bergh E, Simoens I, van Thuyne G, Belpaire C. 2007. A fish-based assessment tool for the ecological quality of the brackish schelde estuary in Flanders (Belgium). Hydrobiologia. 575:141–159. doi:10.1007/s10750-006-0357-z.

Brkić Ž, Larva O, Urumović K. 2010. The quantitative status of groundwater in alluvial aquifers in norther Croatia. Geol Croat. 63:283–298. doi:10.4154/GC.2010.23.

Burkhard B, Müller F, Lill A. 2008. Ecosystem Health Indicators, In: Jørgensen SE, Fath BD, editors. Ecological Indicators. Vol. 2. Oxord Elsevier; p. 1132–1138.

Burkhard B, Santos-Martín F, Nedkov S, Maes J. 2018. An operational framework for integrated mapping and assessment of ecosystems and their Services (MAES). One Ecosyst. 3:e22831. doi:10.3897/oneeco.3.e22831.

Corsi I, Mariottini M, Sensini C, Lancini L, Focardi S. 2003. Fish as bioindicators of brackish ecosystem health : integrating biomarker responses and target pollutant concentrations Les poissons, bioindicateurs de la santé d ' écosystèmes saumâtres : réponses intégrées de biomarqueurs et concentrations de polluant. Oceanol Acta. 26:129–138. doi:10.1016/S0399-1784(02)01237-9.

Costanza R, Norton BG, Haskell BD. 1992. Ecosystem health: new goals for environmental management. Washington D.C: Island Press.

Czúcz B, Condé S 2017. Note on definitions related to ecosystem conditions and their services based on different glossaries. https://bd.eionet.europa.eu/Reports/ETCBDTechnicalWorkingpapers/PDF/Definitions_ecosystem_conditions_and_services.pdf.

De Groot RS, Alkemade R, Braat L, Hein L, Willemen L. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol Complex. 7:260–272. doi:10.1016/j.ecocom.2009.10.006.

Erhard M, Banko G, Abdul Malak D, Santos-Martín F. 2017. Mapping ecosystem types and conditions. In: Burkhard B, Maes J, editors. Mapp Ecosyst Serv. Sofia (Bulgaria): Pensoft Publishers; p. 75–80.

Erhard M, Teller A, Maes J, Meiner A, Berry PM, Smith A, Eales R, Papadopoulou L, Bastrup-Birk A, Ivits É, et al. 2016. Mapping and assessment of ecosystems and their services: mapping and assessing the condition of Europe's ecosystems: progress and challenges. http://ec.europa.eu/environment/nature/knowledge/ecosystem_assessment/pdf/3rdMAESReport_Condition.pdf.

European Commission. 2011. Our life insurance, our natural capital: an EU biodiversity strategy to 2020. http://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/EP_resolution_april2012.pdf.

European Commission, European Environment Agency, Joint Research Centre, European Topic Centre for Biodiversity, European Topic Centre Urban Land Use Systems. 2017. MAES Workshop Assessing and Mapping Ecosystem Condition. Background Paper to support breakout group discussions version of 2017 Jul 11; Brussels, Belgium. Available from: https://circabc.europa.eu/sd/a/a5932b69-ffa5-4d5f-a842-ead4bb8ba623/MAESconditionsBackgroundpaper11July.pdf

European Environment Agency. 2010. The European environment — state and outlook 2010: land use. Copenhagen. https://www.eea.europa.eu/soer/europe/land-use.

European Environment Agency. 2015. State of Europe's seas. https://www.eea.europa.eu/publications/state-of-europes-seas.

European Parliament, Council of the European Union. 2000. Directive 2000/60/EC of the European parliament and of the council of 23 October 2000 establishing a framework for community action in the field of water policy. Off J Eur Parliam. Available at. L327. 1–82. https://eur-lex.europa.eu/resource.html?uri=cellar:5c835afb-2ec6-4577-bdf8-756d3d694eeb.0004.02/DOC_1&format=PDF.

European Parliament, Council of the European Union. 2008. Directive 2008/56/EC of the European parliament and of the council. Off J Eur Union. Available at. 164:19–40. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32008L0056&from=EN.

European Parliament, Council of the European Union. 2010. Directive 2009/147/EC of the European parliament and of the council on the conservation of wild birds. Off J Eur Union. Available at. L 20. 7–25. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32009L0147&from=EN.

Garnier J, Billen G. 2007. Production vs. Respiration in river systems: an indicator of an "ecological status". Sci Total Environ. 375:110–124. doi:10.1016/j.scitotenv.2006.12.006.

Gobert S, Sartoretto S, Rico-Raimondino V, Andral B, Chery A, Lejeune P, Boissery P. 2009. Assessment of the ecological status of Mediterranean French coastal waters as required by the water framework directive using the Posidonia oceanica rapid easy index: PREI. Mar Pollut Bull. 58:1727–1733. doi:10.1016/j.marpolbul.2009.06.012.

Grechi L, Franco A, Palmeri L, Pivato A, Barausse A. 2016. An ecosystem model of the lower Po river for use in ecological risk assessment of xenobiotics. Ecol Modell. 332:42–58. doi:10.1016/j.ecolmodel.2016.03.008.

Haase P, Tonkin JD, Stoll S, Burkhard B, Frenzel M, Geijzendorffer I, Häuser C, Klotz S, Kühn I, McDowell WH, et al. 2018. The next generation of site-based long-term ecological monitoring: linking essential biodiversity variables and ecosystem integrity. Sci Total Environ. 613–614:1376–1384. doi:10.1016/j.scitotenv.2017.08.111.

Haines-Young R, Potschin-Young M 2018. Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the

Revised Structure. https://seea.un.org/sites/seea.un.org/files/lg23_cices_v5.1_final_revised_guidance_03-10-2017.pdf.

Harrington R, Anton C, Dawson TP, de Bello F, Feld C, Haslett JR, Kluvánkova-Oravská T, Kontogianni A, Lavorel S, Luck GW, et al. 2010. Ecosystem services and biodiversity conservation: concepts and a glossary. Biodivers Conserv. 19:2773–2790. doi:10.1007/s10531-010-9834-9.

Harrison P, Berry PM, Simpson G, Haslett JR, Blicharska M, Bucur M, Dunford R, Egoh B, Garcia-Llorente M, Geamănă N, et al. 2014. Linkages between biodiversity attributes and ecosystem services: A systematic review. Ecosyst Serv. 9:191–203. doi:10.1016/j.ecoser.2014.05.006.

HELCOM. 2010. Ecosystem Health of the Baltic Sea. Erweko: HELCOM Initial Holistic Assessment. http://www.helcom.fi/Lists/Publications/BSEP122.pdf.

Herrero J, Castañeda C. 2009. Delineation and functional status monitoring in small saline wetlands of NE Spain. J Environ Manage. 90:2212–2218. doi:10.1016/j.jenvman.2007.06.026.

Ioannou A, Chatzinikolaou Y, Lazaridou M. 2009. A preliminary pressure-impact analysis applied in the Pinios river basin (Thessaly, Central Greece). Water Environ J. 23:200–209. doi:10.1111/j.1747-6593.2008.00130.x.

Ippolito A, Sala S, Faber JH, Vighi M. 2010. Ecological vulnerability analysis: A river basin case study. Sci Total Environ. 408:3880–3890. doi:10.1016/j.scitotenv.2009.10.002.

Jayasinghe RPPK, Amarasinghe US, Newton A. 2015. Evaluation of marine subareas of Europe using life history parameters and trophic levels of selected fish populations. Mar Environ Res. 112:81–90. doi:10.1016/j.marenvres.2015.08.002.

Juanes JA, Guinda X, Puente A, Revilla JA. 2008. Macroalgae, a suitable indicator of the ecological status of coastal rocky communities in the NE Atlantic. Ecol Indic. 8:351–359. doi:10.1016/j.ecolind.2007.04.005.

Kahmen A, Perner J, Audorff V, Weisser W. 2005. Effects of plant diversity, community composition and environmental parameters on productivity in Montane European Grasslands. Oecologia. 142:606–615. doi:10.1007/s00442-004-1749-2.

Kenny A, Skjoldal HR, Engelhard GH, Kershaw PJ, Reid JB. 2009. An integrated approach for assessing the relative significance of human pressures and environmental forcing on the status of Large Marine Ecosystems. Prog Oceanogr. 81:132–148. doi:10.1016/j.pocean.2009.04.007.

Kienast F, Bolliger J, Potschin-Young M, de Groot RS, Verburg PH, Heller I, Wascher D, Haines-Young R. 2009. Assessing landscape functions with broad-scale environmental data: insights gained from a prototype development for Europe. Environ Manage. 44:1099–1120. doi:10.1007/s00267-009-9384-7.

Korpinen S, Meidinger M, Laamanen M. 2013. Cumulative impacts on seabed habitats: an indicator for assessments of good environmental status. Mar Pollut Bull. 74:311–319. doi:10.1016/j.marpolbul.2013.06.036.

Lautenbach S, Kugel C, Lausch A, Seppelt R. 2011. Analysis of historic changes in regional ecosystem service provisioning using land use data. Ecol Indic. 11:676–687. doi:10.1016/j.ecolind.2010.09.007.

Liquete C, Cid N, Lanzanova D, Grizzetti B, Reynaud A. 2016a. Perspectives on the link between ecosystem services and biodiversity: the assessment of the nursery function. Ecol Indic. 63:249–257. doi:10.1016/j.ecolind.2015.11.058.

Liquete C, Piroddi C, Macías D, Druon J-N ZG. 2016b. Ecosystem services sustainability in the Mediterranean Sea: assessment of status and trends using multiple modelling approaches. Sci Rep. 6:34162. doi:10.1038/srep34162.

MA. 2005. Millennium ecosystem assessment synthesis report. Washington (DC): Island press.

Maceda-Veiga A, Green A, de Sostoa A. 2014. Scaled body-mass index shows how habitat quality influences the condition of four fish taxa in north-eastern Spain and provides a novel indicator of ecosystem health. Freshw Biol. 59:1145–1160. doi:10.1111/fwb.12336.

Maes J, Burkhard B. 2017. Conclusions. In: Burkhard B, Maes J, editors. Mapp Ecosyst Serv. Sofia (Bulgaria): Pensoft Publishers; p. 363–365.

Maes J, Teller A, Erhard M, Grizzetti B, Barredo JI, Paracchini ML, Condé S, Somma F, Orgiazzi A, Jones A, et al. 2018. Mapping and assessment of ecosystems and their services: an analytical framework for ecosystem condition. Luxembourg. http://ec.europa.eu/environment/nature/knowledge/ecosystem_assessment/pdf/5thMAESreport.pdf.

Maes J, Teller A, Erhard M, Liquete C, Braat L, Berry PM, Egoh B, Puydarrieux P, Fiorina C, Santos-Martín F 2014. Mapping and assessment of ecosystems and their services: indicators for ecosystem assessments under action 5 of the EU biodiversity strategy to 2020. http://www.naturvardsverket.se/Nerladdningssida/?fileType=pdf&downloadUrl=/Documents/publikationer6400/978-91-620-6626-0.pdf.

Maes J, Teller A, Erhard M, Liquete C, Braat L, Egoh B, Puydarrieux P, Fiorina C, Santos-Martín F, Paracchini ML, et al. 2013. Mapping and assessment of ecosystems and their services: an analytical framework for ecosystem assessments under action 5 of the EU biodiversity strategy to 2020. http://ec.europa.eu/environment/nature/knowledge/ecosystem_assessment/pdf/MAESWorkingPaper2013.pdf.

Marchini A, Munari C, Mistri M. 2008. Functions and ecological status of eight Italian lagoons examined using biological traits analysis (BTA). Mar Pollut Bull. 56:1076–1085. doi:10.1016/j.marpolbul.2008.03.027.

Maseyk FJF, Mackay AD, Possingham HP, Dominati EJ, Buckley YM. 2017. Managing natural capital stocks for the provision of ecosystem services. Conserv Lett. 10:211–220. doi:10.1111/conl.12242.

McDonough K, Hutchinson S, Moore T, Hutchinson JMS. 2017. Analysis of publication trends in ecosystem services research. Ecosyst Serv. 25:82–88. Available at. doi:10.1016/j.ecoser.2017.03.022.

Moher D, Liberati A, Tetzlaff J, Altman DG. 2009. Preferred Reporting Items for Systematic Reviews and Meta-Analyses: the PRISMA Statement (Reprinted from Annals of Internal Medicine). Phys Ther. 89:873–880.

Mulder C, Boit A, Bonkowski M, de Ruiter PC, Mancinelli G, van der Heijden MG, van Wijnen HJ, Vonk JA, Rutgers M. 2011. A belowground perspective on dutch agroecosystems: how soil organisms interact to support ecosystem services. Adv Ecol Res. 44:277–357.

Müller F, Burkhard B, Kroll F. 2010. Resilience, integrity and ecosystem dynamics: bridging ecosystem theory and management. In: Landform-structure, process control. Vol. 115. Berlin (BE): Springer; p. 221–242.

Müller F, Schrautzer J, Reiche EW, Rinker A. 2006. Ecosystem based indicators in retrogressive successions of an agricultural landscape. Ecol Indic. 6:36–82. doi:10.1016/j.ecolind.2005.08.017.

O'Brien A, Townsend K, Hale R, Sharley D, Pettigrove V. 2016. How is ecosystem health defined and measured? A critical review of freshwater and estuarine studies. Ecol Indic. 69:722–729. doi:10.1016/j.ecolind.2016.05.004.

Pascoal C, Pinho M, Cássio F, Gomes P. 2003. Assessing structural and functional ecosystem condition using leaf breakdown: studies on a polluted river. Freshw Biol. 48:2033–2044. doi:10.1046/j.1365-2427.2003.01130.x.

Pinto R, de Jonge VN, Marques JC. 2014. Linking biodiversity indicators, ecosystem functioning, provision of services and human well-being in estuarine systems: application of a conceptual framework. Ecol Indic. 36:644–655. doi:10.1016/j.ecolind.2013.09.015.

Piroddi C, Moutopoulos DK, Gonzalvo J, Libralato S. 2016. Ecosystem health of a mediterranean semi-enclosed embayment (Amvrakikos Gulf, Greece): assessing changes using a modeling approach. Cont Shelf Res. 121:61–73. Available at. doi:10.1016/j.csr.2015.10.007.

Potschin-Young M, Burkhard B, Czúcz B, Santos-Martín F. 2018. Glossary of ecosystem services mapping and assessment terminology. One Ecosyst. 3:e27110. doi:10.3897/oneeco.3.e27110.

Potschin-Young M, Czúcz B, Liquete C, Maes J, Rusch G, Haines-Young R. 2017. Intermediate and supporting ecosystem services: empty concepts? Ecosyst Serv. 27:124–126. doi:10.1016/j.ecoser.2017.09.001.

Potschin-Young M, Haines-Young R, Heink U, Jax K 2016. OpenNESS Glossary. http://www.openness-project.eu/glossary.

Potts T, Burdon D, Jackson E, Atkins J, Saunders J, Hastings E,Langmead O. 2014. Do marine protected areas deliver flows of ecosystem services to support human welfare? Mar Policy. 44:139–148. doi:10.1016/j.marpol.2013.08.011

Putkuri E, Lindholm M, Peltonen A 2013. State of the environment in Finland 2013. Helsinki. https://helda.helsinki.fi/handle/10138/42691.

Roig N, Sierra J, Nadal M, Moreno-Garrido I, Nieto E, Hampel M, Gallego EP, Schuhmacher M, Blasco J. 2015. Assessment of sediment ecotoxicological status as a complementary tool for the evaluation of surface water quality: the Ebro river basin case study. Sci Total Environ. 503–504:269–278. doi:10.1016/j.scitotenv.2014.06.125.

Rova S, Pranovi F, Müller F. 2015. Provision of ecosystem services in the lagoon of Venice (Italy): an initial spatial assessment. Ecohydrol Hydrobiol. 15:13–25. doi:10.1016/j.ecohyd.2014.12.001.

Schneiders A, van Daele T, Landuyt VW, Van Reeth W. 2012. Biodiversity and ecosystem services: complementary approaches for ecosystem management? Ecol Indic. 21:123–133. doi:10.1016/j.ecolind.2011.06.021.

Schröder W, Nickel S, Jenssen M, Riediger J. 2015. Methodology to assess and map the potential development of forest ecosystems exposed to climate change and atmospheric nitrogen deposition: A pilot study in Germany. Sci Total Environ. 521–522:108–122. doi:10.1016/j.scitotenv.2015.03.048.

Smith A, Harrison P, Pérez-Soba M, Archaux F, Blicharska M, Egoh B, Erős T, Fabrega Domenech N, György Á, Haines-Young R, et al. 2017. How natural capital delivers ecosystem services: A typology derived from a systematic review. Ecosyst Serv. 26:111–126. doi:10.1016/j.ecoser.2017.06.006.

Spanish National Ecosystem Assessment. 2014. Ecosystems and biodiversity for human wellbeing. Synthesis of the Key findings. Madrid. http://www.ecomilenio.es/ecosystems-and-biodiversity-for-human-wellbeing-snea-synthesis-of-key-findings-download/3661.

Stein H, Kellermann C, Schmidt SI, Brielmann H, Steube C, Berkhoff SE, Fuchs A, Hahn HJ, Thulin B, Griebler C. 2010. The potential use of fauna and bacteria as ecological indicators for the assessment of groundwater quality. J Environ Monit. 12:242–254. doi:10.1039/c0em00160k.

Stevens M, Demolder H, Jacobs S, Michels H, Schneiders A, Simoens I, Spanhove T, van Gossum P, van Reeth W, Peymen J 2015. Flanders Regional Ecosystem Assessment - State and Trends. Synthesis Report. Brussels. Available from: https://pureportal.inbo.be/portal/files/9004761/Stevens_etal_2015_FlandersRegionalEcosystemAssessment_State_Trends.pdf

Sutela T, Aroviita J, Keto A. 2013. Assessing ecological status of regulated lakes with littoral macrophyte, macroinvertebrate and fish assemblages. Ecol Indic. 24:185–192. doi:10.1016/j.ecolind.2012.06.015.

Tecchio S, Chaalali A, Raoux A, Tous Rius A, Lequesne J, Girardin V, Lassalle G, Cachera M, Riou P, Lobry J, et al. 2016. Evaluating ecosystem-level anthropogenic impacts in a stressed transitional environment: the case of the Seine estuary. Ecol Indic. 61:833–845. doi:10.1016/j.ecolind.2015.10.036.

Tecchio S, Rius AT, Dauvin JC, Lobry J, Lassalle G, Morin J, Bacq N, Cachera M, Chaalali A, Villanueva MC, et al. 2015. The mosaic of habitats of the Seine estuary: insights from food-web modelling and network analysis. Ecol Modell. 312:91–101. doi:10.1016/j.ecolmodel.2015.05.026.

TEEB. 2010. The economics of ecosystems and biodiversity: mainstreaming the economics of nature: A synthesis of the approach, conclusions and recommendations of TEEB. http://www.teebweb.org/publication/mainstreaming-the-economics-of-nature-a-synthesis-of-the-approach-conclusions-and-recommendations-of-teeb/.

Törnroos A, Bonsdorff E, Bremner J, Blomqvist M, Josefson AB, Garcia C, Warzocha J. 2015. Marine benthic ecological functioning over decreasing taxonomic richness. J Sea Res. 98:49–56. doi:10.1016/j.seares.2014.04.010.

UK National Ecosystem Assessment. 2011. The UK national ecosystem assessment: synthesis of the key findings. Oxford. http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx.

United Nations, European Commission, Food and Agricultural Organization of the United Nations, International Monetary Fund, Organization for Economic Co-operation and Development, The World Bank. 2014. System of environmental-economic accounting 2012 - Central framework. New York.

von der Ohe P, de Deckere E, Prüß A, Muñoz I, Wolfram G, Villagrasa M, Ginebreda A, Hein M, Brackà W. 2009. Toward an integrated assessment of the ecological and chemical status of european river basins. Integr Environ Assess Manag. 5:50–61. doi:10.1897/IEAM_2008-043.1.

Young TF, Sanzone S 2002. A Framework for Assessing and Reporting on Ecological Condition: An SAB Report. https://yosemite.epa.gov/sab/sabproduct.nsf/7700D7673673CE83852570CA0075458A/$File/epec02009.pdf.

3

Assessment of the relationships between agroecosystem condition and the ecosystem service soil erosion regulation in Northern Germany

Rendon, P., Steinhoff-Knopp, B., Saggau, P., & Burkhard, B.

PLOS One (2020), 15(12): e0234288

PLOS ONE

RESEARCH ARTICLE

Assessment of the relationships between agroecosystem condition and the ecosystem service soil erosion regulation in Northern Germany

Paula Rendon[1]*, **Bastian Steinhoff-Knopp**[1], **Philipp Saggau**[2], **Benjamin Burkhard**[1,3]

1 Institute of Physical Geography & Landscape Ecology, Leibniz University of Hannover, Hannover, Germany, **2** Institute of Geography, Christian Albrechts University, Kiel, Germany, **3** Leibniz Centre for Agricultural Landscape Research (ZALF), Müncheberg, Germany

* rendon@phygeo.uni-hannover.de

Citation: Rendon P, Steinhoff-Knopp B, Saggau P, Burkhard B (2020) Assessment of the relationships between agroecosystem condition and the ecosystem service soil erosion regulation in Northern Germany. PLoS ONE 15(12): e0234288. https://doi.org/10.1371/journal.pone.0234288

Editor: RunGuo Zang, Chinese Academy of Forestry, CHINA

Received: May 22, 2020

Accepted: November 19, 2020

Published: December 7, 2020

Peer Review History: PLOS recognizes the benefits of transparency in the peer review process; therefore, we enable the publication of all of the content of peer review and author responses alongside final, published articles. The editorial history of this article is available here: https://doi.org/10.1371/journal.pone.0234288

Data Availability Statement: The data underlying the results presented in the study are available in the Supporting information S2.

Abstract

Ecosystems provide multiple services that are necessary to maintain human life. Agroecosystems are very productive suppliers of biomass-related provisioning ecosystem services, e.g. food, fibre, and energy. At the same time, they are highly dependent on good ecosystem condition and regulating ecosystem services such as soil fertility, water supply or soil erosion regulation. Assessments of this interplay of ecosystem condition and services are needed to understand the relationships in highly managed systems. Therefore, the aim of this study is twofold: First, to test the concept and indicators proposed by the European Union Working Group on Mapping and Assessment of Ecosystems and their Services (MAES) for assessing agroecosystem condition at a regional level. Second, to identify the relationships between ecosystem condition and the delivery of ecosystem services. For this purpose, we applied an operational framework for integrated mapping and assessment of ecosystems and their services. We used the proposed indicators to assess the condition of agroecosystems in Northern Germany and regulating ecosystem service *control of erosion rates*. We used existing data from official databases to calculate the different indicators and created maps of environmental pressures, ecosystem condition and ecosystem service indicators for the Federal State of Lower Saxony. Furthermore, we identified areas within the state where pressures are high, conditions are unfavourable, and more sustainable management practices are needed. Despite the limitations of the indicators and data availability, our results show positive, negative, and no significant correlations between the different pressures and condition indicators, and the control of erosion rates. The idea behind the MAES framework is to indicate the general condition of an ecosystem. However, we observed that not all proposed indicators can explain to what extent ecosystems can provide specific ecosystem services. Further research on other ecosystem services provided by agroecosystems would help to identify synergies and trade-offs. Moreover, the definition of a reference condition, although complicated for anthropogenically highly modified

Funding: The publication of this article was funded by the Open Access Publishing Fund of Leibniz Universität Hannover.

Competing interests: The authors have declared that no competing interests exist.

agroecosystems, would provide a benchmark to compare information on the condition of the ecosystems, leading to better land use policy and management decisions.

1. Introduction

Human well-being is strongly dependent on ecosystems, their biodiversity, condition, functionality and capacity to deliver multiple services. Ecosystem condition is clearly linked to ecosystem services and indicates the overall quality of an ecosystem unit in terms of its capacity to generate ecosystem services [1]. Assessing ecosystem condition can help to understand to what extent ecosystems can provide services in a sufficient quantity and quality. Some studies have focused on the links between natural capital and ecosystem services [2] and the relationships between biodiversity and ecosystem services [3]. However, understanding and quantifying the relationships between ecosystem condition and the provision of ecosystem services remains a research gap [4, 5].

International and European Union (EU) policies have integrated ecosystem condition into sustainability and conservation targets, comprising concepts such as ecosystem state, quality, status, health, integrity and functioning [6]. The Sustainable Development Goals (SDGs), for instance, were adopted by the United Nations in 2015, as "a call for action to end poverty, protect the planet and improve the lives and prospects of everyone everywhere" [7]. In particular, for biodiversity, goal 15 aims to protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and stop biodiversity loss.

In 2011, the EU adopted the Biodiversity Strategy to 2020 [8] and established six targets to halt the loss of biodiversity and ecosystem services in the EU by 2020. These targets aim to protect species and habitats, to maintain and restore ecosystems, sustainable agriculture and use of forests, sustainable fishing and healthy seas, to fight invasive alien species and stop the loss of biodiversity. The 7th Environmental Action Programme (EAP) was adopted by the EU in 2013 [9] and reinforces the targets and actions of the Biodiversity Strategy. The EAP aims to protect natural capital, stimulate resource-efficient, low-carbon growth and innovation, and safeguard human health and well-being while respecting Earth's limits [9]. Some topics that need further action at EU and national level are the protection of soils and the sustainable use of land and forest resources.

The EU has a dedicated working group, Mapping and Assessment of Ecosystems and their Services (MAES) [10], to support the implementation of Action 5 of Target 2 of the EU Biodiversity Strategy to 2020. Action 5 requires that all Member States map and assess the state of ecosystems and their services in their territory, assess the economic value of such services and integrate these values into accounting and reporting systems at the national and EU level [8]. MAES has developed a conceptual framework [11] and suggested a series of indicators to assess the condition of different ecosystem types including agroecosystems [12]. However, these indicators still need to be tested at EU, national/sub-national and regional level; and the links with ecosystem services need further investigation. Our study tests the indicators suggested by MAES for environmental pressures, ecosystem condition and the relationships with ecosystem services on a regional level, specifically in agroecosystems in Northern Germany with a focus on the soil erosion narrative.

Agroecosystems account for almost half of the land use area in the EU [13]. In Germany, more than half of the surface area is used for agriculture [14]. Agricultural land provides, on the one hand, multiple ecosystem services, especially biomass-related services such as food, fibre, fodder or energy, which are essential for human well-being [15, 16]. On the other hand, agriculture itself is strongly dependent on ecosystem services such as nutrient regulation, water supply, pollination (for selected crop species) or soil erosion regulation [17, 18]. Changes in the condition of agroecosystems may impair the availability of these services. Environmental pressures such as soil erosion, soil biodiversity decline, soil compaction, organic matter decline, soil sealing, and contamination, together with changing climate and water regimes, degrade these ecosystems [19]. Maintaining the good condition of agroecosystems is essential to guarantee resilience, halt biodiversity loss and preserve the sustainable provision of multiple ecosystem services.

Agroecosystems are strongly modified semi-natural systems and are managed with a strong focus on provisioning services [20]. These ecosystem service outputs are, at least in conventional farming, based on substantial anthropogenic human system inputs including fertilizer, insecticides, herbicides, energy, labour and machinery use and in some cases also irrigation water [21]. Besides ecosystem service outputs, agroecosystem service delivery has significant environmental effects such as greenhouse gas emissions, biodiversity loss or water eutrophication [18].

Due to the often long-term human interference in these systems, there are difficulties in defining a (natural) reference condition of agroecosystems. Agroecosystem condition cannot only be based on the physical and ecological properties of plants and soils but must take human interventions of agroecosystems into account [16]. An agroecosystems is in good condition when it supports biodiversity and supplies multiple provisioning, regulating and cultural ecosystem services, and there is no depletion of abiotic resources such as water, soil and air [12]. Nevertheless, the establishment of threshold values to determine whether an agroecosystem is in good or bad condition is still under debate [22]. Time reference condition as, for example, before the industrial revolution and/or different reference times as in other ecosystem types are not available. Besides these temporal issues, the involvement of multiple stakeholders (farmers, policy makers, planners, consumers, environmental groups), who may have different interests and perceptions about the condition of agroecosystems [23], hampers the reference state definition.

This study focuses on the ecosystem service *control of erosion rates* and takes into account that soil erosion by water is a major problem in soil conservation in the EU [24, 25]. Soil erosion by water accounts for the largest share of soil loss in Central European agricultural ecosystems, especially in areas with steep slopes [26]. Unsuitable management activities threaten croplands by increasing the vulnerability of soils to erode [24]. The objective of this study is to conduct an integrated assessment of ecosystems for one exemplary ecosystem service in a specific ecosystem type and by using the indicators proposed by MAES. We apply an operational framework for integrated mapping and assessment of ecosystems and their services suggested by Burkhard et al. [27] in agroecosystems in the Northern German Federal State of Lower Saxony. For this purpose, we follow a stepwise approach: i) the identification of the policy objective "healthy soils"(in our study exemplified by soil erosion regulation); ii) the identification and mapping of agroecosystems; and iii) the selection, quantification and mapping of indicators of agroecosystem condition.

The main goal of this study is to test the feasibility of the indicators proposed by MAES for the assessment of agroecosystem condition at a regional level. Thereby, we hope to improve the methodology and to increase the applicability of the MAES framework and indicators. The results will be relevant for other (also non-EU/MAES-related) comparable indicator-based studies in agroecosystems, their condition and ecosystem services.

The article is organized as follows: First, we describe the methodological approach used for the integrated assessment. Then we show maps of the different indicators to evaluate the environmental pressures, ecosystem condition and the ecosystem service *control of erosion rates*.

We also statistically analyse the relationships between ecosystem condition and the *control of erosion rates*. Then we discuss the main limitations of the indicators and the relationship between environmental pressures, ecosystem condition and ecosystem services and conclude with recommendations for further improvement of the MAES framework and their indicators.

2. Methods

2.1 Study area

Lower Saxony is a federal state located in the north-west of the Federal Republic of Germany, adjoins the North Sea and has an area of 47,620 km^2 (Fig 1). Its climate is characterized as sub-oceanic with average temperatures ranging from 8.3 to 9.5˚C and mean precipitation values ranging from 654 mm in the south-west to 840 mm in the central and northern areas [28]. Agriculture is the main land use with 2.6 million hectares (approximately 53.7% of the total

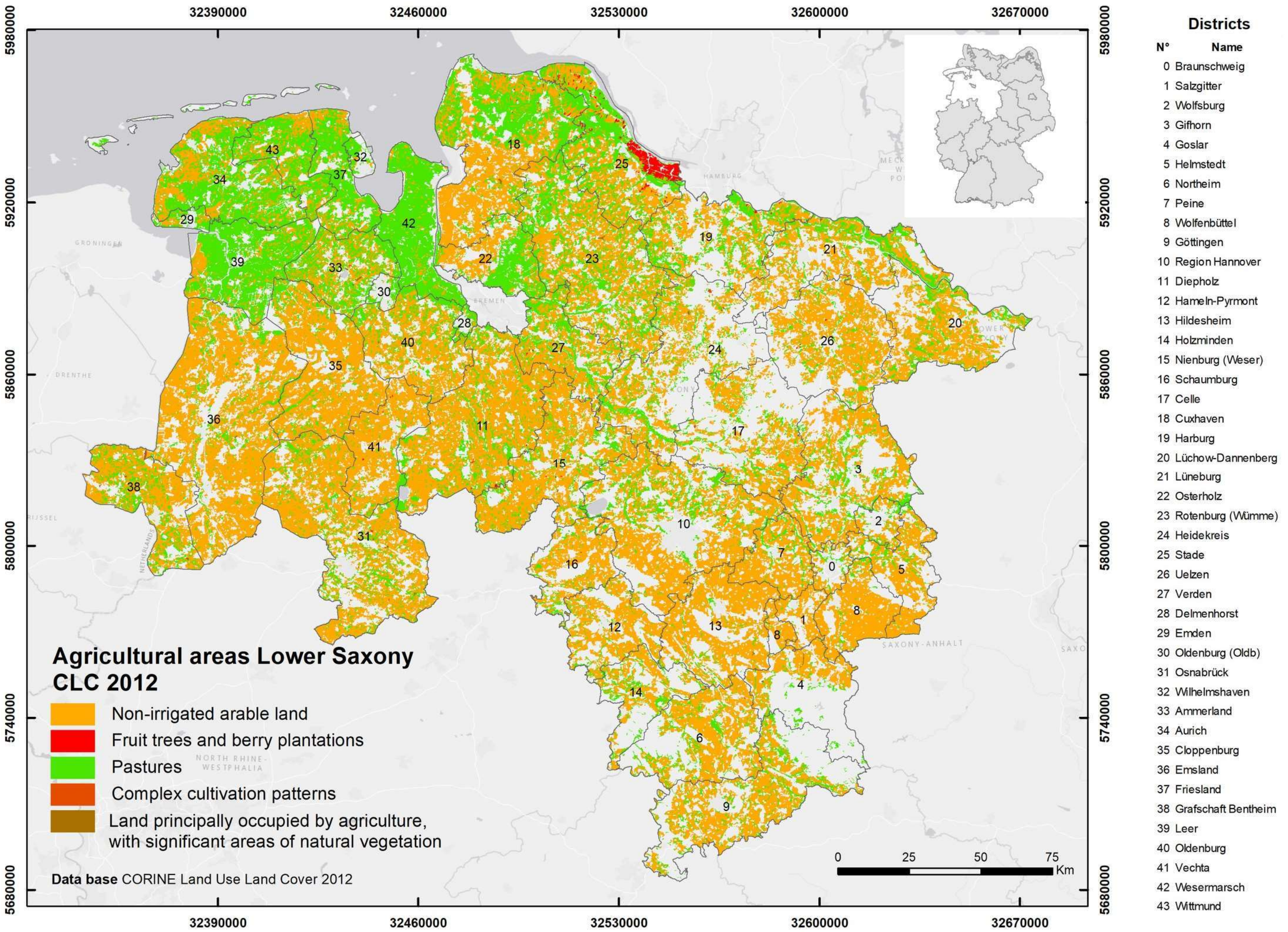

Fig 1. Agricultural areas in Lower Saxony. (Based on CORINE Land Use Land Cover data, of 2012 obtained from the European Environmental Agency [30] and administrative units from the German Federal Agency for Cartography and Geodesy © GeoBasis-DE / BKG (2017) [31]).

https://doi.org/10.1371/journal.pone.0234288.g001

territory), of which 1.9 million hectares are arable land, 0.7 million hectares are permanent grassland and around 20,000 hectares are permanent crops [29].

More than half of the arable land in Lower Saxony is used to grow cereals (mainly winter wheat and winter barley); the remaining area is used for fodder crops, oilseeds, or potatoes. The farm sizes are very diverse and range from a few hectares of specialized horticultural businesses to large arable farms with several hundred hectares. On average, the farms have a size of 83 ha and about 75% of all farms keep animals, especially dairy cattle and pigs. Farm type, specialization and size can be used as a function of soil fertility, climate conditions and historical land use strategies [29].

2.2 Conceptual framework

In this study, we used the operational framework proposed by Burkhard et al. [27] that guides integrated mapping and assessment of ecosystems and their services. Fig 2 provides a summary of the framework that entails nine steps: (1) theme identification; (2) identification of ecosystem type; (3) mapping of ecosystem type; (4) definition of ecosystem condition and identification of ecosystem services to be delivered by agroecosystems; (5) selection of indicators for ecosystem condition and ecosystem services; (6) quantification of ecosystem condition and ecosystem services indicators; (7) mapping ecosystem condition and ecosystem services; (8) integration of results; and (9) dissemination and communication of results. These steps are described in detail in the next paragraphs.

2.3 Theme identification: Policy objective healthy soils (Step 1)

The first step of the operational framework refers to the *question and theme identification*, which must be addressed in the ecosystem assessment in order to be relevant for policy,

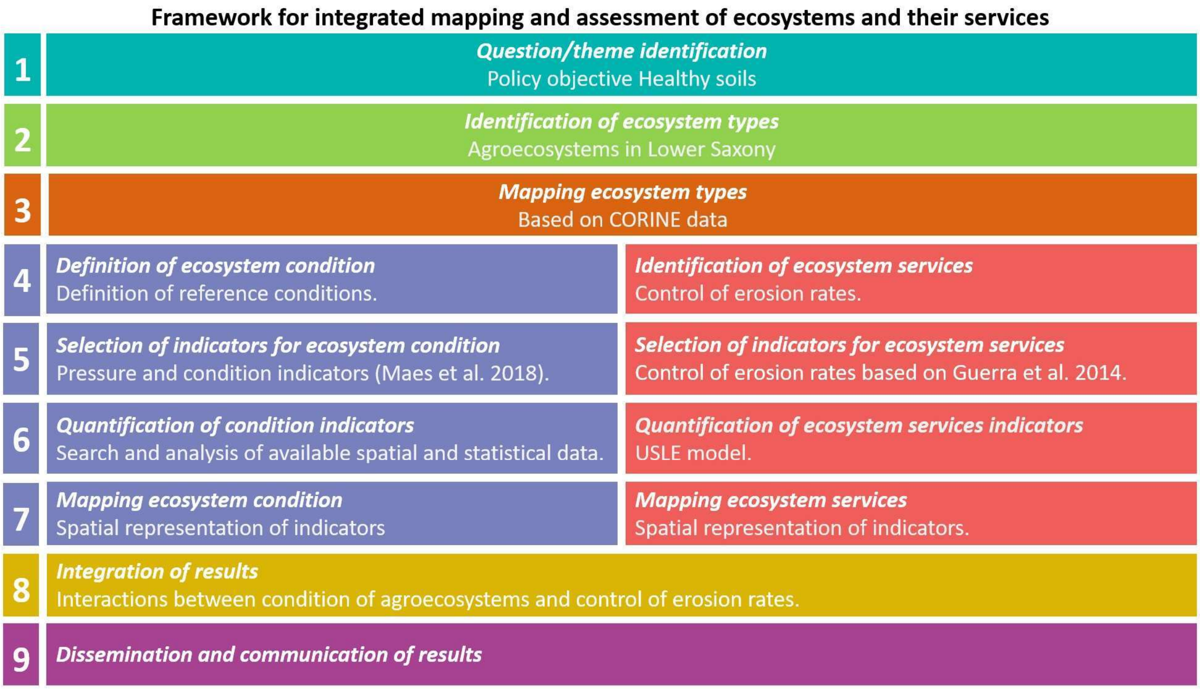

Fig 2. Conceptual framework applied for integrated mapping and assessment of agroecosystems and the ecosystem service *control of erosion rates*. (based on Burkhard et al. [27]).

https://doi.org/10.1371/journal.pone.0234288.g002

society, business or science. In this case study, we identified the policy objective *maintaining healthy soils*. Healthy soils, especially for agriculture, have high functionality, including biodiversity, fertility and the capacity to sustainably deliver multiple ecosystem services. These services include food and fibre, climate and water regulation, water purification, carbon sequestration, nutrient cycling and provision of habitat for biodiversity [32]. At the same time, the delivery of other ecosystem services such as water supply and regulation, pollination and soil erosion regulation should not be impaired [33].

For this study, we focus on the ecosystem service *control of erosion rates*. Fig 3 shows the condition attributes that determine the delivery of this ecosystem service. Soil condition and the presence of semi-natural areas within or in the vicinity of the agricultural fields are key for the delivery of this service [34, 35]. Additionally, livestock can affect this service by altering the structural condition of soils [36]. Crop rotations and crop types as well as the state of the landscape in which the agroecosystem is embedded are also important to control erosion rates [37]. In this case, the main typologies of environmental pressures, habitat and land conversion, climate change, input nutrients and pesticides, overexploitation, and introduction of invasive species, as proposed by Maes et al. [12], affect the condition attributes.

Healthy soils have been included as a relevant issue in national and European policies. In Germany, for instance, the objective of maintaining and preserving healthy soils was established in the Federal Soil Protection Act of March 17th 1998 [38] and in the Federal Soil Protection and Contaminated Sites Ordinance of July 12th 1999 [39]. Both regulations aim to sustainably secure and restore the soil functions by protecting soils against harmful changes and remediating contaminated sites. At EU level, the European Commission adopted the Thematic Strategy for Soil Protection [40] to protect soils across the EU. Although the proposal for

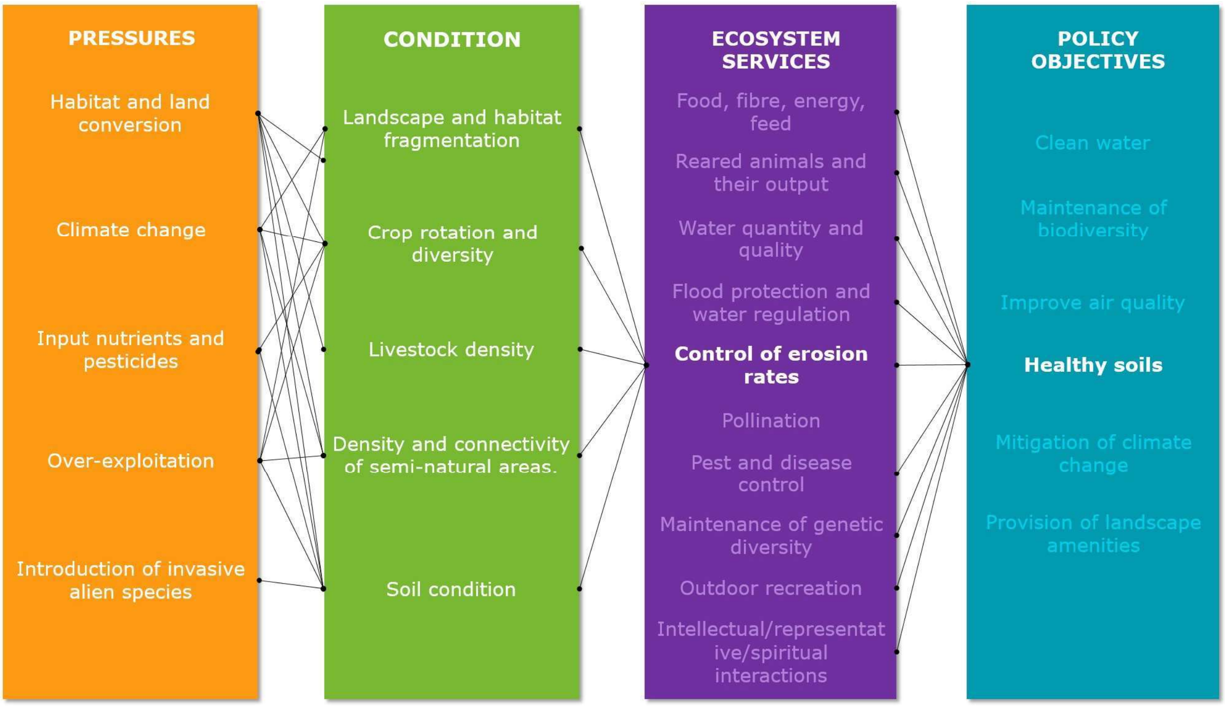

Fig 3. Synthesis of links between environmental pressures, condition, ecosystem service *control of erosion rates* and policy objectives in agroecosystems. (based on Maes et al. [12]).

https://doi.org/10.1371/journal.pone.0234288.g003

a Soil Framework Directive was withdrawn by the Commission in 2014, the 7th Environmental Action Programme (EAP) came into force in 2014 [9]. The objective of the EAP is the protection and sustainable use of soils. Under priority objective 1, the EAP states that by 2020 "land is managed sustainably in the Union, soil is adequately protected and remediation of contaminated sites is well underway". More efforts are required to reduce soil erosion and increase soil organic matter, as well as to remediate contaminated sites.

2.4. Identification and mapping of the ecosystem type agroecosystems (Steps 2 and 3)

The second step of the operational framework refers to the identification of the ecosystem type (s). In this study the ecosystem type is agroecosystems which are "communities of plants and animals interacting with their physical and chemical environments that have been modified by people to produce food, fibre, fuel, and other products for human consumption and processing" [41]. Maes et al. [11] proposed a classification of ecosystem types for MAES, in which cropland and grassland belong to the ecosystem type agroecosystems. We selected cropland ecosystems in Lower Saxony because croplands are the main provider of ecosystem services such as biomass used as food, fodder or as an energy source. Croplands are threatened by mismanagement and external pressures such as droughts or floods caused by climate change. Due to management-induced bare soils during the year, cropland soils are especially affected by soil erosion.

The third step entails the mapping of the previously identified agroecosystems. For this purpose, we used the CORINE Land Cover Data for the year 2012 [42], particularly the CORINE Land Cover type 2. Agricultural areas, which includes: 211. Non-irrigated arable land, 221. Vineyards, 231. Pastures, 242. Complex cultivation patterns, and 243. Land principally occupied by agriculture, with significant areas of natural vegetation (see Fig 1).

2.5. Definition of ecosystem condition and identification of ecosystem services delivered by agroecosystems (Step 4)

The fourth step define the ecosystem condition and the delivered ecosystem services. Agroecosystems are usually purposely heavily modified ecosystems [18], and it is not feasible to compare their condition with undisturbed natural ecosystems. Table 1 shows the median values and the available reference values used in this study to determine the condition of the agroecosystems in Lower Saxony, based on the selected indicators.

We selected the ecosystem service *control of erosion rates*, because soil erosion is one of the main threats to soils [40] with negative impacts on crop production, water quality, mudslides, eutrophication, biodiversity and carbon stock loss [26]. Soils are the medium on which crops are grown and their functionality is the base for biomass production, storage, filtration and transformation of nutrients and water. Furthermore, healthy soils are essential for biodiversity conservation and act as carbon storage pools. Soils are the platform for human activities, provide raw materials and store geological and archaeological heritage [43]. Soil degradation leads to the decline of many ecosystem services [44] and soil erosion decreases soil surface and then soil thickness. Especially the loss of the humus-rich, fertile topsoil layers leads to a reduction of soil functionality and the capacity to provide ecosystem services. If these losses are not compensated by soil formation, soil erosion will threaten sustainable crop production as well as water regulation and filtration capacities [45].

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

Indicator class	Indicator	Description	Units	Spatial resolution	Year	Median Lower Saxony	Reference value	Source
				Pressure indicators				
Habitat conversion and degradation	Change in ecosystem extent	Change in the area (size) of the ecosystem within the years 2006 and 2012.	% per year	100 m	2006–2012	0	N.A.	[30]
Climate	Mean annual temperature	Annual mean of the monthly averaged mean daily air temperature in 2 m height above ground	°C	1000 m	1988–2018	9.74	N.A.	[46]
	Mean annual precipitation	Annual sum of monthly precipitation.	mm		1988–2018	765.7	N.A.	
	Drought index	Annual mean of drought index after de Martonne.	mm °C^{-1}		1995–2018	37.78	Very humid (35–55) [47]	
	Precipitation 10 mm	Number of days with precipitation ≥ 10 mm per year.	Number of days		1988–2018	19.45	N.A.	
	Precipitation 20 mm	Number of days with precipitation ≥ 20 mm per year.				3.77	N.A.	
	Precipitation 30 mm	Number of days with precipitation ≥ 30 mm per year.				0.91	N.A.	
	Beginning of vegetation period	Number of consecutive days of the year. It indicates the beginning of the first spring	Consecutive days of the year		1992–2018	82.68	N.A.	
	Summer soil moisture	Modelled trend in soil moisture over a depth of 60 cm.	% NFK (usable field capacity)		1991–2010	66.82	N.A.	
Others	Soil erosion	Amount of soil loss per hectare in a year (Actual soil loss)	t ha^{-1} per year	50 m	2010	0.13	0–3 [48]	[49]
	Loss of organic matter	Percentage of soil organic carbon loss per year.	Mg C ha^{-1} per year	1000 m	2013	0.015	N.A.	[50–52]
				Ecosystem condition indicators				
Structural ecosystem attributes (general)	Crop diversity	Average number of crops in a 10 km diameter per municipality	Number of crops	50 m	2018	44.41	N.A.	[53]
	Density of semi-natural areas	Percentage of semi-natural areas	%	50 m	2012	13.16	N.A.	[42]
	Share of fallow land in Utilized Agricultural Area (UAA)	Percentage of arable land that is not being used for agricultural purposes within the UAA.	%	Municipality	2010	0.80	N.A.	[54]
	Share of arable land in Utilized Agricultural Area (UAA)	Percentage of land used for the production of crops within the UAA	%	Municipality	2010	81.32	N.A.	[54]
	Share of permanent crops in Utilized Agricultural Area (UAA)	Percentage of land used for permanent crops within the UAA	%	Municipality	2010	0	N.A.	[54]
	Livestock Density	Stock of animals (cattle, sheep, goats, equidae, pigs, poultry and rabbits) converted in livestock units (LU) per hectare of UAA.	LU ha^{-1}	Municipality	2010	0.81	N.A.	[54]

(*Continued*)

Table 1. (Continued)

Indicator class	Indicator	Description	Units	Spatial resolution	Year	Median Lower Saxony	Reference value	Source
Structural soil attributes	Soil Organic Carbon (SOC)	Concentration of topsoil organic carbon.	% or gr kg^{-1}	1000 m	2010	2.03	1–2% [55]	[56]
	Soil erodibility	Susceptibility of soil to erosion by runoff and raindrop impact.	K factor [t $ha^{-1}N^{-1}$]	50 m	2010	0.21	N.A.	[56, 57]
	Bulk density	Weight of soil per cubic meter	t m^{-3}	500 m	2015	1.3	1.6 g cm^{-3} for sandy and sandy loam soils 1.75 g cm^{-3} for coarse textured soils (with clay content < 17.5%) [55]	[58, 59]
Ecosystem services indicators								
Control of erosion rates	Soil erosion risk	Potential soil loss.	t ha^{-1} per year	50 m	2010	0.81	N.A.	[49]
	Prevented soil erosion	Difference of potential and actual soil loss	t ha^{-1} per year	50 m	2010	0.67	N.A.	[56]
	Provision capacity	Share of mitigation of soil erosion (0 to 1)	Dimensionless	50 m	2010	0.85	From 0 (no mitigation) to 1 (complete mitigation)	[56]

https://doi.org/10.1371/journal.pone.0234288.t001

2.6. Selection of indicators for agroecosystem condition and the ecosystem service *control of erosion rates* (Step 5)

The fifth step refers to the selection of the indicators for the assessment of ecosystem condition and ecosystem services. As this study aims to test the framework and indicators proposed by MAES, we chose the indicators for pressures and condition of agroecosystems presented in the 5th MAES report [12]. We selected the indicators for the ecosystem service *control of erosion rates* based on existing literature and the frameworks used by Guerra et al. [60] and Steinhoff-Knopp and Burkhard [25].

2.6.1. Criteria for selecting indicators. Ecosystem condition indicators allow us to assess the overall quality of an ecosystem and its main characteristics that underpin its capacity to deliver ecosystem services [1]. These indicators, together with information on ecosystem extent and services, constitute the main inputs for integrated ecosystem assessments that analyse the links between ecosystem condition, habitat quality and biodiversity, ecosystem services, and the consequences for human wellbeing [27].

Maes et al. [12] highlighted the main characteristics that ecosystem condition indicators must have to inform policies related to the use and protection of natural resources. First, ecosystem condition indicators need to be aligned with the MAES framework in which socioeconomic systems are linked with ecosystems through the flow of ecosystem services and the drivers that affect ecosystems. Second, they need to support the objectives of the EU environmental legislation and the objectives of the natural capital accounts. Third, they need to be policy-relevant, which means that they support EU environmental policies, related national policies and any other policies. Fourth, they need to be spatially explicit by considering the distribution of ecosystems and their use, and they need to be specific for each ecosystem type. Fifth, they need to contribute to measuring progress/trends against a policy baseline towards different policy targets.

We therefore, adopted the following criteria for selecting the indicators.

- Relevancy: Indicators are relevant to the ecosystem service *control of erosion rates*. There is a clear connection between the condition parameter and the provision of the ecosystem service. This connection was determined based on the authors' expertise and scientific literature.
- Availability of data: The data have an appropriate resolution for the study area and are detailed enough to recognize regional features (i.e. spatially explicit or data at the level of municipalities).
- Quantifiable: Indicators are quantifiable and data can be compared among municipalities.
- Reliability: Both the quantification and monitoring of the indicators are reliable (i.e. data obtained from officially reported data sets).

A total of 23 indicators was selected for the assessment, whereof 11 correspond to pressure (including seven climate indicators), nine correspond to ecosystem condition and 3 to the ecosystem service *control of erosion rates*.

2.7. Quantification of indicators (Step 6)

The sixth step refers to the quantification of indicators for condition and the ecosystem service *control of erosion rates*. We conducted a data search in several public databases of the EU, Germany and Lower Saxony. Table 1 provides detailed descriptions of the indicators, including the spatial resolution, year of data collection and data sources. We used the best data available concerning temporal and spatial resolution, which may not be optimal for this kind of study as discussed in section 4.1. We describe the calculation of each indicator in detail in S1 File.

2.8. Mapping of indicators (Step 7)

This step refers to the spatial visualisation of the ecosystem condition and the ecosystem service *control of erosion rates* indicators in maps. All the indicators were edited in ArcGIS 10.7 for representation and analysis. They were aggregated to the level of the municipalities to facilitate their comparison.

2.9. Integration of results (Step 8)

The integration of results refers to the analysis of relationships and interactions between the condition of agroecosystems and the supply of the ecosystem service *control of erosion rates*. This analysis was conducted from two different angles: the assessment of statistical correlations and the analysis of the spatial distributions and relationships from the compiled maps.

For the statistical correlations, seven classes of the ecosystem service *control of erosion rates* were selected, based on the classification proposed by Steinhoff-Knopp and Burkhard [25] for the potential soil erosion by water on croplands in Lower Saxony. The classes are *No to very low* (less than 1 t ha^{-1} per year eroded soil), *very low* (1 to 5 t ha^{-1} per year), *low* (>5 to 10 t ha^{-1} per year), *medium* (>10 to 15 t ha^{-1} per year), *high* (>15 to 30 t ha^{-1} per year), *very high* (>30 to 55 t ha^{-1} per year) and *extremely high* ($\geq$ 55 t ha^{-1} per year). We considered the environmental pressures and ecosystem condition indicators per class, which are also used to classify the actual *control of erosion rates*. We applied the Akaike Information Criterion (AIC) [61] technique to estimate the likelihood of the different pressure and condition indicators to predict the values of the seven *control of erosion rates* classes mentioned above. Additionally, we applied the Kruskal-Wallis rank test [62] on the class medians to detect significant differences between the seven classes. We carried out the Jonckheere-Terpstra test [63, 64] to identify positive or negative relationships between the delivery of the ecosystem service *control of erosion* *rates* and the environmental pressures and ecosystem condition. The statistical work was conducted in RStudio (version 1.2.1335) [65].

To show an integrated overview of the distribution of the ecosystem service *control of erosion rates*, the potential soil loss, and the pressures and condition in Lower Saxony, we normalized all the variables and standardized them to a 0 to 100 scale to make them comparable. When looking at pressures and condition variables, we considered the presumable effect of each of them based on the supply of the ecosystem service *control of erosion rates* to do the normalization. The pressure indicators *drought index* and *summer soil moisture*, for instance, are thought to have a positive effect on the supply of the ecosystem service. Regarding the drought index (Ia DM) in regions classified as arid (Ia DM<10) and semi-arid (10≤ Ia DM<20) in the *index of the Martonne*, the protective cover provided by plants against rain splash decreases with increased aridity [66]. This means that the higher the value of the drought index, the lower the erosion risk. Similarly, higher soil moisture maximizes vegetation cover, resulting in the minimization of sediment transport capacity, with important differences between clay and sandy soil textures [67]. On the other hand, the condition indicators *fallow land*, *livestock density* and *soil erodibility (K factor)* have a presumably negative effect on the *control of erosion rates*. Poor structural stability, as well as less plant cover in fallow systems result in an increased erosion risk [68], and a higher percentage of fallow land, indicating areas with bare soil, increases the soil erosion in a specific area. Likewise, the trampling of livestock disturbs and loosens soil, which makes it easier for soil to be removed by agents of transport and therefore increases erodibility [36]. These five indicators were multiplied by -1 in order to take into account the positive or negative effects when normalizing the original data. We then calculated the average of the normalized values for the pressure and condition indicators.

To spatially visualize the relationships between the indicators, we created maps showing the overlaps of pressures, condition, soil erosion risk, and provision capacity. This analysis allowed us to identify how pressures and condition related to soil erosion risk and the provision capacity of the agroecosystems are able to control soil erosion.

2.10. Dissemination and communication of results (Step 9)

This step refers to the preparation of maps and other accompanying material for effective dissemination and communication of the results. According to Burkhard et al. [27], the results must be communicated to potentially interested decision makers and other stakeholders in order to answer the initial question(s) posed in Step 1. However, for this assessment, we did not involve stakeholders and these results have not been communicated.

3. Results

The results of the assessment are presented in maps showing the distribution of the indicators of environmental pressures, ecosystem condition and the ecosystem service *control of erosion rates* within Lower Saxony. Other graphs and maps show the integration of results and the relationships between pressures, condition and *control of erosion rates* (see data per municipality in S2 File).

3.1. Mapping and assessment of agroecosystem condition in Lower Saxony

3.1.1 Pressure indicators. *Change in ecosystem extent.* Agroecosystems in Lower Saxony did not experience significant changes in extent from the year 2006 to 2012. Most of the municipalities showed changes from 0% to +/- 0.7%. The changes were more significant in Amelinghausen (district of Lüneburg), in the north-east, with an increase of 19%; Himmelpforten (Stade), in the north, (17,4%); and Seedorf (Rotenburg -Wümme), also in the north with an increase of 16,5%. On the other hand, there were significant reductions in the size of agroecosystems in Spelle (Emsland) in the west and Wallmoden (Goslar) in the south-east, both with a decrease of 18% (Fig 4a).

Climate. Mean annual temperatures in Lower Saxony ranged from 6.7 ˚C to 10.3 ˚C, with the lowest temperatures recorded in mountainous areas such as the Harz (district of Goslar) in the south-east (Fig 4b).

Mean annual precipitation in Lower Saxony ranged from 570 mm in the east to 1344 mm in the mountainous region located in the south-east. Precipitation in the coastal areas and mountainous regions in the south ranged from 826 mm to 1021 mm (Fig 4c).

Drought index or aridity index of the Martonne ranged from humid (28) to extremely humid (78) across the study area. Municipalities located in mountainous regions (Braunlage and Clausthal-Zellerfeld in the district of Goslar) had the lowest aridity together with areas with continental weather (near the state of Saxony Anhalt and some areas close to North-Rhine Westphalia) (Fig 4d).

The number of days with precipitation higher than 10, 20 and 30 mm was negatively correlated with the amount of rainfall that was recorded. For instance, the number of days with precipitation between 10 and 20 mm ranged from 2 to 42 days, while the number of days with precipitation between 20 mm and 30 mm ranged from 0.6 to 4.6 days (Fig 4e–4g). Similarly to other climatic parameters, higher precipitations occurred in mountainous and coastal regions.

The *beginning of the vegetation period* in Lower Saxony ranged from 77 to 106 consecutive days of the year during the period between 1992 and 2018 across the different municipalities. The areas with the latest beginning of spring are located in the mountainous regions (Braunlage and Clausthal-Zellerfeld in the district of Goslar), which is related to the prevalence of lower temperatures throughout the year (Fig 4h).

Summer soil moisture in Lower Saxony was in line with the precipitation levels showing percentages of plant-available water ranging from 60% in the eastern area to 83% in the south-east (mainly mountainous regions) (Fig 4i).

Soil erosion had on average the highest values in the southern mountainous region (56.9 t ha^{-1} per year in the municipality of Wenzen, district of Holzminden), whereas municipalities located in the Lower Saxonian German Plain in the northern half of the state, showed median values below 1.8 t ha^{-1} per year of eroded soil (Fig 4j).

Loss of organic matter had the highest values in the southern region (0.48 Mg C ha^{-1} per year, in Braunlage, district of Goslar). Other municipalities in the east and north showed values ranging from 0.06 to 0.1 Mg C ha^{-1} per year. The median for the whole federal state was 0.02 Mg C ha^{-1} per year (Fig 4k).

3.1.2 Ecosystem condition indicators. *Crop diversity* varied greatly across Lower Saxony. North-eastern and central areas had the highest values, ranging from 55 to 73 crop species. On the other hand, the North Sea islands in the north-west and the mountainous area in the district of Goslar in the south-east had values below 21 crop species. (Fig 4l).

The *density of semi-natural areas* was higher in the north-west of Lower Saxony, with percentages as high as 91% in the municipality of Ovelgönne (district of Wesermarsch) and more than 87% in Engelschoff (district of Stade), mostly represented in the form of pastures. Almost 40% of the municipalities distributed in the southern and north-eastern regions had less than 10% of semi-natural areas and feature intensive agriculture, mainly with cereal crops (Fig 4m).

The *share of fallow land in UAA* was relatively low in Lower Saxony, with a median of 0.86% for the whole federal state. Almost 60% of the municipalities had a percentage of fallow land below 1.3%, whereas only 41 municipalities had values higher than 6.7%, mainly located in the eastern region (Fig 4n).

The *share of arable land in UAA* was higher in areas with a lower density of semi-natural elements, which is in line with theoretical expectations, where most regions with intensive agriculture tend to have low values of semi-natural vegetation [69]. Municipalities with a share of arable land of around 80% were mainly distributed in the east and south-west of the federal state, where the production of crops such as grains and wheat was high (Fig 4o).

Share of permanent crops in UAA was low in Lower Saxony, with only five municipalities located in the district Stade in the north, with a share higher than 80%. This region, "Altes Land", has mostly tree and berry fruits and fruit tree plantations. The median share of permanent crops for the federal state was 0%, since only 175 municipalities out of the 959 in the study have values higher than 0% (Fig 4p).

Livestock density was higher in the western part of Lower Saxony with the highest number of livestock units per hectare (LU ha^{-1}) in the districts of Vechta and Cloppenburg with values between 2 and 3.6 LU ha^{-1}. Values smaller than 0.4 LU ha^{-1} were recorded in the eastern part of the state and on some islands in the north-west. The median livestock density of the federal state was 0.8 LU ha^{-1} (Fig 4q).

Soil Organic Carbon concentrations were high in the north-west of the state with levels of topsoil organic carbon higher than 4% (Fig 4r). This concurs with large peatland areas in north-western Germany. The levels were higher than the threshold values between 1 and 2% as estimated by Kibblewhite et al. [55]. On the other hand, levels of soil organic carbon in the eastern region of the study area, especially in the district of Wolfenbüttel, were lower than 0.9%, which could be an indication of potential degradation.

Soil erodibility (K factor) values ranged from 0 to 0.5 t h ha^{-1} N^{-1}. Some municipalities in the south-east and the north-west had higher mean values (Fig 4s). Additionally, the topsoils in these areas had high contents of silt, which makes them highly erodible [24].

Bulk density in Lower Saxony ranged from 0.97 t m^{-3} to 1.53 t m^{-3}, distributed throughout all the municipalities (Fig 4t). These values were lower than the threshold levels for sandy and sandy loam soils of 1.6 g cm^{-3} estimated by Huber et al. [48].

3.1.3 Ecosystem service indicators. *Soil erosion risk* showed high values in the south-east part of the study area with values higher than 140 t ha^{-1} per year, which are considered extremely high according to Steinhoff-Knopp & Burkhard [25]. However, the median value of the entire state was calculated to be 0.8 t ha^{-1}per year, and some municipalities even showed mean values as low as 0.01 t ha^{-1} per year (Fig 4u).

Prevented soil erosion showed a concentration of high values mainly in the south-east (Fig 4v). As previously mentioned, this area also showed high values of potential and actual soil losses. However, the actual soil loss was considerably lower than the calculated soil loss potential, resulting in a high ecosystem service provision in this area.

Provision capacity was relatively high in Lower Saxony with values ranging from 0.74 to 0.95 across all municipalities (Fig 4w). The mean value of 0.85 indicates that most parts of the study area are protected against soil erosion.

3.2. Relationships between agroecosystem condition and control of erosion rates

3.2.1 Analysis of the relationships between indicators. In order to understand the relationships between agroecosystem condition and supply of the ecosystem service *control of erosion rates*, we analysed the likelihood of the different indicators (excluding soil erosion, soil erosion risk, and soil erodibility which were partly included in the calculation of the ecosystem service) to predict the classes from *no to very low* to *extremely high* mentioned in section 2.9. Our results show that the indicators that best predicted these classes are: *loss of organic carbon*,

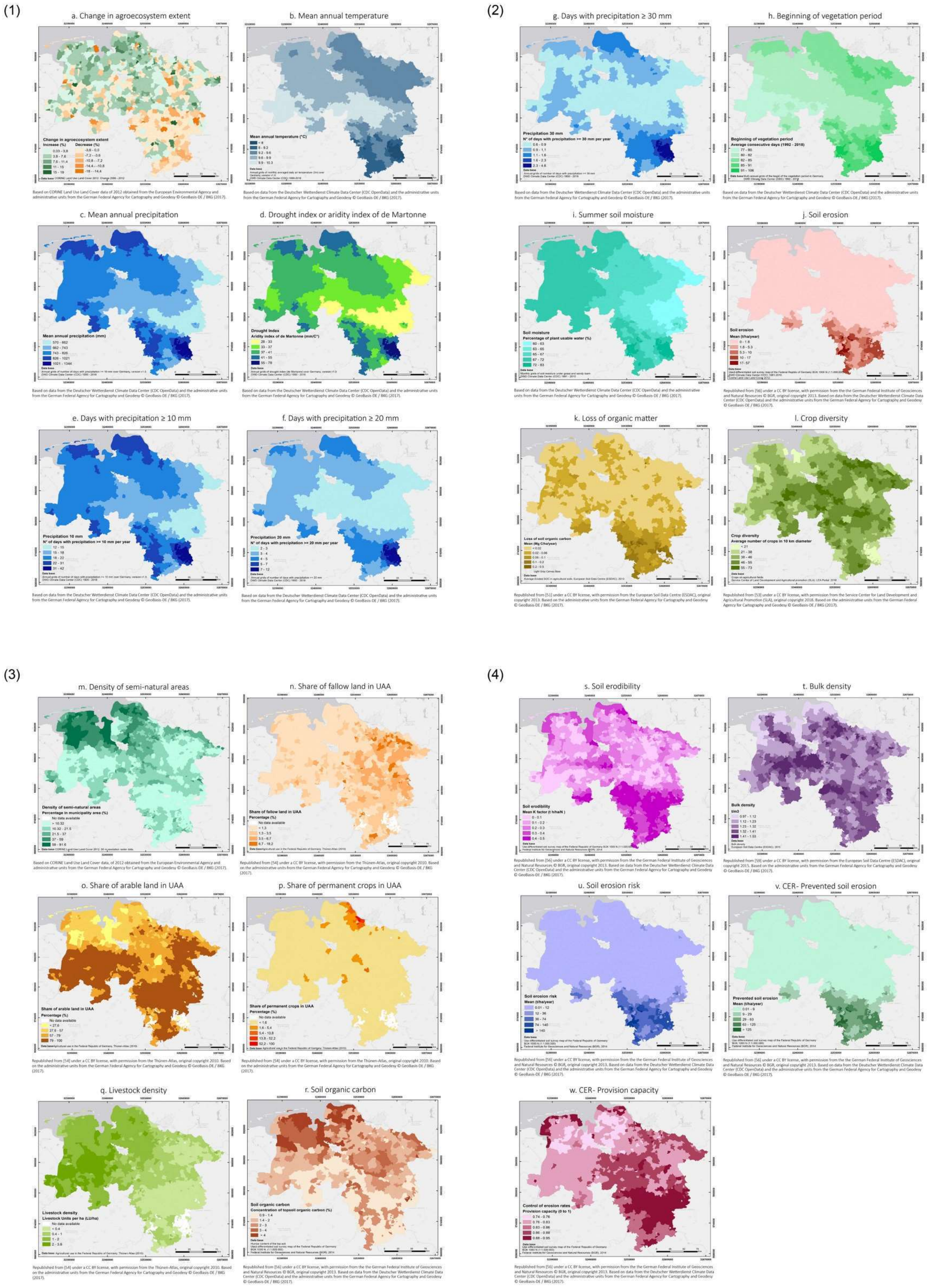

Fig 4. Maps of indicators of environmental pressure, ecosystem condition and control of erosion rates in Lower Saxony. (Larger maps are provided in the S3 File).

https://doi.org/10.1371/journal.pone.0234288.g004

followed by *mean annual temperature* and *beginning of vegetation period*. In contrast, the indicators that had the lowest likelihood to predict the classes were *mean annual precipitation* and *crop diversity* (see AIC rank on Fig 5 and S1 Table).

Additionally, we analysed the relationships between pressures and ecosystem condition and the ecosystem service *control of erosion rates* (see S1 Table). Our results show that the rates of control of erosion were slightly higher in areas with increased *ecosystem extent* ($p < 0.05$) (Fig 5a). For the climatic variables, we found negative, positive as well as not significant correlations. For instance, the *control of erosion rates* was extremely high in areas with lower temperatures ($p < 0.05$) (Fig 5b). On the other hand, high and extremely high *control of erosion rates* occurred in areas where variables such as *drought index*, *days with precipitations higher than 20 and 30 mm* and *beginning of vegetation period* were high ($p < 0.05$). However, the relationships between *mean annual precipitation*, *days with precipitation higher than 10 mm*, *summer soil moisture* and the *control of erosion rates* were not significant ($p = 0.2$), ($p = 0.2$), and ($p = 0.7$), respectively (Fig 5c–5i). Higher values of *soil erosion* and *loss of soil organic matter* occurred in ecosystems providing higher *control of erosion rates* ($p < 0.05$) (Fig 5j and 5k). In contrast, the *density of semi-natural areas* and *livestock density* were low where the *control of erosion rates* was high or very high ($p < 0.05$) (Fig 5m and 5q). Moreover, there was no significant relationship between *crop diversity*, *fallow land*, *arable land*, and *soil organic carbon* with the *control of erosion rates* ($p = 1$) (Fig 5l and 5n–5r). The condition indicators *soil erodibility*, *bulk density*, *soil organic carbon* and the ecosystem service indicators *soil erosion risk* and *provision capacity* showed a positive relationship with the *control of erosion rates* ($p < 0.05$) (Fig 5s–5v).

3.2.2 Overlaps between environmental pressures, ecosystem condition, soil erosion risk and ecosystem service provision capacity. Fig 6 shows the spatial distribution of environmental pressures and condition in relation to the ecosystem service *provision capacity* in Lower Saxony. Fig 6a shows the superimposition of the normalized values of condition and provision capacity. Areas with high provision capacity and high condition levels (darker colours on the right top corner) were mainly located in the north-western and central regions. High provision capacity was also found in municipalities with medium condition levels located mainly in the southern and eastern regions (dark blue). High provision capacity was not necessarily associated with a high level of ecosystem condition in municipalities such as the Harz in the south-east (light green). However, this area is mostly covered by mountainous forest and is part of a national park. On the contrary, some municipalities in the north-west showed low provision capacity, but medium condition levels (light blue). No municipalities in Lower Saxony had low provision capacity and low condition levels.

The spatial distribution of provision capacity and pressures shows that most of the study area had medium levels of pressures and medium or high provision capacity (darker blue colours) (Fig 6b). The highest provision capacity occurred in the central and north-western regions where the pressures had a medium level (dark green). Low provision capacities and medium pressure levels occurred in the north-west, especially in some municipalities of the districts of Cuxhaven, Rotemburg (Wümme) and Wesermarsch.

Fig 7 shows the spatial distribution of soil erosion risk, environmental pressures and condition in relation to the ecosystem service provision capacity in Lower Saxony. Fig 7a shows the superimposition of the normalized values of soil erosion risk, condition and provision capacity. Areas with high provision capacity, high condition levels, and low erosion risk (top of the triangle) were mainly located in the north-western region. High provision capacity was also found in municipalities with medium condition levels and low erosion risk located mainly in the western region and (to a lesser extent) in the north-east (blue colour on the right side of the triangle) and the south (grey colour on the right side of the triangle). Medium provision

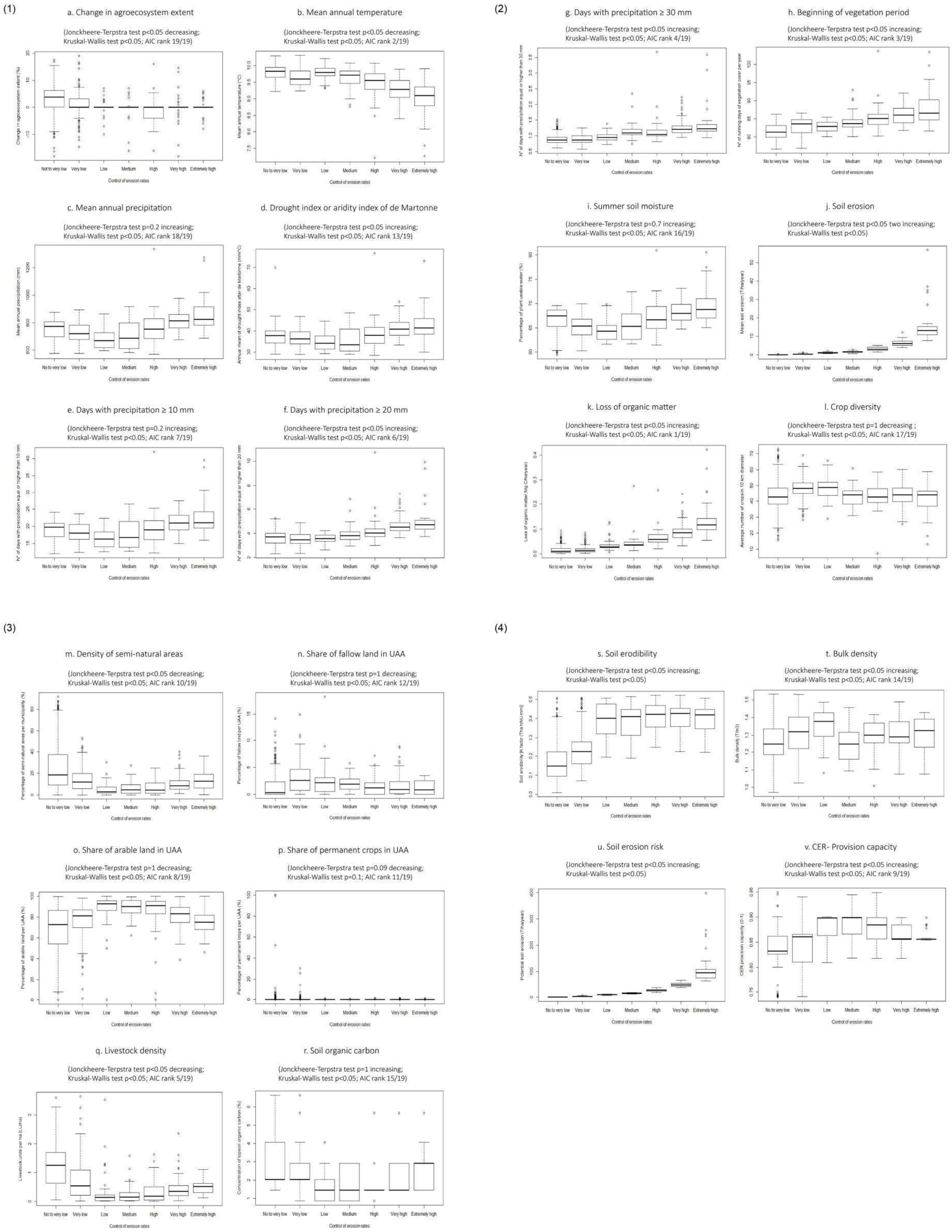

Fig 5. Relationships between the indicators of environmental pressures and condition, and the ecosystem service *control of erosion rates*.

https://doi.org/10.1371/journal.pone.0234288.g005

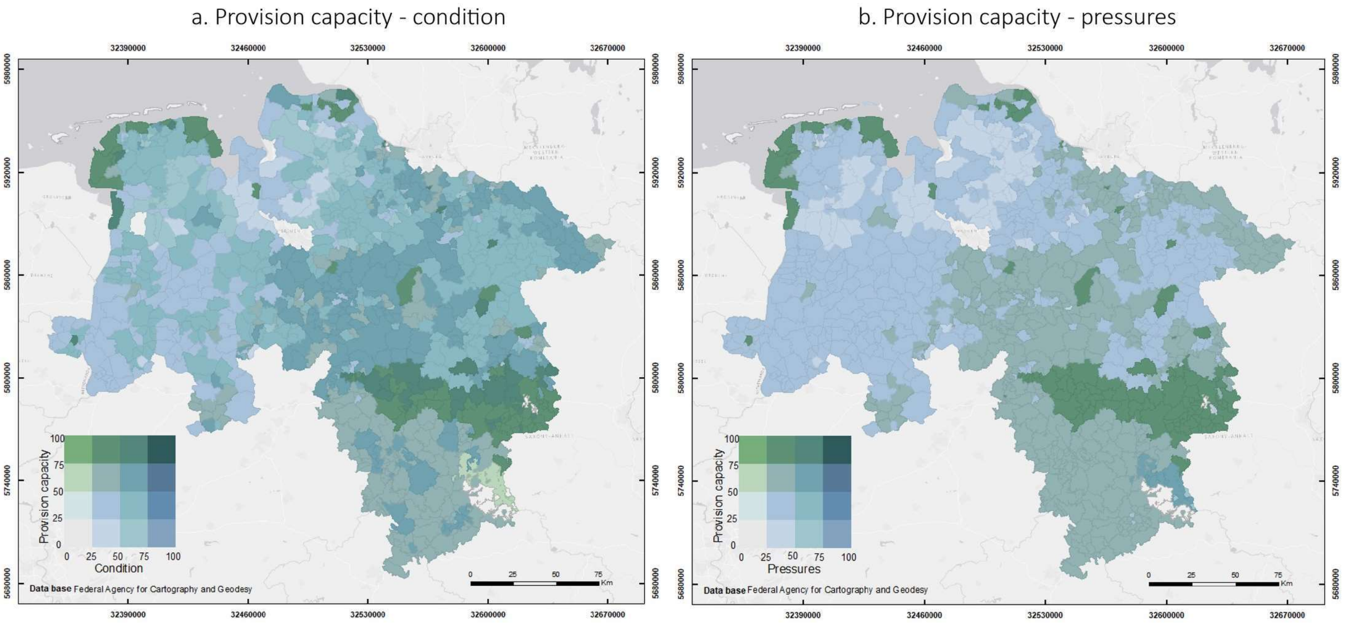

Fig 6. Spatial representation of the overlap between environmental pressures and condition, and ecosystem service provision capacity. (a) Overlap between provision capacity and condition. (b) Overlap between provision capacity and pressures. Darker areas represent high provision capacity and high level of condition in (a), or high provision capacity but high pressures in (b). Lighter areas represent lower provision capacity and low level of condition in (a) or lower provision capacity and low pressures in (b). Based on data from the administrative units from the German Federal Agency for Cartography and Geodesy © GeoBasis-DE / BKG (2017) [31]. (Larger maps in S3 File).

https://doi.org/10.1371/journal.pone.0234288.g006

capacity, condition and erosion risk were found in the district of Holzminden in the south (grey colour in the middle of the triangle). On the other hand, high provision capacity was evident in areas with medium-low condition levels and medium erosion risk (plum colours on the right side of the triangle). These mismatches were evident in the north-western and western regions. Medium provision capacity, low condition levels and high erosion risk (tan colour at the bottom of the triangle) were evident in the southern region, especially in the municipality of Wenzen, also in the district of Holzmiden.

The spatial distribution of provision capacity, pressures and condition shows that most of the study area had medium levels of pressures, medium condition levels and medium or high provision capacity (grey colours in the middle and on the right side of the triangle) (Fig 7b). The highest provision capacity was evident in the districts of Northeim and Goslar, but the condition levels in these areas were low and the pressures were medium. The lowest provision capacity occurred in the north-western region where the pressures had medium levels and condition had high levels (green colour on the left side of the triangle).

4. Discussion

The analysis based on the operational MAES framework proposed by Maes et al. [12] and the complied maps provide good results concerning the relationships between ecosystem condition and soil erosion regulating ecosystem service in Northern Germany. In the following, we will discuss the limitations of the applied indicators and provide recommendations for improved applications.

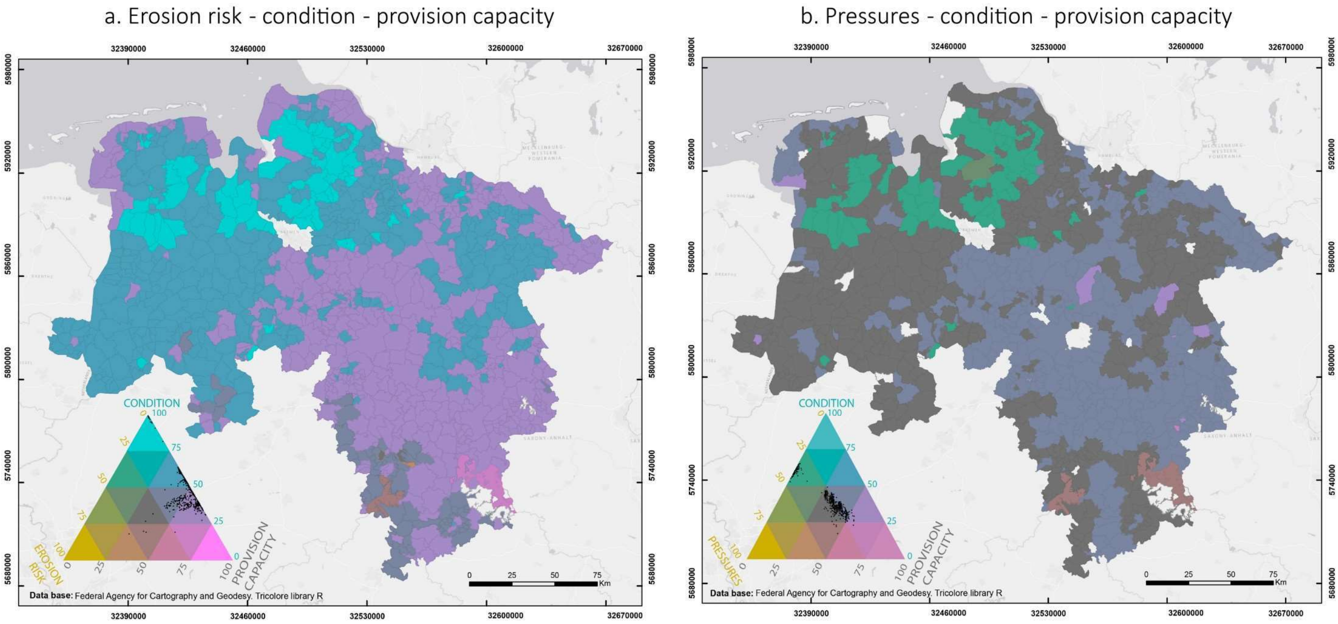

Fig 7. Spatial representation of the overlap between environmental pressures, condition, soil erosion risk and ecosystem service provision capacity. (a) Overlap between soil erosion risk, condition and provision capacity. (b) Overlap between pressures, condition and provision capacity. The colours and dots in the triangle show the distribution of the municipalities across the different indicators in percentage. Darker areas represent medium values of all the variables. Dots close to the lower right corner indicate high values of provision capacity. Dots close to the lower left corner indicate high erosion risk in (a) and high pressures in (b). Dots close to the upper corner indicate high condition levels. Based on data from the administrative units from the German Federal Agency for Cartography and Geodesy © GeoBasis-DE / BKG (2017) [31]. (Larger maps in S3 File).

https://doi.org/10.1371/journal.pone.0234288.g007

4.1. Limitations of the indicators for ecosystem condition and control of erosion rates

4.1.1 Data availability. One aim of this study was to test the conceptual framework for assessing ecosystem condition as suggested by the MAES working group on a regional level. As shown for the soil erosion narrative, the availability and accessibility of spatial and statistical data is a major obstacle when quantifying specific indicators and thus limits the application and explanatory power of such assessment studies. We identified different reasons for data *unavailability/inaccessibility*:

1. Absolute unavailability: Data do not exist.
2. Relative unavailability: Data are not freely available (inaccessibility).
3. Spatial mismatch: Data are only available or exist for highly aggregated areas.
4. Temporal mismatch: Data are not available for the study timeframe.

For some proposed indicators, data simply do not exist *(absolute unavailability)*. In our case study region, for instance, data on crop rotation and soil biodiversity were never collected. As the MAES working group proposes a high number of indicators of which some are very detailed such as fertilizers and pesticides use, this obstacle will certainly occur in other EU regions and for other indicators. Besides, data for model-testing on a regional case is usually not available due to the high costs and time required for measuring (in contrast to catchment or plot scales) [70].

The *relative unavailability* is due to the circumstance that environmental and spatial data collected by public authorities is not freely accessible. A trend exist to make data freely available [71, 72]. Regulations on the EU level promote free data availability [73], but compete with national (or federal state) data protection laws. For instance, in our case study region, some statistical data on the municipality level for crop rotation were not available due to German data protection regulations.

Spatial mismatches are well documented issues in environmental studies [74] and occur mainly from different spatial resolutions of geodata or mismatched administrative units [75]. Rescaling and aggregating data to one spatial resolution is a valid and common scientific practice [76–78] and was also implemented in this study for indicators related to climate, soil characteristics, and soil erosion. This makes data comparable without improving the information density of lower resolution data. However, the actual explanatory power of downscaled data is not as high as the provided spatial resolution would suggest. Spatial mismatches occur especially in studies that apply a high number of indicators. The proposed MAES framework and our case study are excellent examples of this issue.

Temporal mismatches are also a well-known problem [79]. They arise mainly from mismatching survey cycles and emerge often in combination with relative unavailability of data. Assessing the temporal variations of data is key to handle temporal mismatches [75]. This way, comparable timeframes can be identified and data can be included in environmental studies.

4.1.2 Environmental pressures indicators. *Change in ecosystem extent* was calculated, between the years 2006 and 2012. This is the timeframe of the CORINE land use change layers that was closest to the reference year we used to calculate the soil erosion. However, this two-year comparison leads to some limitations when assessing trends because this timespan does not provide enough information to draw precise conclusions. Additionally, the indicator, as it is proposed here, only shows the degree of change, but not the drivers of change. So when comparing it with an ecosystem service such as *control of erosion rates*, the simple percentage does not indicate whether the change is positive or negative for the provision of the service. The analysis of the causes of change is important to understand the relationships between ecosystem condition and ecosystem services. Nevertheless, such an analysis was beyond the scope of this study.

Climate indicators were calculated based on climate data for Germany for at least 30 years whenever they were available. There are, however, some uncertainties with these data, and also other data used in the study. These uncertainties are due to the different data collection methods and to interpolations and missing or erroneous observations. Also, for the climatic variables, the measurement network has changed over time and this affects the comparison of grid fields for the different years [46]. Another possible limitation of the *climate indicators* is that the MAES framework does not provide guidelines about which specific parameters should be used to assess climate change. We selected the most relevant indicators based on the possible influence on the ecosystem service *control of erosion rates*. This selection may seem arbitrary when looking at the general condition of the ecosystem or when making comparisons with other ecosystem services other than control of soil erosion. However, it is a valid approach to generate data on soil erosion rates on the regional scale of this case study.

Soil erosion was calculated for arable land only, without taking grasslands and forests into account. Additionally, this calculation was made only for water erosion, without including wind erosion, which is a major problem in the northern part of Lower Saxony. Therefore, the actual soil erosion might be higher than our results. Moreover, the calculation of soil erosion was made using the USLE equation that has some limitations, including the underestimation of the impact in thalwegs and gully erosion [45]. Furthermore, we used agricultural statistics and common assumptions about the effects of management practices and conservation measures for the estimation of soil erosion [49]. This approach is less precise and less explicit than the use of detailed monitoring data [25].

Loss of organic matter values were obtained from the ESDAC, which was considered a useful and sufficient approach for our study area. However, these values were originally calculated for the continental scale of Europe. This means that some uncertainties exist regarding the accuracy of the results for the regional level. Although the application of the CENTURY model at smaller scales is technically feasible, there is still a lack of input data [51]. The collection and processing of these data would help to improve the accuracy of the results. However, this is very time-consuming and also out of the scope of our study.

4.1.3 Ecosystem condition indicators. *Crop diversity* was estimated based on the number of crop species in a diameter of 10 km (see S1 File). Although this calculation differs from the one suggested in the MAES framework (N° of crops/10 km × 10 km), it is also an approximation that reflects crop diversity. However, similar to the indicator *change in ecosystem extent*, this indicator does not provide sufficient information to determine whether this number is favourable or not when comparing it with the ecosystem service *control of erosion rates*. For this, an indicator such as the types of crops in a specific area would be more useful, since some cultures are more prone to soil erosion than others. However, we did not use it in this study, since we aimed to test the feasibility of the proposed indicators.

The *density of semi-natural areas* was calculated in regard to the area of each municipality and not specifically in regard to the area of the agricultural land. This could overshadow the results since we did not determine the exact location of these elements, but instead estimated their proportion within the municipalities. Additionally, the lack of suitable metrics prevented us from calculating the shares of some semi-natural elements such as semi-natural grasslands, hedgerows and buffer strips that could have provided a different picture when it comes to identifying the role of semi-natural vegetation in the provision of ecosystem services such as *control of erosion rates*.

The indicators *share of fallow land*, *share of arable land and share of permanent crops in utilized agricultural areas* as well as *livestock density* were calculated based on official statistical data at the level of the municipalities. However, important factors such as the duration and the management of the fallow land can show different results regarding soil erosion in comparison to cultivated lands [68]. Nonetheless, the results could have been more precise and comparable with other indicators, if spatially explicit data were available for the study area. Furthermore, the bare number of livestock units per hectare does not provide sufficient information about the possible impact of livestock on the provision of ecosystem services, because the different types of livestock as well as their management (e.g. landless production systems vs. grassland-based) are not assessed by this indicator.

Soil Organic Carbon was calculated based on spatially explicit data on humus content in the topsoil and then using common conversion factors to obtain the concentration of soil organic carbon. These spatially explicit results were upscaled to the level of the municipalities to allow for the comparison between indicators. However, as with other indicators, this leads to uncertainties in the results, since the soil characteristics, in this case soil organic carbon, are not homogeneously distributed within the area of the municipalities.

Soil erodibility (K factor of the USLE) showed the same limitations as the calculated soil erosion described before. Furthermore, upscaling these values to the level of the municipalities can increase uncertainty. Taking into consideration that the contents of silt, sand, clay, and organic matter, as well as other parameters needed to calculate the K factor, usually vary within short distances, these generalizations can lead to less accurate results.

Bulk density values were obtained from the ESDAC, which is a useful and sufficient approach for our study, similar to the indicator loss of organic matter mentioned before.

However, bulk density shows the same limitations as the indicator loss of organic carbon as these values were also calculated for a continental scale. The processing of available regional data would improve the accuracy of the results, but this would be a very complicated and time-consuming approach. It is also worth noting the high annual variability of this indicator, which can be another source of uncertainty in this type of study.

4.1.4 Ecosystem service indicators. The indicator *potential soil loss* (erosion risk) was calculated by assuming that the whole arable land is bare soil, taking into account only natural soil erosion by water. The results aggregated to the municipalities provide an estimate of erosion risk and could show a general picture of the ecosystem condition, when combined with other indicators.

As mentioned in Section 3.1.3, low values of *prevented soil loss* also occurred in areas with low *soil erosion risk*, but this does not necessarily mean that the service supply is low. This shows that only calculating the *prevented soil loss* is insufficient to determine the actual ecosystem service supply [25].

The *provision capacity* indicator, which reflects the proportion of the potential soil loss that is mitigated by the ecosystem service *control of erosion rates*, allows us to identify the service supply and to possibly assess different management practices. However, as with other indicators, provision capacity was upscaled to the municipality level and some aspects—e.g. the presence and distance to watercourses, relief characteristics like thalwegs, presence of tramlines and wheel tracks, as well as management measures, which affect the provision capacity [24, 80] —could not be identified and hence the results are less accurate than they would be with spatially explicit data.

4.2. Relationships between agroecosystem condition and control of erosion rates

Since the adoption of the EU Biodiversity Strategy to 2020, the mapping and assessment of European ecosystems and their services has increased [81–83]. However, understanding the interdependencies between biodiversity, ecosystem functioning, and ecosystem services is still a major challenge [5, 6]. Although several studies provide evidence of the positive relationships between biodiversity, natural capital, and ecosystem services [2, 84], there is no consensus on what these links are and how they concretely operate [85].

Although we were not able to establish the causalities among the indicators with the correlation analysis, we observed some strong relations. Almost all the environmental pressure indicators are strongly related to the ecosystem service *control of erosion rates*, except for the indicators *change in ecosystem extent*, *mean annual precipitation*, *days with precipitation equal or higher than 10 mm*, and *summer soil moisture*. Regarding the ecosystem condition indicators, we identified that five out of nine indicators are strongly related to control of erosion rates. The indicators *crop diversity*, *share of fallow and arable land*, and *soil organic carbon* do not show a strong correlation. As expected, there is a positive correlation between the ecosystem service indicators *potential soil loss* and *provision capacity*, and the *control of erosion rates*. This analysis should be perceived with caution, due to the limitations mentioned above.

4.3. Recommended land management measures to reduce soil erosion

Based on the overlaps presented in Section 3.2.2., we identified municipalities as priority areas in which the risk of soil erosion is medium or high, the provision capacity is low, and the condition levels are low. Our data identified these *problematic* municipalities in the district of

Northeim in the southern region of Lower Saxony. For these areas, it is necessary to implement measures to reduce the impact of pressures, improve the ecosystem condition, and soil conservation. Also, analyses on local scale must be carried out and measures must take into account site-specific characteristics such as soil, crop varieties, soil degradation, and farming practices [19]. When looking at the types of crops in Northeim, we identified cereals, grasslands, oilseeds, pastures for extensive grazing, and fodder plants (maize) as the main agricultural land uses. These crops may have effects on soil degradation and erosion problems due to excessive tillage and crop residue removal [45]. Therefore, the implementation of conservation farming is essential to solve these problems because it can reduce soil erosion by ensuring the protection of the soil surface with residue retention and increased water infiltration.

The techniques applied in conservation farming include permanent soil cover with crop residues, which protects the ground surface and provides organic material, thereby improving soil quality. Other methods are the growth of diverse crop species in the same field and crop rotation, especially crops such as legumes and grasses [36]. Conservation farming also involves minimum soil disturbance that has positive effects on biotic soil activity and leads to increased stability of soil aggregates [37]. Another practice that could be applied to guarantee soil conservation in vulnerable areas is agroforestry. This practice integrates trees with animals or crops or both, increasing the fixation of nitrogen and the return of organic matter to the soil, preserving the fertility and structure of the soil [36]. All these measures impact soil condition and hence agroecosystems condition. Therefore, indicators that address these measures should be taken into account when assessing agroecosystems. The indicators proposed by the MAES working group, as they are implemented in this study, are not able to fully address agroecosystem condition relevant for soil protection and soil erosion prevention. Indicators that analyse the effect of crop species on soil erosion, the use of cover crops and other soil conservation measures should be included in the list of proposed indicators.

4.4. Potential for MAES/policy implementation

The normalization presented in Section 2.9 and Figs 6 and 7 aimed to facilitate the comparison between indicators and to visualize the relations between ecosystem condition, environmental pressures, erosion risk, and provision capacity. It is worth highlighting that these representations are somewhat arbitrary and other combinations and overlaps of indicators could provide different results. Furthermore, the results presented here come from a methodological study, aiming at testing an existing framework and respective indicators. These results do not yet have the potential to be used for policy decisions or implement the measures described above to improve the condition or reduce the pressures on agroecosystems, at least not without more detailed, and if possible, spatially explicit data. Our maps should provide a general idea of the environmental pressures, ecosystem condition, soil erosion risk, and the *control of erosion rates* provision capacity in Lower Saxony. These maps should also raise awareness for areas where special attention should be paid to avoid or mitigate ecosystem degradation.

Composite indicators can be used to provide insights on environmental condition, as well as sustainability, quality of life, and economy [86]. Such indicators have been useful in policy analysis and public communication because they seem to be straightforward comprehensible for the general public [87]. However, we did not develop a composite indicator for three main reasons: First, it could add an extra ambiguity to the results. Second, the suggested indicators would not be sufficient to build a trustworthy index, since threshold levels that would help to determine the overall condition have not been defined for all the indicators. Third, it was not the aim of the study to come up with a composite indicator.

5. Conclusions

This is- to our knowledge—the first study that tests the MAES framework and indicators for the assessment of the condition of agroecosystems in a regional scale case study. Our study also analyses the relationships between ecosystem condition and the provision of a selected ecosystem service, in particular, *control of erosion rates*. This assessment can identify the suitability of these indicators, check the data availability for respective indicator quantification and describe ecosystem condition on a regional scale.

Although we were not able to establish clear causalities among the indicators, our results identified positive, negative, and no significant correlations between the different pressures and condition indicators, and *control of erosion rates* despite their limitations and data availability. The idea behind the MAES framework is to show the general condition of an ecosystem in the context of ecosystem services supply. However, when looking at the relationships between ecosystem condition and ecosystem services, we observed that not all proposed indicators are suitable to explain to what extent agroecosystems can provide specific ecosystem services. Condition indicators on crop management and soil conservation measures, which are directly linked to the ecosystem service *control of erosion rates*, are missing in the list of indicators proposed in the MAES framework. Additionally, if indicators are to be applied in national or regional scale studies, it is important to consider that trend and high-resolution data are not always available in sufficient quality and resolution. These limitations may undermine the results and hence their comparability with other regions.

Future research should also assess other ecosystem services/ecosystem services bundles provided by agroecosystems. These assessments would facilitate the identification of synergies and trade-offs, both between ecosystem services and between ecosystem condition parameters that may have a different degree of influence on ecosystem services. Besides, a more precise definition of reference conditions, although complicated for agroecosystems, is essential to provide more accurate information on the condition of the ecosystem, which should lead to better policy and management decisions.

Supporting information

S1 File. Quantification of indicators.
(PDF)

S2 File. Indicators per municipality.
(XLSX)

S3 File. Maps.
(PDF)

S1 Table. Results of tests per indicator.
(XLSX)

Acknowledgments

This study is partly based on the outcomes of the Lower Saxonian soil erosion monitoring programme funded by the Lower Saxonian State Authority for Mining, Energy, and Geology (LBEG). We wish to thank the staff of the Joint Research Centre of the European Commission who gave conceptual support. We also thank Tobias Kreklow and Jan Bierwirth for their collaboration in processing the data and Angie Faust for checking the language of the manuscript.

Author Contributions

Conceptualization: Paula Rendon, Bastian Steinhoff-Knopp, Benjamin Burkhard.

Data curation: Paula Rendon, Bastian Steinhoff-Knopp, Philipp Saggau.

Investigation: Paula Rendon, Bastian Steinhoff-Knopp, Philipp Saggau.

Methodology: Paula Rendon, Bastian Steinhoff-Knopp, Benjamin Burkhard.

Supervision: Benjamin Burkhard.

Writing – original draft: Paula Rendon, Bastian Steinhoff-Knopp.

Writing – review & editing: Paula Rendon, Bastian Steinhoff-Knopp, Philipp Saggau, Benjamin Burkhard.

References

1. Potschin-Young M, Haines-Young R, Heink U, Jax K. OpenNESS Glossary [Internet]. 2016.: http://www.openness-project.eu/glossary
2. Maseyk FJF, Mackay AD, Possingham HP, Dominati EJ, Buckley YM. Managing Natural Capital Stocks for the Provision of Ecosystem Services. Conserv Lett [Internet]. 2017; 10(2):211–20. Available from: https://conbio.onlinelibrary.wiley.com/doi/full/10.1111/conl.12242
3. Harrison P, Berry PM, Simpson G, Haslett JR, Blicharska M, Bucur M, et al. Linkages between biodiversity attributes and ecosystem services: A systematic review. Ecosyst Serv [Internet]. 2014; 9:191–203. Available from: https://www.sciencedirect.com/science/article/pii/S2212041614000576
4. Bastian O. The role of biodiversity in supporting ecosystem services in Natura 2000 sites. Ecol Indic [Internet]. 2013; 24:12–22. Available from: https://www.sciencedirect.com/science/article/abs/pii/S1470160X12002154
5. Schneiders A, van Daele T, Landuyt VW, Van Reeth W. Biodiversity and ecosystem services: complementary approaches for ecosystem management? Ecol Indic [Internet]. 2012; 21:123–33. Available from: https://www.sciencedirect.com/science/article/abs/pii/S1470160X11001920
6. Rendon P, Erhard M, Maes J, Burkhard B. Analysis of trends in mapping and assessment of ecosystem condition in Europe. Ecosyst People [Internet]. 2019; 15(1):156–72. Available from: https://www.tandfonline.com/doi/full/10.1080/26395916.2019.1609581
7. United Nations. Report of the Inter-Agency and Expert Group on Sustainable Development Goal Indicators E/CN.3/2016/2/Rev.1. New York; 2016.
8. European Commission. Our life insurance, our natural capital: an EU biodiversity strategy to 2020 [Internet]. 2011. http://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/EP_resolution_april2012.pdf
9. European Parliament, Council of the European Union. Decision No 1386/2013/EU of the European Parliament and of the Council of 20 November 2013 on a General Union Environment Action Programme to 2020 "Living well, within the limits of our planet" [Internet]. Official Journal of the European Union. 2013. http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32013D1386&from=EN
10. Biodiversity Information System for Europe (BISE). Mapping and Assessment of Ecosystems and their Services (MAES) [Internet]. 2019 [cited 2019 Nov 28]. https://biodiversity.europa.eu/maes
11. Maes J, Teller A, Erhard M, Liquete C, Braat L, Egoh B, et al. Mapping and Assessment of Ecosystems and their Services: An analytical framework for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020. 2013.
12. Maes J, Teller A, Erhard M, Grizzetti B, Barredo JI, Paracchini ML, et al. Mapping and Assessment of Ecosystems and their Services: An analytical framework for ecosystem condition. Luxembourg; 2018.
13. European Commission. Food & Farming: Focus on Land [Internet]. 2015. https://ec.europa.eu/info/sites/info/files/food-farming-fisheries/events/documents/oulook-conference-2015-brochure-land_en.pdf
14. German Environment Agency. Environment and agriculture [Internet]. Dessau-Roßlau; 2018. www.umweltbundesamt.de
15. Power AG. Can Ecosystem Services Contribute to Food Security? In: Potschin-Young M, Haines-Young R, Fish R, Turner RK, editors. Routledge Handbook of Ecosystem Services. London and New York: Routledge; 2016. p. 491–500.

16. Burkhard B, Hotes S, Wiggering H. Agro(eco)system services-supply and demand from fields to society. Land [Internet]. 2016; 5(2):3–6. Available from: https://www.mdpi.com/2073-445X/5/2/9/htm

17. Zhang W, Ricketts TH, Kremen C, Carney K, Swinton SM. Ecosystem services and dis-services to agriculture. Ecol Econ [Internet]. 2007; 64(2):253–60. Available from: https://www.sciencedirect.com/science/article/abs/pii/S0921800907001462

18. Power AG. Ecosystem services and agriculture: Tradeoffs and synergies. Philos Trans R Soc B Biol Sci [Internet]. 2010; 365(1554):2959–71. Available from: https://royalsocietypublishing.org/doi/full/10.1098/rstb.2010.0143 PMID: 20713396

19. Virto I, Imaz M, Fernández-Ugalde O, Gartzia-Bengoetxea N, Enrique A, Bescansa P. Soil Degradation and Soil Quality in Western Europe: Current Situation and Future Perspectives. Sustainability [Internet]. 2015; 7(1):313–65. Available from: https://www.mdpi.com/2071-1050/7/1/313

20. Wiggering H, Weißhuhn P, Burkhard B. Agrosystem services: An additional terminology to better understand ecosystem services delivered by agriculture. Landsc Online [Internet]. 2016; 49(1):1–15. Available from: https://landscape-online.org/index.php/lo/article/view/LO.201649

21. Burkhard B, Bethwell C, Daedlow K, Reckling M, Sattler C, Zander P. Towards an enhanced indication of provisioning ecosystem services in agroecosystems. In: Landscape 2018 Frontiers of agricultural landscape research [Internet]. Berlin: ZALF; 2018. p. 126–7. https://communications.ext.zalf.de/sites/land2018/SiteCollectionDocuments/Landscape2018_Book_of_Abstracts.pdf#page=233

22. Bordt M. A summary and review of approaches, data, tools and results of existing and previous ecosystem accounting work on spatial units, scaling and aggregation methods and approaches. Vol. 0, 2.a.2. New York; 2015.

23. Peterson EE, Cunningham SA, Thomas M, Collings S, Bonnett GD, Harch B. An assessment framework for measuring agroecosystem health. Ecol Indic [Internet]. 2017; 79:265–75. Available from: https://www.sciencedirect.com/science/article/pii/S1470160X17301772

24. Steinhoff-Knopp B, Burkhard B. Soil erosion by water in Northern Germany: long-term monitoring results from Lower Saxony. Catena [Internet]. 2018; 165:299–309. Available from: https://www.sciencedirect.com/science/article/abs/pii/S0341816218300626

25. Steinhoff-Knopp B, Burkhard B. Mapping Control of Erosion Rates: Comparing Model and Monitoring Data for Croplands in Northern Germany. One Ecosyst [Internet]. 2018; 3:e26382. Available from: https://oneecosystem.pensoft.net/articles.php?id=26382

26. Panagos P, Borrelli P, Poesen J, Ballabio C, Lugato E, Meusburger K, et al. The new assessment of soil loss by water erosion in Europe. Environ Sci Policy [Internet]. 2015; 54:438–47. Available from: https://www.sciencedirect.com/science/article/pii/S1462901115300654

27. Burkhard B, Santos-Martín F, Nedkov S, Maes J. An operational framework for integrated Mapping and Assessment of Ecosystems and their Services (MAES). One Ecosyst [Internet]. 2018; 3:e22831. Available from: https://oneecosystem.pensoft.net/articles.php?id=22831

28. Helmholtz-Zentrum Geesthacht, North German Climate Monitor, Helmholtz-Zentrum Geesthacht. North German Climate Monitor [Internet]. 2019 [cited 2019 Feb 18]. https://www.norddeutscher-klimamonitor.de

29. Lower Saxonian Ministry of Food Agriculture and Consumer Protection. Agrarland Nummer 1 [Internet]. 2019. https://www.ml.niedersachsen.de/themen/landwirtschaft/landwirtschaft-in-niedersachsen-4513.html

30. European Environment Agency. Land Cover Change (LCC) 2006–2012 [Internet]. 2019 [cited 2019 Jul 31]. https://land.copernicus.eu/pan-european/corine-land-cover/lcc-2006-2012?tab=metadata

31. German Federal Agency for Cartography and Geodesy. Administrative areas 1: 250,000 (levels), as of 01.01. (VG250 01.01.) [Internet]. 2017 [cited 2019 Jun 17]. https://gdz.bkg.bund.de/index.php/default/digitale-geodaten/verwaltungsgebiete/verwaltungsgebiete-1-250-000-ebenen-stand-01-01-vg250-ebenen-01-01.html

32. Rinot O, Levy GJ, Steinberger Y, Svoray T, Eshel G. Soil health assessment: A critical review of current methodologies and a proposed new approach. Sci Total Environ [Internet]. 2019; 648:1484–91. Available from: https://www.sciencedirect.com/science/article/pii/S0048969718332224 PMID: 30340293

33. Kibblewhite MG, Ritz K, Swift M. Soil health in agricultural systems. Philos Trans R Soc B Biol Sci [Internet]. 2008; 363(1492):685–701. Available from: https://doi.org/10.1098/rstb.2007.2178 PMID: 17785275

34. Thomsen M, Faber JH, Sorensen PB. Soil ecosystem health and services—Evaluation of ecological indicators susceptible to chemical stressors. Ecol Indic [Internet]. 2012; 16:67–75. Available from: https://www.sciencedirect.com/science/article/pii/S1470160X11001427

35. Špulerová J, Petrovič F, Mederly P, Mojses M, Izakovičcová Z. Contribution of traditional farming to ecosystem services provision: Case studies from Slovakia. Land [Internet]. 2018; 7(2):74. Available from: https://www.mdpi.com/2073-445X/7/2/74

36. Morgan R. Soil erosion and conservation. John Wiley and Sons Ltd; 2009.
37. Palm C, Blanco-Canqui H, DeClerck F, Gatere L, Grace P. Conservation agriculture and ecosystem services: An overview. Agric Ecosyst Environ [Internet]. 2014; 187:87–105. Available from: http://dx.doi.org/10.1016/j.agee.2013.10.010
38. Bundesgesetzblatt. German Federal Soil Protection Act [Internet]. 1998 p. 502–10. http://www.gesetze-im-internet.de/bbodschg/index.html
39. Bundesgesetzblatt. German Federal Soil Protection and Contaminated Sites Ordinance (BBodSchV) [Internet]. 1999 p. 1554–82. http://www.gesetze-im-internet.de/bbodschv/index.html
40. European Commission. Thematic Strategy for Soil Protection (Comunication) [Internet]. Brussels; 2006. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A52006DC0231
41. Altieri M. Agroecology: the science of natural resource management for poor farmers in marginal environments. Agric Ecosyst Environ [Internet]. 2002; 93:1–24. Available from: https://www.sciencedirect.com/science/article/abs/pii/S0167880902000853
42. European Environment Agency. CORINE Land Cover (CLC) [Internet]. 2012 [cited 2019 Aug 23]. https://land.copernicus.eu/pan-european/corine-land-cover
43. Tóth G, Gardi C, Bódis K, Ivits É, Aksoy E, Jones A, et al. Continental-scale assessment of provisioning soil functions in Europe. Ecol Process [Internet]. 2013; 2(1):32. Available from: https://link.springer.com/article/10.1186/2192-1709-2-32
44. Lal R. Restoring soil quality to mitigate soil degradation. Sustain [Internet]. 2015; 7(5):5875–95. Available from: https://www.mdpi.com/2071-1050/7/5/5875
45. Boardman J, Poesen J. Soil erosion processes. In: Boardman J, Poesen J, editors. Soil erosion in Europe. John Wiley and Sons Ltd; 2007. p. 480.
46. Deutscher Wetterdienst. Climate Data Center (CDC OpenData) [Internet]. 2018 [cited 2018 Oct 8]. https://opendata.dwd.de/climate_environment/CDC/
47. Martonne DE. Traite de Geographie Physique: 3 tomes. Paris; 1925.
48. Huber S, Prokop G, Arrouays D, Banko G, Bispo A, Jones RJA, et al. Environmental assessment of soil for monitoring: volume I indicators and criteria [Internet]. Vol. I, Office for the Official Publications of the European Communities. 2008. http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.397.1624&rep=rep1&type=pdf
49. Saggau P, Bug J, Gocht A, Krure K. Aktuelle Bodenerosionsgefährdung durch Wind und Wasser in Deutschland. Bodenschutz [Internet]. 2017; 4(17):120–5. Available from: https://doi.org/10.37307/j.1868-7741.2017.04.04
50. ESDAC. Pan-European SOC stock of agricultural soils [Internet]. 2014 [cited 2018 Oct 10]. https://esdac.jrc.ec.europa.eu/content/pan-european-soc-stock-agricultural-soils
51. Lugato E, Paustian K, Panagos P, Jones A, Borrelli P. Quantifying the erosion effect on current carbon budget of European agricultural soils at high spatial resolution. Glob Chang Biol [Internet]. 2016; 22(5):1976–84. Available from: https://onlinelibrary.wiley.com/doi/full/10.1111/gcb.13198 PMID: 26679897
52. Lugato E, Panagos P, Bampa F, Jones A, Montanarella L. A new baseline of organic carbon stock in European agricultural soils using a modelling approach. Glob Chang Biol. 2014 Jan; 20(1):313–26. https://doi.org/10.1111/gcb.12292 PMID: 23765562
53. SLA—LEA. Landentwicklung und Agrarförderung Niedersachsen LEA—Portal [Internet]. 2019 [cited 2019 Apr 3]. https://sla.niedersachsen.de/landentwicklung/LEA/
54. Gocht A, Röder N, Meyer-Borstel H. Thünen-Atlas: Landwirtschaftliche Nutzung (1999–2010) [Internet]. Braunschweig; 2014. https://gdi.thuenen.de/lr/agraratlas
55. Kibblewhite MG, Jones RJA, Montanarella L, Baritz R, Huber S, Arrouays D, et al. Environmental Assessment of Soil for Monitoring Volume VI: Soil Monitoring System for Europe [Internet]. Vol. EUR 23490, JRC Scientific and Technical Reports. 2008. http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.397.1333&rep=rep1&type=pdf
56. Bundesanstalt für Geowissenschaften und Rohstoffe (BGR). Bodenübersichtskarte der Bundesrepublik Deutschland 1:1.000.000 [Internet]. 2019 [cited 2019 Apr 20]. https://produktcenter.bgr.de/terraCatalog/DetailResult.do?fileIdentifier=A95A723E-1274-4601-9E60-27079436F1F3
57. Auerswald K, Elhaus D. Ableitung der Bodenerodierbarkeit K anhand der Bodenart. Bodenschutz [Internet]. 2013; 4(13):109–13. Available from: https://www.researchgate.net/profile/Karl_Auerswald/publication/259331462_Ableitung_der_Bodenerodierbarkeit_K_anhand_der_Bodenart/links/565f1f9a08aefe619b284d96.pdf
58. Ballabio C, Panagos P, Monatanarella L. Mapping topsoil physical properties at European scale using the LUCAS database. Geoderma [Internet]. 2016; 261:110–23. Available from: http://dx.doi.org/10.1016/j.geoderma.2015.07.006

59. ESDAC. Topsoil physical properties for Europe (based on LUCAS topsoil data) [Internet]. 2016 [cited 2019 Jul 17]. https://esdac.jrc.ec.europa.eu/content/topsoil-physical-properties-europe-based-lucas-topsoil-data

60. Guerra C, Pinto-Correia T, Metzger MJ. Mapping Soil Erosion Prevention Using an Ecosystem Service Modeling Framework for Integrated Land Management and Policy. Ecosystems [Internet]. 2014; 17 (5):878–89. Available from: https://link.springer.com/article/10.1007/s10021-014-9766-4

61. Akaike H. An information criterion. Math Sci. 1976;(14):5–7.

62. Kruskal WH, Wallis W. Use of Ranks in One-Criterion Variance Analysis. J Am Stat Assoc [Internet]. 1952;(47):583–621. Available from: https://doi.org/10.1080/01621459.1952.10483441

63. Terpstra TJ. The asymptotic normality and consistency of kendall's test against trend, when ties are present in one ranking. Indag Math [Internet]. 1952; 55:327–33. Available from: http://dx.doi.org/10.1016/S1385-7258(52)50043-X

64. Jonckheere A. A Distribution-Free k-sample Test Against Ordered Alternatives. Biometrika [Internet]. 1954; 41(1):133–45. Available from: https://www.jstor.org/stable/2333011

65. Team RStudio. RStudio: Integrated Development for R. RStudio, Inc. [Internet]. Boston; 2015. http://www.rstudio.com/.

66. Solte J, Tesfai M, Øygarden L, Kværnø S, Kizer J, Verheijen F, et al. Soil threats in Europe: deliverable 2.1 RECARE project. European Commission DG Joint Research Centre. 2016.

67. Istanbulluoglu E, Bras RL. On the dynamics of soil moisture, vegetation, and erosion: Implications of climate variability and change. Water Resour Res [Internet]. 2006; 42(6):1–17. Available from: https://agupubs.onlinelibrary.wiley.com/doi/epdf/10.1029/2005WR004113

68. Nielsen DC, Calderón FJ. Fallow Effects on Soil. In: Hatfield JL, Sauer TJ, editors. Soil Management: Building a Stable Base for Agriculture [Internet]. John Wiley and Sons Ltd; 2011. p. 287–300. https://acsess.onlinelibrary.wiley.com/doi/pdf/10.2136/2011.soilmanagement.c19

69. García-Feced C, Weissteiner CJ, Baraldi A, Paracchini ML, Maes J, Zulian G, et al. Semi-natural vegetation in agricultural land: European map and links to ecosystem service supply. Agron Sustain Dev [Internet]. 2014; 35(1):273–83. Available from: https://link.springer.com/content/pdf/10.1007/s13593-014-0238-1.pdf

70. Batista PVGV, Davies J, Silva MLNLLN, Quinton JN. On the evaluation of soil erosion models: Are we doing enough? Earth-Science Rev [Internet]. 2019; 197(June):102898. Available from: https://doi.org/10.1016/j.earscirev.2019.102898

71. Burkhard B, Maes J, Geneletti D, Stoev P, Penev L. One Ecosystem: Innovation in ecology and sustainability research publishing. One Ecosyst [Internet]. 2016; 1:1–7. Available from: https://oneecosystem.pensoft.net/article/9255/

72. Brunsdon C. Quantitative methods I: Reproducible research and quantitative geography. Prog Hum Geogr [Internet]. 2016; 40(5):687–96. Available from: https://journals.sagepub.com/doi/10.1177/0309132515599625

73. Craglia M, Annoni A. INSPIRE: An innovative approach to the development of spatial data infrastructures in Europe. In: Research and theory in advancing spatial data infrastructure concepts. 2007. p. 93–105.

74. Henle K, Potts SG, Kunin WE, Matsinos YG, Similä J, Pantis JD, et al. Scaling in ecology and biodiversity conservation [Internet]. Scaling in Ecology and Biodiversity Conservation. 2014. e1169 p. http://www.scales-project.net/files/DOWNLOAD2/AB_article_1169.pdf

75. Hou Y, Burkhard B, Müller F. Uncertainties in landscape analysis and ecosystem service assessment. J Environ Manage [Internet]. 2013; 127:S117–31. Available from: http://dx.doi.org/10.1016/j.jenvman.2012.12.002 PMID: 23291281

76. Reid W V, Berkes F, Wilbanks T, Capistrano D. Bridging scales and knowledge systems: concepts and applications in ecosystem assessment. Washington D.C: Island Press; 2006. 351 p.

77. Malinga R, Gordon LJ, Jewitt G, Lindorg R. Mapping ecosystem services across scales and continents —A review. Ecosyst Serv [Internet]. 2015; 13:57–63. Available from: https://www.sciencedirect.com/science/article/abs/pii/S2212041615000078

78. Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron DR, et al. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front Ecol Environ [Internet]. 2009; 7(1):4–11. Available from: https://esajournals.onlinelibrary.wiley.com/doi/full/10.1890/080023

79. Willemen L. It's about time: Advancing spatial analyses of ecosystem services and their application. Ecosyst Serv [Internet]. 2020; 44(February):101125. Available from: https://doi.org/10.1016/j.ecoser.2020.101125

80. Saggau P, Kuhwald Duttmann. Integrating Soil Compaction Impacts of Tramlines Into Soil Erosion Modelling: A Field-Scale Approach. Soil Syst [Internet]. 2019; 3(3):51. Available from: https://www.mdpi.com/2571-8789/3/3/51
81. Santos-Martín F, Geneletti D, Burkhard B. Mapping and assessing ecosystem services: Methods and practical applications. One Ecosyst [Internet]. 2019; 4:1–8. Available from: https://oneecosystem.pensoft.net/article/35904/download/pdf/
82. Santos-Martín F, Viinikka A, Mononen L, Brander LM, Vihervaara P, Liekens I, et al. Creating an operational database for ecosystems services mapping and assessment methods. One Ecosyst [Internet]. 2018; 3. Available from: https://oneecosystem.pensoft.net/article/26719/
83. Geneletti D, Adem Esmail B, Cortinovis C. Identifying representative case studies for ecosystem services mapping and assessment across Europe. One Ecosyst [Internet]. 2018; 3(ii). Available from: https://oneecosystem.pensoft.net/article/25382/
84. Smith A, Harrison P, Pérez-Soba M, Archaux F, Blicharska M, Egoh B, et al. How natural capital delivers ecosystem services: A typology derived from a systematic review. Ecosyst Serv [Internet]. 2017; 26:111–26. Available from: https://www.sciencedirect.com/science/article/abs/pii/S2212041617300086
85. Bastian O, Haase D, Grunewald K. Ecosystem properties, potentials and services—The EPPS conceptual framework and an urban application example. Ecol Indic [Internet]. 2012; 21:7–16. Available from: https://www.sciencedirect.com/science/article/abs/pii/S1470160X11000677
86. Pissourios IA. An interdisciplinary study on indicators: A comparative review of quality-of-life, macroeconomic, environmental, welfare and sustainability indicators. Ecol Indic [Internet]. 2013; 34:420–7. Available from: http://dx.doi.org/10.1016/j.ecolind.2013.06.008
87. Joint Research Centre- European Commission. Handbook on Constructing Composite Indicators: Methodology and User Guide [Internet]. OECD publishing; 2008. https://www.oecd-ilibrary.org/economics/handbook-on-constructing-composite-indicators-methodology-and-user-guide_9789264043466-en

4

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

Rendon, P., Steinhoff-Knopp, B., & Burkhard, B.

Ecosystem Services (2022), 53, 101387

Ecosystem Services 53 (2022) 101387

Contents lists available at ScienceDirect

Ecosystem Services

journal homepage: www.elsevier.com/locate/ecoser

Full Length Article

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

Paula Rendon [a,*], Bastian Steinhoff-Knopp [a], Benjamin Burkhard [a,b]

[a] Institute of Physical Geography & Landscape Ecology, Leibniz University of Hannover, Schneiderberg 50, 30167 Hannover, Germany
[b] Leibniz Centre for Agricultural Landscape Research (ZALF) e. V. Eberswalder Straße 84, 15374 Müncheberg, Germany

A R T I C L E I N F O

Keywords:
Ecosystem status
Erosion control
Pressures
Condition indicators
Agricultural soils
European Biodiversity Strategy
MAES

A B S T R A C T

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

* Corresponding author.
E-mail addresses: rendon@phygeo.uni-hannover.de (P. Rendon), steinhoffknopp@phygeo.uni-hannover.de (B. Steinhoff-Knopp), burkhard@phygeo.unihannover.de (B. Burkhard).

https://doi.org/10.1016/j.ecoser.2021.101387
Received 29 April 2021; Received in revised form 30 September 2021; Accepted 13 November 2021

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

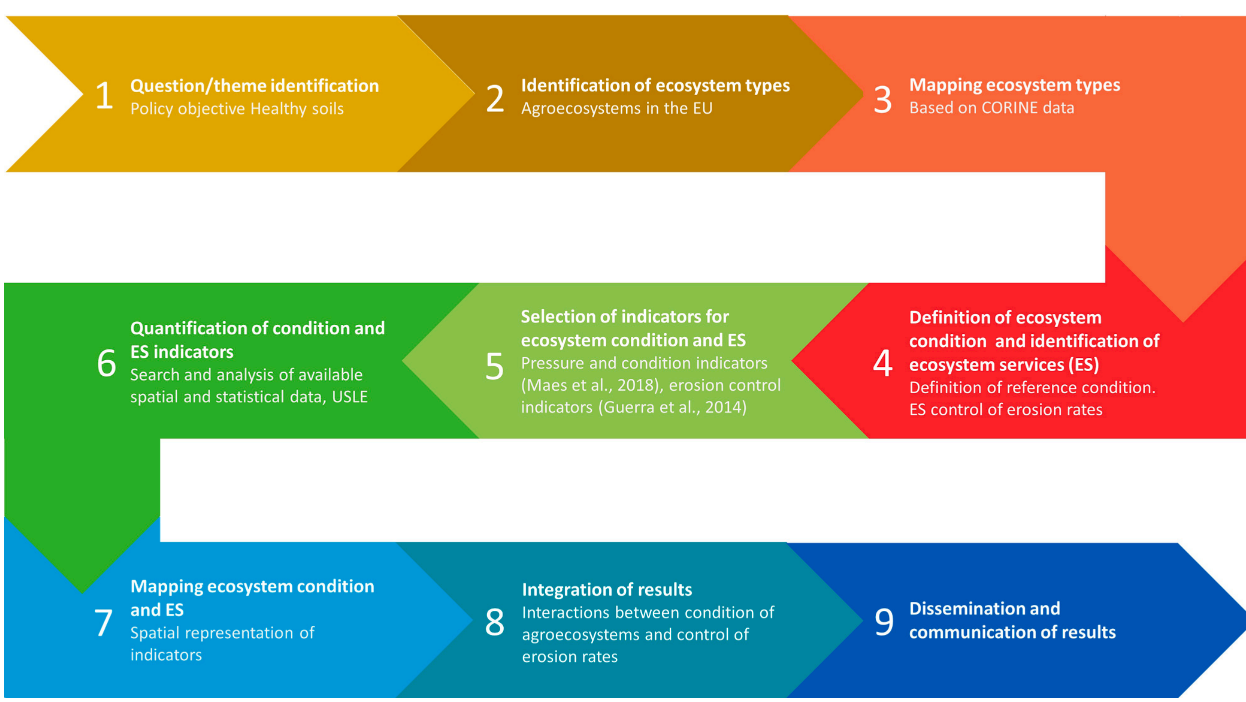

Fig. 1. Framework for integrated mapping and assessment of ecosystems and their services. (Step 9 is not included in this study). Based on Burkhard et al. (2018).

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 ≥ 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 ≥ 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
							(ESDAC, 2014b; Panagos et al., 2014a, 2012)
	Bulk density	BD	Weight of soil per cubic meter.	t m^{-3}	500 m	2015	(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).

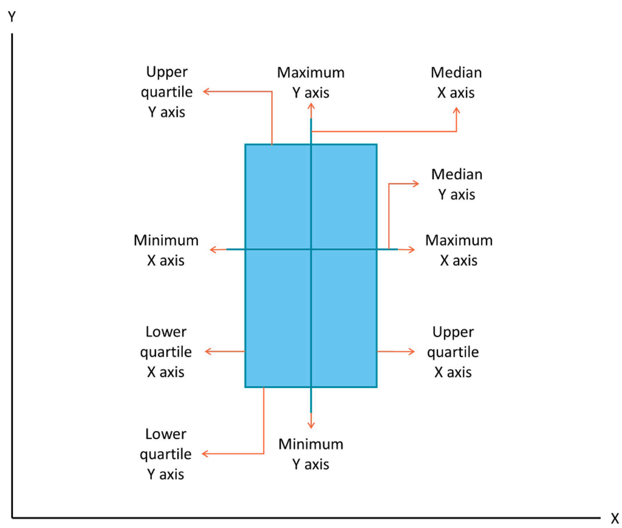

Fig. 2. Example 2D Box plot.

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

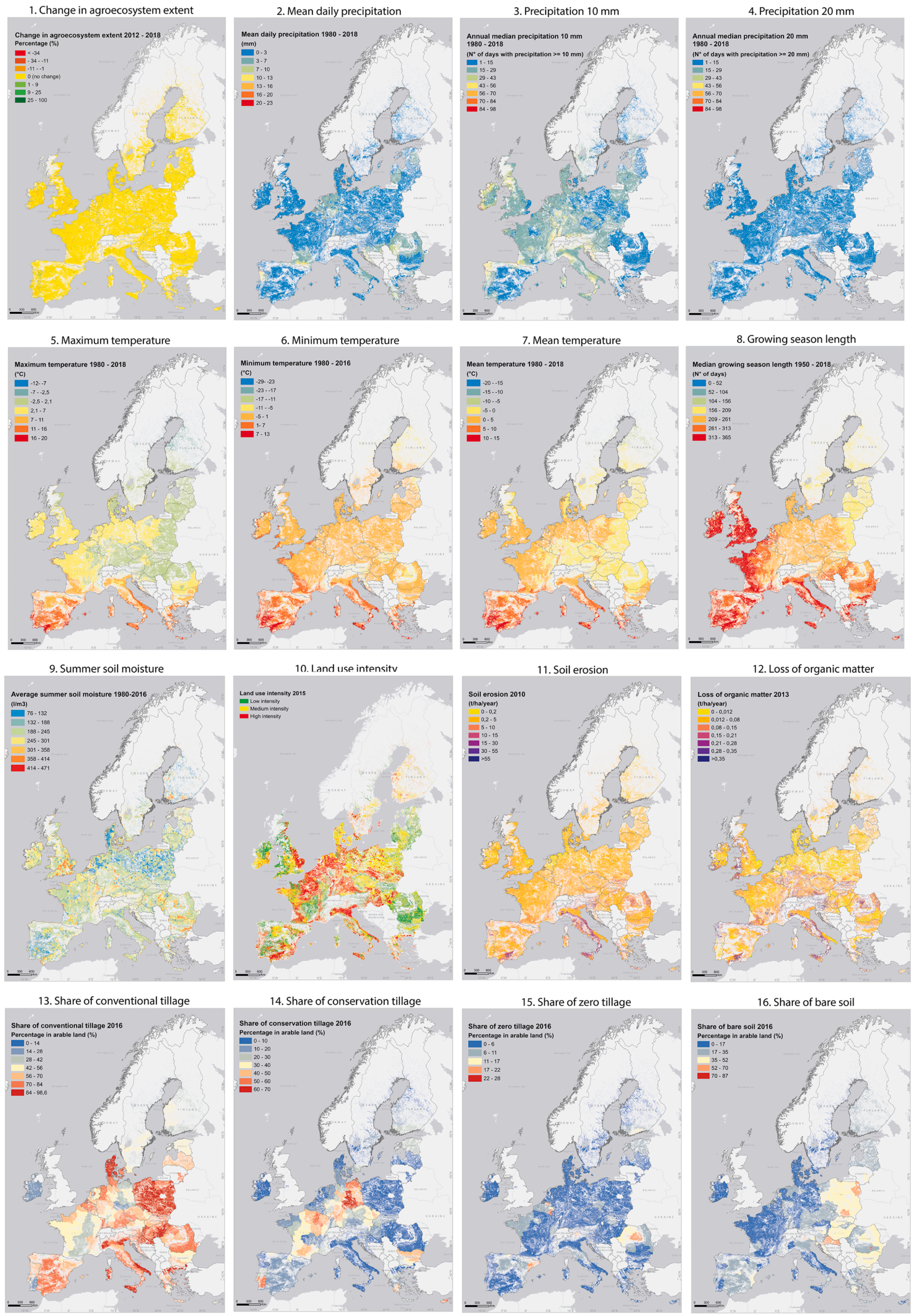

Fig. 3. Maps of indicators of environmental pressures, ecosystem condition and control of erosion rates in the EU (larger maps are provided in the Supplementary Information S4).

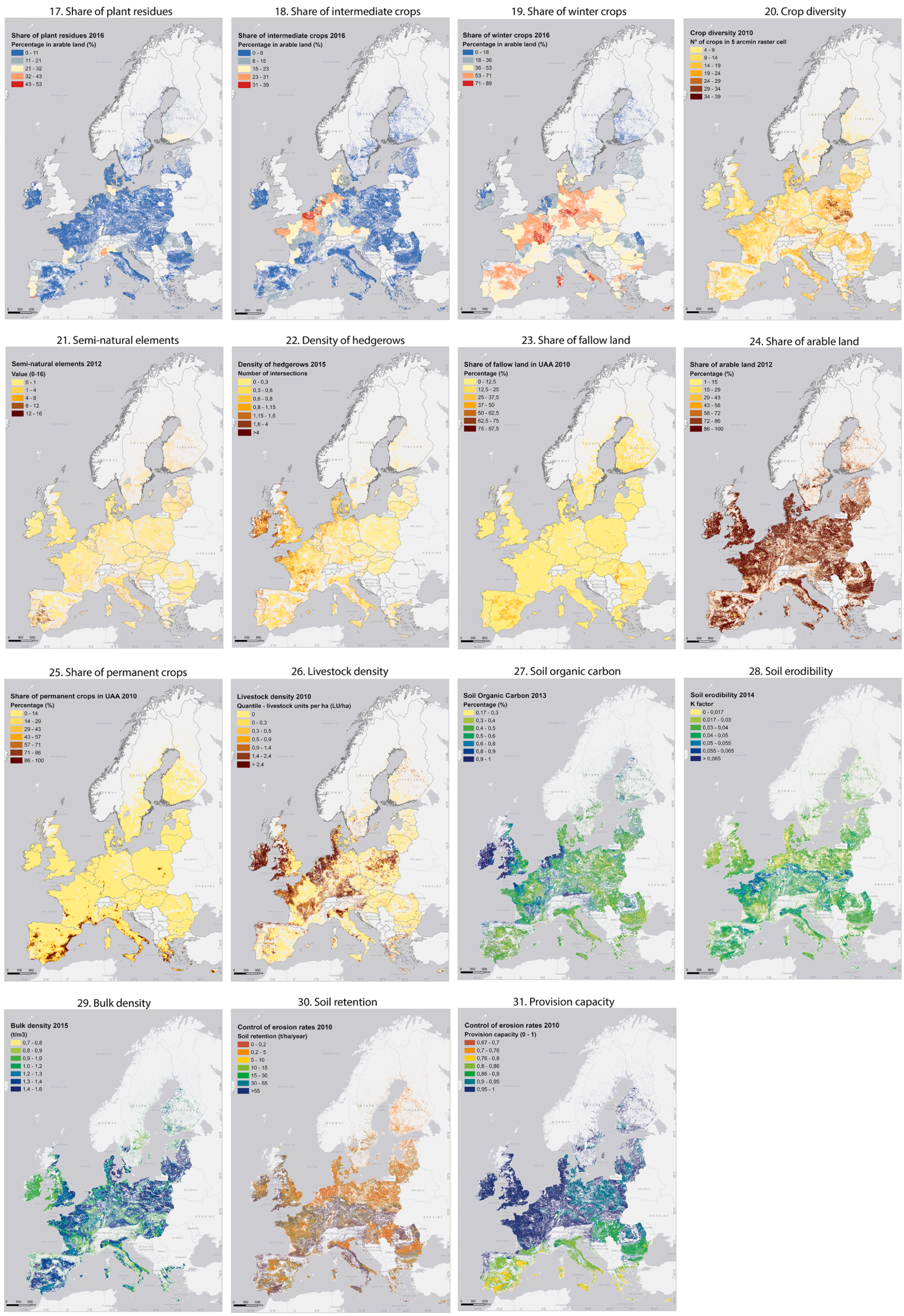

Fig. 3. (*continued*).

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

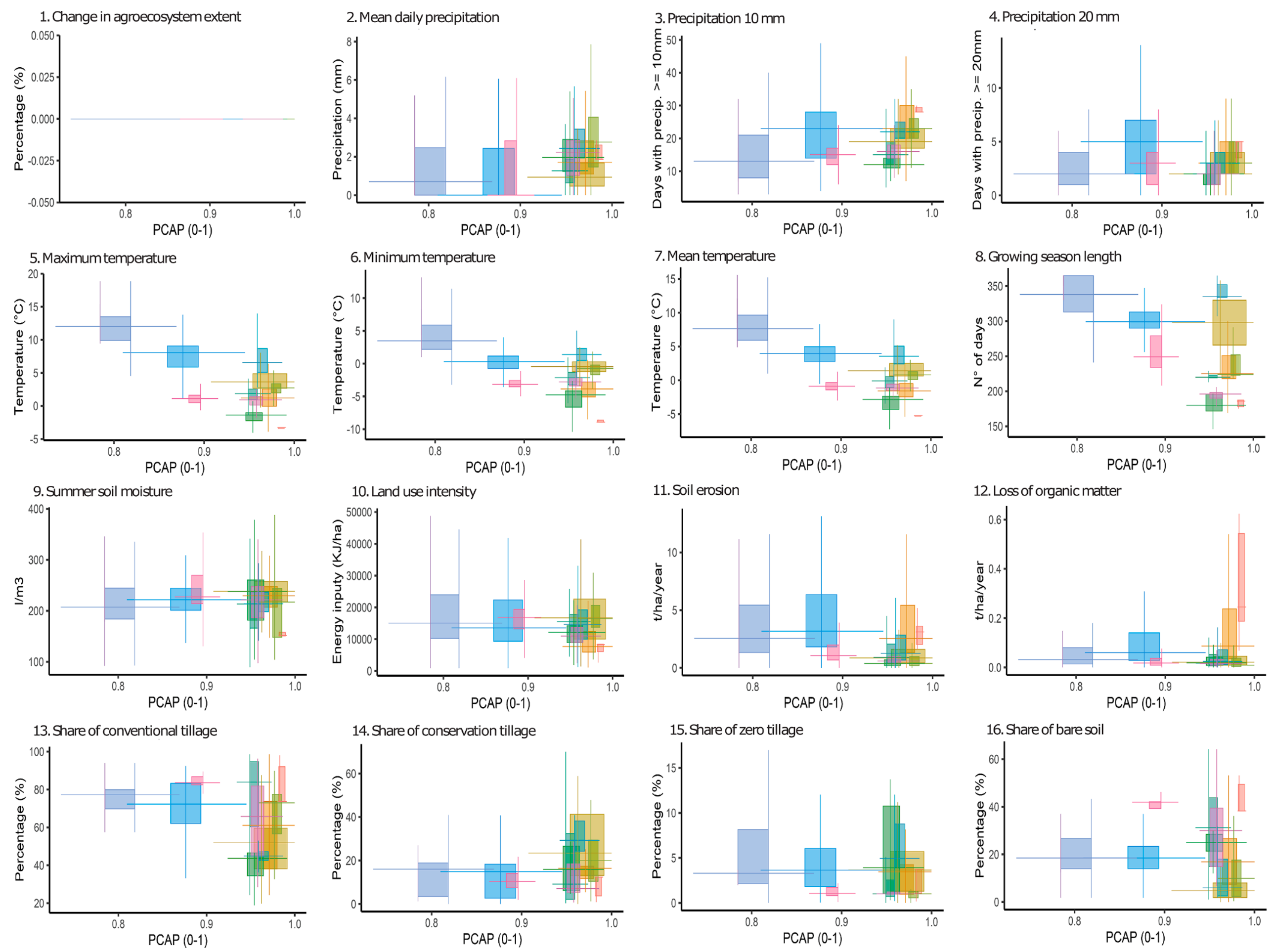

Fig. 4. Relationships between the indicators of environmental pressures and condition and the capacity of the ecosystem to control erosion rates (PCAP) per environmental zone (*EZ)*. ALN: Alpine North, ALS: Alpine South, ATC: Atlantic Central, ATN: Atlantic North, BOR: Boreal, CON: Continental, LUS: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

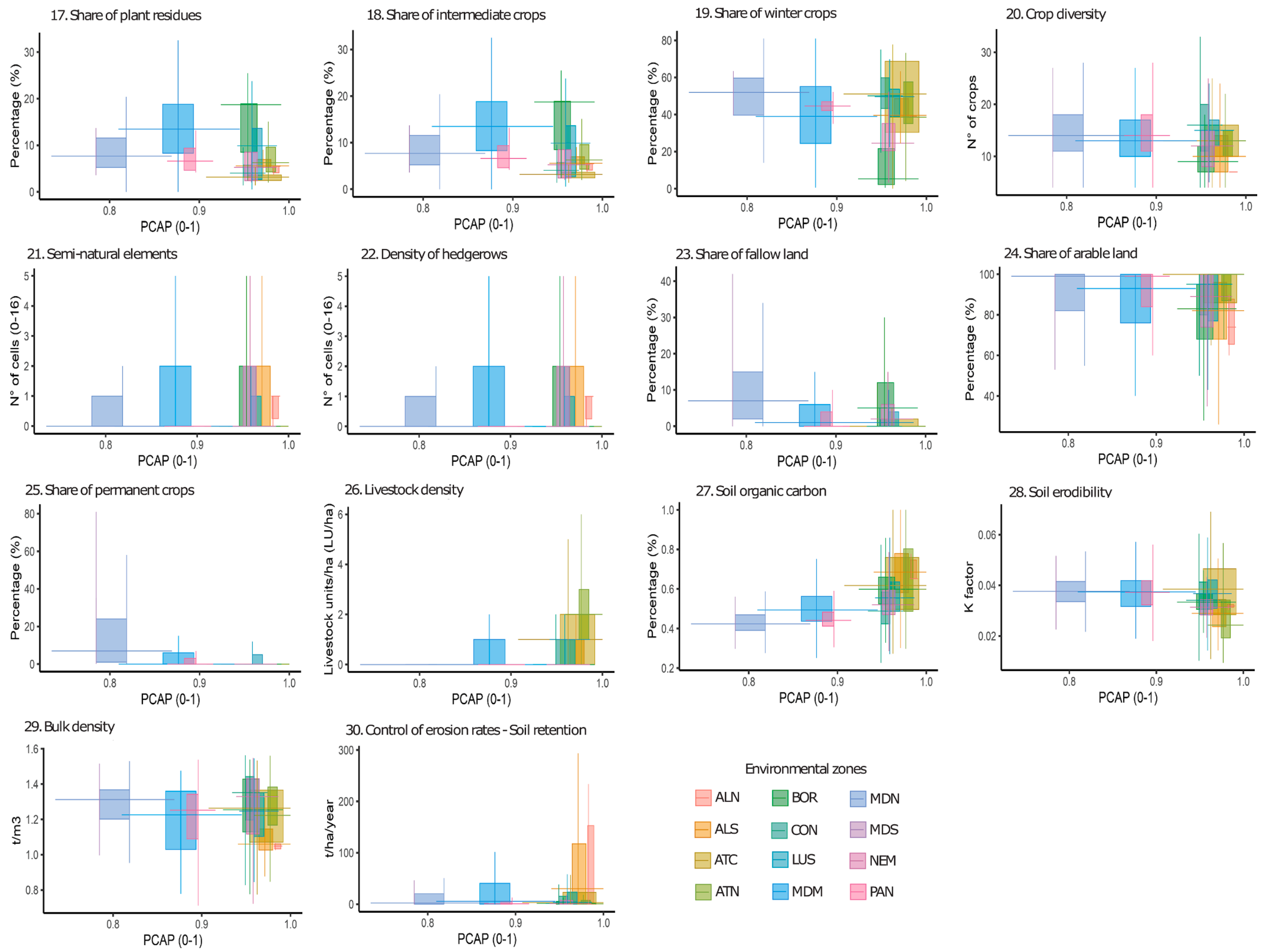

Fig. 4. (*continued*).

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.

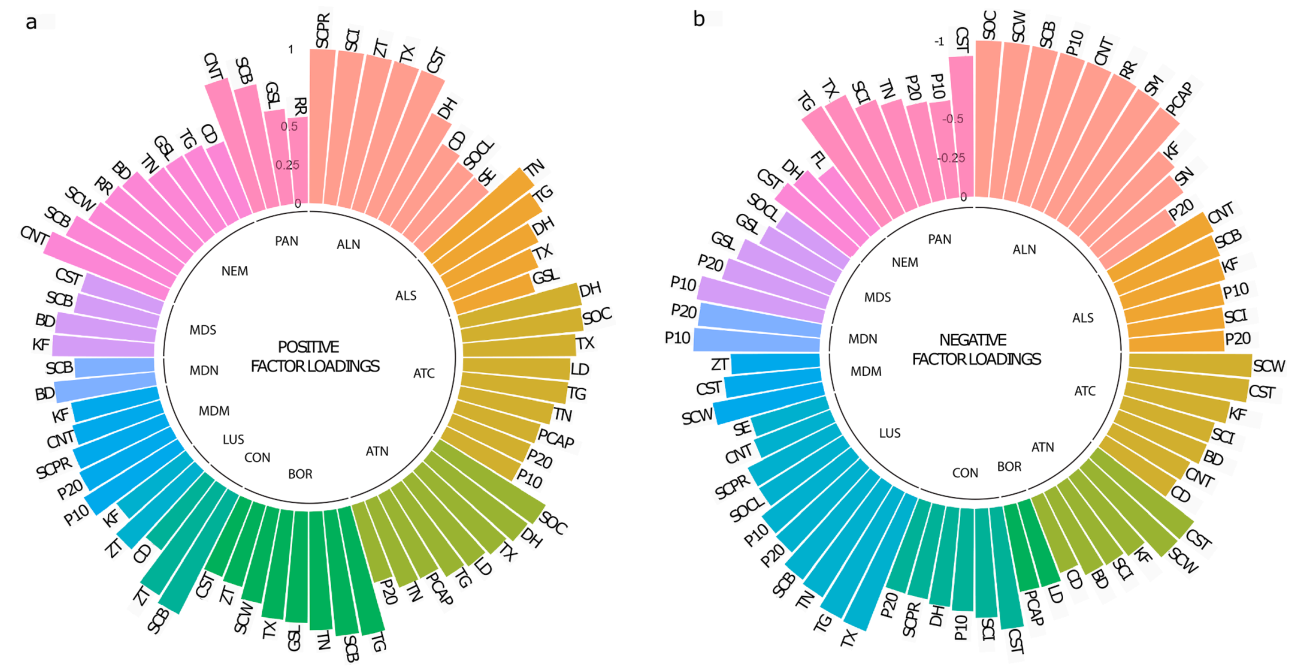

Fig. 5. High significance of indicators based on the PCA per Environmental Zone (*EZ*): (a) significant positive factor loadings (>0.5), (b) significant negative factor loadings (<-0.5). ALN: Alpine North, ALS: Alpine South, ATC: Atlantic Central, ATN: Atlantic North, BOR: Boreal, CON: Continental, LUS: 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 *EZ*s 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 *EZs* 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 *EZs*, 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 *EZs* 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 *EZs* 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 *EZs* and the typical management practices. There are broad differences in the climatic conditions, growing seasons, agricultural practices and crops and crop rotations within the *EZs*. 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 *EZs* (Panagos et al., 2015a).

It is also worth noting that the values within the *EZs* 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 *EZs*. 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 *EZs* 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.

Funding

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

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.

Acknowledgements

We wish to thank the staff of the Joint Research Centre of the European Commission, especially Joachim Maes, Maria Luisa Paracchini, Carlo Rega and Sara Vallecillo, who provided datasets, and conceptual and technical support. We also thank Angie Faust for checking the language of the manuscript and the anonymous reviewers for their valuable feedback on an earlier version of this paper.

Appendix A. Supplementary data

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

References

Auerswald, K., Fiener, P., Martin, W., Elhaus, D., 2014. Use and misuse of the K factor equation in soil erosion modeling: An alternative equation for determining USLE nomograph soil erodibility values. Catena 118, 220–225. https://doi.org/10.1016/j.catena.2014.01.008.

Ballabio, C., Panagos, P., Monatanarella, L., 2016. Mapping topsoil physical properties at European scale using the LUCAS database. Geoderma 261, 110–123. https://doi.org/10.1016/j.geoderma.2015.07.006.

Balmford, A., Fisher, B., Green, R.E., Naidoo, R., Strassburg, B., Turner, R.K., Rodrigues, A.S.L., 2011. Bringing ecosystem services into the real world : an operational framework for assessing the economic consequences of losing wild nature. Environ. Resour. Econ. 48, 161–175. https://doi.org/10.1007/s10640-010-9413-2.

Bernués, A., Rodríguez-Ortega, T., Ripoll-Bosch, R., Alfnes, F., Moreira, F., 2014. Socio-Cultural and Economic Valuation of Ecosystem Services Provided by Mediterranean Mountain Agroecosystems. PLoS One 9 (7), e102479. https://doi.org/10.1371/journal.pone.0102479.

Boardman, J., Poesen, J., 2007. Soil erosion processes. In: Boardman, J., Poesen, J. (Eds.), Soil Erosion in Europe. John Wiley and Sons Ltd.

Borrelli, P., Robinson, D.A., Fleischer, L.R., Lugato, E., Ballabio, C., Alewell, C., Meusburger, K., Modugno, S., Schütt, B., Ferro, V., Bagarello, V., Oost, K.V., Montanarella, L., Panagos, P., 2017. An assessment of the global impact of 21st century land use change on soil erosion. Nat. Commun. 8 (1) https://doi.org/10.1038/s41467-017-02142-7.

Borrelli, P., Robinson, D.A., Panagos, P., Lugato, E., Yang, J.E., Alewell, C., Wuepper, D., Montanarella, L., Ballabio, C., 2020. Land use and climate change impacts on global soil erosion by water (2015–2070). Proc. Natl. Acad. Sci. U. S. A. 117 (36), 21994–22001. https://doi.org/10.1073/pnas.2001403117.

Burkhard, B., de Groot, R.S., Costanza, R., Seppelt, R., Jørgensen, S.E., Potschin-Young, M., 2012. Solutions for sustaining natural capital and ecosystem services. Ecol. Indic. 21, 1–6. https://doi.org/10.1016/j.ecolind.2012.03.008.

Burkhard, B., Kandziora, M., Hou, Y., Müller, F., 2014. Ecosystem service potentials, flows and demands – Concepts for spatial localisation, indication and quantification. Landsc. Online 34, 1–32. https://doi.org/10.3097/LO.201434.

Burkhard, B., Santos-Martín, F., Nedkov, S., Maes, J., 2018. An operational framework for integrated Mapping and Assessment of Ecosystems and their Services (MAES). One Ecosyst. 3, e22831 https://doi.org/10.3897/oneeco.3.e22831.

Ciscar, J.-C., Iglesias, A., Feyen, L., Szabo, L., Van Regemorter, D., Amelung, B., Nicholls, R., Watkiss, P., Christensen, O.B., Dankers, R., Garrote, L., Goodess, C.M., Hunt, A., Moreno, A., Richards, J., Soria, A., 2011. Physical and economic consequences of climate change in Europe. Proc. Natl. Acad. Sci. U.S.A. 108 (7), 2678–2683. https://doi.org/10.1073/pnas.1011612108.

Copernicus, 2019. Copernicus Climate Change Service [WWW Document]. URL https://climate.copernicus.eu/index.php/ (accessed 7.31.19).

Cornes, Richard C., van der Schrier, Gerard, van den Besselaar, Else J.M., Jones, Philip D., 2018. An ensemble version of the E-OBS temperature and precipitation data sets. J. Geophys. Res. Atmos. 123 (17), 9391–9409. https://doi.org/10.1029/2017JD028200.

Doran, John W., Zeiss, Michael R., 2000. Soil health and sustainability: managing the biotic component of soil quality. Appl. Soil Ecol. 15 (1), 3–11.

ESDAC, 2014a. Pan-European SOC stock of agricultural soils [WWW Document]. URL https://esdac.jrc.ec.europa.eu/content/pan-european-soc-stock-agricultural-soils (accessed 10.10.18).

ESDAC, 2014b. Soil Erodibility (K- Factor) High Resolution dataset for Europe [WWW Document]. URL https://esdac.jrc.ec.europa.eu/content/soil-erodibility-k-factor-high-resolution-dataset-europe (accessed 7.22.19).

ESDAC, 2015. Soil erosion by water (RUSLE2015) [WWW Document]. URL https://esdac.jrc.ec.europa.eu/content/soil-erosion-water-rusle2015 (accessed 8.6.19).

ESDAC, 2016. Topsoil physical properties for Europe (based on LUCAS topsoil data) [WWW Document]. URL https://esdac.jrc.ec.europa.eu/content/topsoil-physical-properties-europe-based-lucas-topsoil-data (accessed 7.17.19).

European Commission, 2011. Our life insurance, our natural capital: an EU biodiversity strategy to 2020.

European Commission, 2016. LUISA - Territorial modelling platform [WWW Document]. URL https://ec.europa.eu/jrc/en/luisa (accessed 6.18.20).

European Commission, 2017. Modernising and simplifying the CAP. Background Document: Climate and Environmental challenges facing EU agriculture and rural areas. Brussels.

European Commission, 2019. The common agricultural policy at a glance [WWW Document]. URL https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/cap-glance_en (accessed 8.2.19).

European Commission, 2020a. EU Biodiversity Strategy for 2030. Bringing nature back into our lives. Brussels.

European Commission, 2020b. A Farm to Fork Strategy for a fair, healthy and environmentally-friendly food system. Brussels.

European Commission, 2021. Healthy soils – new EU soil strategy - Consultation [WWW Document]. URL https://ec.europa.eu/info/law/better-regulation/have-your-say/initiatives/12634-Healthy-soils-new-EU-soil-strategy/public-consultation (accessed 4.28.21).

European Environment Agency, 2006. Land accounts for Europe 1990-2000 Towards integrated land and ecosystem accounting.

European Environment Agency, 2012. CORINE Land Cover (CLC) [WWW Document]. URL https://land.copernicus.eu/pan-european/corine-land-cover (accessed 8.23.19).

European Environment Agency, 2019a. The European environment-state and outlook 2020. Knowledge for transition to a sustainable Europe. https://doi.org/10.2800/96749.

European Environment Agency, 2019b. Land Cover Change (LCC) 2012-2018 [WWW Document]. URL https://land.copernicus.eu/pan-european/corine-land-cover/lcc-2006-2012?tab=metadata (accessed 7.31.19).

EUROSTAT, 2010a. Cropping patterns [WWW Document]. Farm Struct. Surv. 2010. URL https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agri-environmental_indicator_-_cropping_patterns (accessed 8.6.19).

EUROSTAT, 2010b. Livestock patterns [WWW Document]. Farm Struct. Surv. 2010. URL https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agri-environmental_indicator_-_livestock_patterns (accessed 8.6.19).

EUROSTAT, 2019a. Glossary: Arable land [WWW Document]. URL https://ec.europa.eu/eurostat/statistics-explained/index.php/Glossary:Arable_land (accessed 8.6.19).

EUROSTAT, 2019b. Glossary:Utilised agricultural area (UAA) [WWW Document]. URL https://ec.europa.eu/eurostat/statistics-explained/index.php/Glossary:Utilised_agricultural_area_(UAA) (accessed 6.25.20).

EUROSTAT, 2019c. Agri-environmental indicator - tillage practices [WWW Document]. URL https://ec.europa.eu/eurostat/statistics-explained/index.php/Agri-environmental_indicator_-_tillage_practices (accessed 6.25.20).

EUROSTAT, 2019d. Agri-environmental indicator - soil erosion [WWW Document]. URL https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agri-environmental_indicator_-_soil_erosion (accessed 8.6.19).

EUROSTAT, 2020. Glossary:Agricultural area (AA) [WWW Document]. URL https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Glossary:Agricultural_area_(AA) (accessed 6.25.20).

Felipe-Lucia, María R., Soliveres, Santiago, Penone, Caterina, Fischer, Markus, Ammer, Christian, Boch, Steffen, Boeddinghaus, Runa S., Bonkowski, Michael, Buscot, François, Fiore-Donno, Anna Maria, Frank, Kevin, Goldmann, Kezia, Gossner, Martin M., Hölzel, Norbert, Jochum, Malte, Kandeler, Ellen, Klaus, Valentin H., Kleinebecker, Till, Leimer, Sophia, Manning, Peter, Oelmann, Yvonne, Saiz, Hugo, Schall, Peter, Schloter, Michael, Schöning, Ingo, Schrumpf, Marion, Solly, Emily F., Stempfhuber, Barbara, Weisser, Wolfgang W., Wilcke, Wolfgang, Wubet, Tesfaye, Allan, Eric, 2020. Land-use intensity alters networks between biodiversity, ecosystem functions, and services. Proc. Natl. Acad. Sci. U. S. A. 117 (45), 28140–28149. https://doi.org/10.1073/pnas.2016210117.

García-Feced, Celia, Weissteiner, Christof Jakob, Baraldi, Andrea, Paracchini, Maria Luisa, Maes, Joachim, Zulian, Grazia, Kempen, Markus, Elbersen, Berien, Pérez-Soba, Marta, 2015. Semi-natural vegetation in agricultural land: European map and links to ecosystem service supply. Agron. Sustain. Dev. 35 (1), 273–283. https://doi.org/10.1007/s13593-014-0238-1.

Guerra, Carlos A., Pinto-Correia, Teresa, Metzger, Marc J., 2014. Mapping Soil erosion prevention using an ecosystem service modeling framework for integrated land management and policy. Ecosystems 17 (5), 878–889. https://doi.org/10.1007/s10021-014-9766-4.

Hatziiordanou, L., Fitoka, E., Hadjicharalampous, E., Eleftheria Votsi, N., Palaskas, D., Abdul Malak, D., 2019. Indicators for mapping and assessment of ecosystem condition and of the ecosystem service habitat maintenance in support of the EU biodiversity strategy to 2020. One Ecosyst. 4 https://doi.org/10.3897/oneeco.4.e32704.

Joint Research Centre (JRC), 2015. Green lines for the EU [WWW Document].

Kirkby, M.J., Jones, R., Irvine, B., Gobin, A., Govers, G., Cerdan, O., van Rompaey, A., Le Bissonnais, Y., Daroussin, J., King, D., Montanarella, L., Grimm, M., Vieillefont, V., Puigdefábregas, J., Boer, M., Kosmas, C., Yassoglou, N., Tsara, M., Mantel, S., van Lynden, G., Huting, J., 2004. Pan-European Soil Erosion Risk Assessment: The PESERA map.

Kirkby, M.J., Irvine, B.J., Jones, R.J.A., Govers, G., Boer, M., Cerdan, O., Daroussin, J., Gobin, A., Grimm, M., Le Bissonnais, Y., Kosmas, C., Mantel, S., Puigdefabregas, J., van Lynden, G., 2008. The PESERA coarse scale erosion model for Europe. I. - Model rationale and implementation. Eur. J. Soil Sci. 59, 1293–1306. https://doi.org/10.1111/j.1365-2389.2008.01072.x.

Kokkoris, I.P., Dimopoulos, P., Xystrakis, F., Tsiripidis, I., 2018. National scale ecosystem condition assessment with emphasis on forest types in Greece. One Ecosyst. 3, e25434 https://doi.org/10.3897/oneeco.3.e25434.

Kurnik, Blaž, Kajfež-Bogataj, Lučka, Horion, Stephanie, 2015. An assessment of actual evapotranspiration and soil water deficit in agricultural regions in Europe. Int. J. Climatol. 35 (9), 2451–2471. https://doi.org/10.1002/joc.2015.35.issue-910.1002/joc.4154.

Lugato, Emanuele, Panagos, Panos, Bampa, Francesca, Jones, Arwyn, Montanarella, Luca, 2014. A new baseline of organic carbon stock in European agricultural soils using a modelling approach. Glob. Chang. Biol. 20 (1), 313–326. https://doi.org/10.1111/gcb.12292.

Lugato, Emanuele, Paustian, Keith, Panagos, Panos, Jones, Arwyn, Borrelli, Pasquale, 2016. Quantifying the erosion effect on current carbon budget of European agricultural soils at high spatial resolution. Glob. Chang. Biol. 22 (5), 1976–1984. https://doi.org/10.1111/gcb.13198.

Maes, J., Fabrega, N., Zulian, G., Barbosa, A., Vizcaino, P., Ivits, É., Polce, C., Vandecasteele, I., Marí Rivero, I., Guerra, C., Perpiña Castillo, C., Vallecillo, S., Baranzelli, C., Barranco, R., Batista e Silva, F., Jacobs-Crisoni, C., Trombetti, M., Lavalle, C., 2015. Mapping and Assessment of Ecosystems and their Services: Trends in ecosystems and ecosystem services in the European Union between 2000 and 2010.

Maes, J., Teller, A., Erhard, M., Condé, S., Vallecillo, S., Barredo, J.I., Paracchini, M.L., Abdul Malak, D., Trombetti, M., Vigiak, O., Zulian, G., Addamo, A.M., Grizzetti, B., Somma, F., Hagyo, A., Vogt, P., Polce, C., Jones, A., Marin, A.I., Ivits, E., Mauri, A., Rega, C., Czúcz, B., Ceccherini, G., Pisoni, E., Ceglar, A., De Palma, P., Cerrani, I., Meroni, M., Caudullo, G., Lugato, E., Vogt, J.V., Spinoni, J., Cammalleri, C., Bastrup-Birk, A., San Miguel, J., San Román, S., Kristensen, P., Christiansen, T., Zal, N., de Roo, A., Cardoso, A.C., Pistocchi, A., Del Barrio Alvarellos, I., Tsiamis, K., Gervasini, E., Deriu, I., La Notte, A., Abad Viñas, R., Vizzarri, M., Camia, A., Robert, N., Kakoulaki, G., Garcia Bendito, E., Panagos, P., Ballabio, C., Scarpa, S., Montanarella, L., Orgiazzi, A., Fernandez Ugalde, O., Santos-Martín, F., 2020a. Mapping and Assessment of Ecosystems and their Services: An EU ecosystem assessment, EUR 30161 EN, Publications Office of the European Union, Ispra, ISBN 978-92-76-17833-0, JRC120383, https://doi.org/10.2760/757183.

Maes, J., Teller, A., Erhard, M., Condé, S., Vallecillo, S., Barredo, J.I., Paracchini, M.L., Abdul Malak, D., Trombetti, M., Vigiak, O., Zulian, G., Addamo, A.M., Grizzetti, B., Somma, F., Hagyo, A., Vogt, P., Polce, C., Jones, A., Marin, A.I., Ivits, E., Mauri, A., Rega, C., Czúcz, B., Ceccherini, G., Pisoni, E., Ceglar, A., De Palma, P., Cerrani, I., Meroni, M., Caudullo, G., Lugato, E., Vogt, J.V., Spinoni, J., Cammalleri, C., Bastrup-Birk, A., San Miguel, J., San Román, S., Kristensen, P., Christiansen, T., Zal, N., de Roo, A., Cardoso, A.C., Pistocchi, A., Del Barrio Alvarellos, I., Tsiamis, K., Gervasini, E., Deriu, I., La Notte, A., Abad Viñas, R., Vizzarri, M., Camia, A., Robert, N., Kakoulaki, G., Garcia Bendito, E., Panagos, P., Ballabio, C., Scarpa, S., Montanarella, L., Orgiazzi, A., Fernandez Ugalde, O., Santos-Martín, F., 2020b. Mapping and Assessment of Ecosystems and their Services: An EU ecosystem assessment - supplement, EUR 30161 EN, Publications Office of the European Union, Luxembourg. ISBN 978-92-76-22954-4, JRC120383. 2, https://doi.org/10.2760/519233.

Maes, J., Teller, A., Erhard, M., Grizzetti, B., Barredo, J.I., Paracchini, M.L., Condé, S., Somma, F., Orgiazzi, A., Jones, A., Zulian, G., Petersen, J.-E., Marquardt, D., Kovacevic, V., Abdul Malak, D., Marin, A., Czúcz, B., Mauri, A., Loffer, P., Bastrup-Birk, A., Biala, K., Christiansen, T., Werner, B., 2018. Mapping and Assessment of Ecosystems and their Services: An analytical framework for ecosystem condition. Luxembourg. https://doi.org/10.2779/12398.

Maes, J., 2010. Soil erosion control [WWW Document]. Eur. Comm. Jt. Res. Cent. URL http://data.europa.eu/89h/4ccdbbf0-fc7c-4fd7-bd8b-f11a06f5df0b (accessed 6.3.19).

Metzger, M.J., Bunce, R.G.H., Jongman, R.H.G., Mücher, C.A., Watkins, J.W., 2005. A climatic stratification of the environment of Europe. Glob. Ecol. Biogeogr. 14, 549–563. https://doi.org/10.1111/j.1466-822X.2005.00190.x.

Müller, F., Burkhard, B., Kroll, F., 2010. Resilience, integrity and ecosystem dynamics: Bridging ecosystem theory and management, in: Landform-Structure, Process Control. pp. 221–242. https://doi.org/10.1007/978-3-540-75761-0_14.

Nedkov, Stoyan, Zhiyanski, Miglena, Dimitrov, Stelian, Borisova, Bilyana, Popov, Anton, Ihtimanski, Ivo, Yaneva, Rositsa, Nikolov, Petar, Bratanova-Doncheva, Svetla, 2017. Mapping and assessment of urban ecosystem condition and services using integrated

index of spatial structure. One Ecosyst. 2, e14499. https://doi.org/10.3897/oneeco.2.e14499.
Panagos, P., Ballabio, C., Borrelli, P., Meusburger, K., Klik, A., Rousseva, S., Tadić, M.P., Michaelides, S., Hrabalíková, M., Olsen, P., Aalto, J., Lakatos, M., Rymszewicz, A., Dumitrescu, A., Beguería, S., Alewell, C., 2015a. Rainfall erosivity in Europe. Sci. Total Environ. 511, 801–814. https://doi.org/10.1016/j.scitotenv.2015.01.008.
Panagos, P., Borrelli, P., Meusburger, K., 2015b. A new European slope length and steepness factor (ls-factor) for modeling soil erosion by water. Geosciences 5, 117–126. https://doi.org/10.3390/geosciences5020117.
Panagos, P., Borrelli, P., Meusburger, K., Alewell, C., Lugato, E., Montanarella, L., 2015c. Estimating the soil erosion cover-management factor at the European scale. Land Use Policy 48, 38–50. https://doi.org/10.1016/j.landusepol.2015.05.021.
Panagos, P., Borrelli, P., Meusburger, K., van der Zanden, E.H., Poesen, J., Alewell, C., 2015d. Modelling the effect of support practices (P-factor) on the reduction of soil erosion by water at European scale. Environ. Sci. Policy 51, 23–34. https://doi.org/10.1016/j.envsci.2015.03.012.
Panagos, P., Borrelli, P., Poesen, J., Ballabio, C., Lugato, E., Meusburger, K., Montanarella, L., Alewell, C., 2015e. The new assessment of soil loss by water erosion in Europe. Environ. Sci. Policy 54, 438–447. https://doi.org/10.1016/j.envsci.2015.08.012.
Panagos, P., Borrelli, P., Robinson, D., 2015f. Tacking soil loss across Europe, 195 Nature 526, 195. https://doi.org/10.1038/526195a.
Panagos, P., Ballabio, C., Meusburger, K., Spinoni, J., Alewell, C., Borrelli, P., 2017. Towards estimates of future rainfall erosivity in Europe based on REDES and WorldClim datasets. J. Hydrol. 548, 251–262. https://doi.org/10.1016/j.jhydrol.2017.03.006.
Panagos, Panos, Meusburger, Katrin, Van Liedekerke, Marc, Alewell, Christine, Hiederer, Roland, Montanarella, Luca, 2014b. Assessing soil erosion in Europe based on data collected through a European network. Soil Sci. Plant Nutr. 60 (1), 15–29. https://doi.org/10.1080/00380768.2013.835701.
Panagos, Panos, Imeson, Anton, Meusburger, Katrin, Borrelli, Pasquale, Poesen, Jean, Alewell, Christine, 2016. Soil conservation in Europe: wish or reality? L. Degrad. Dev. 27 (6), 1547–1551. https://doi.org/10.1002/ldr.2538.
Panagos, Panos, Ballabio, Cristiano, Poesen, Jean, Lugato, Emanuele, Scarpa, Simone, Montanarella, Luca, Borrelli, Pasquale, 2020. A soil erosion indicator for supporting agricultural, environmental and climate policies in the European Union. Remote Sens. 12 (9), 1365. https://doi.org/10.3390/rs12091365.
Panagos, P., Meusburger, K., Alewell, C., Montanarella, L., 2012. Soil erodibility estimation using LUCAS point survey data of Europe. Environ. Model. Softw. 30, 143–145. https://doi.org/10.1016/j.envsoft.2011.11.002.
Panagos, P., Meusburger, K., Ballabio, C., Borrelli, P., Alewell, C., 2014a. Soil erodibility in Europe: a high-resolution dataset based on LUCAS. Sci. Total Environ. 479–480, 189–200. https://doi.org/10.1016/j.scitotenv.2014.02.010.
Panagos, Panos, Standardi, Gabriele, Borrelli, Pasquale, Lugato, Emanuele, Montanarella, Luca, Bosello, Francesco, 2018. Cost of agricultural productivity loss due to soil erosion in the European Union: From direct cost evaluation approaches to the use of macroeconomic models. L. Degrad. Dev. 29 (3), 471–484. https://doi.org/10.1002/ldr.v29.310.1002/ldr.2879.
Paul, Carsten, Kuhn, Kristin, Steinhoff-Knopp, Bastian, Weißhuhn, Peter, Helming, Katharina, 2021. Towards a standardization of soil-related ecosystem service assessments. Eur. J. Soil Sci. 72 (4), 1543–1558. https://doi.org/10.1111/ejss.v72.410.1111/ejss.13022.
Pérez-Soba, M., Elbersen, B., Kempen, M., Braat, L., Staristky, I., Kaphengst, T., Andersen, E., Germer, L., Smith, L., Rega, C., Paracchini, M.L., 2015. Agricultural biomass as provisioning ecosystem service: quantification of energy flows. Publications Office of the European Union, Luxembourg.
Potschin-Young, M., Haines-Young, R., Heink, U., Jax, K., 2016. OpenNESS Glossary [WWW Document]. URL http://www.openness-project.eu/glossary (accessed 2.17.21).
Power, Alison G., 2010. Ecosystem services and agriculture: Tradeoffs and synergies. Philos. Trans. R. Soc. B Biol. Sci. 365 (1554), 2959–2971. https://doi.org/10.1098/rstb.2010.0143.
Recanati, F., Maughan, C., Pedrotti, M., Dembska, K., Antonelli, M., 2019. Assessing the role of CAP for more sustainable and healthier food systems in Europe: a literature review. Sci. Total Environ. 653, 908–919. https://doi.org/10.1016/j.scitotenv.2018.10.377.
Rega, C., Bartual, A.M., Bocci, G., Sutter, L., Albrecht, M., Moonen, A.C., Jeanneret, P., van der Werf, W., Pfister, S.C., Holland, J.M., Paracchini, M.L., 2018. A pan-European model of landscape potential to support natural pest control services. Ecol. Indic. 90, 653–664. https://doi.org/10.1016/j.ecolind.2018.03.075.
Rendon, Paula, Erhard, Markus, Maes, Joachim, Burkhard, Benjamin, 2019. Analysis of trends in mapping and assessment of ecosystem condition in Europe. Ecosyst. People 15 (1), 156–172. https://doi.org/10.1080/26395916.2019.1609581.
Rendon, P., Steinhoff-Knopp, B., Saggau, P., Burkhard, B., 2020. Assessment of the relationships between agroecosystem condition and soil erosion regulating ecosystem service in Northern Germany. PLoS One 15, 1–28. https://doi.org/10.1101/2020.05.26.116285.
Rinot, O., Levy, G.J., Steinberger, Y., Svoray, T., Eshel, G., 2019. Soil health assessment: a critical review of current methodologies and a proposed new approach. Sci. Total Environ. 648, 1484–1491. https://doi.org/10.1016/j.scitotenv.2018.08.259.
Roche, P.K., Campagne, C.S., 2017. From ecosystem integrity to ecosystem condition: a continuity of concepts supporting different aspects of ecosystem sustainability. Curr. Opin. Environ. Sustain. 29, 63–68. https://doi.org/10.1016/j.cosust.2017.12.009.
RStudio Team, 2015. RStudio: Integrated Development for R. RStudio, Inc.
Solte, J., Tesfai, M., Øygarden, L., Kværnø, S., Kizer, J., Verheijen, F., Panagos, P., Ballabio, C., Hessel, R., 2016. Soil threats in Europe: deliverable 2.1 RECARE project. Eur. Commission DG Joint Res. Centre. https://doi.org/10.2788/488054.
Steinhoff-Knopp, B., Burkhard, B., 2018. Mapping control of erosion rates: comparing model and monitoring data for croplands in northern Germany. One Ecosyst. 3, e26382 https://doi.org/10.3897/oneeco.3.e26382.
Temme, A.J.A.M., Verburg, P.H., 2011. Mapping and modelling of changes in agricultural intensity in Europe. Agric. Ecosyst. Environ. 140 (1-2), 46–56. https://doi.org/10.1016/j.agee.2010.11.010.
Turpin, N., ten Berge, H., Grignani, C., Guzmán, G., Vanderlinden, K., Steinmann, H.H., Siebielec, G., Spiegel, H., Perret, E., Ruysschaert, G., Laguna, A., Giráldez, J.V., Werner, M., Raschke, I., Zavattaro, L., Costamagna, C., Schlatter, N., Berthold, H., Sandén, T., Baumgarten, A., 2017. An assessment of policies affecting Sustainable Soil Management in Europe and selected member states. Land use policy 66, 241–249. https://doi.org/10.1016/j.landusepol.2017.04.001.
United Nations, 2020. System of Environmental Economic Accounting (SEEA) [WWW Document]. URL https://seea.un.org/ecosystem-accounting (accessed 10.10.20).
Van Oost, K., Govers, G., De Alba, S., Quine, T.A., 2006. Tillage erosion: a review of controlling factors and implications for soil quality. Prog. Phys. Geogr. 30 (4), 443–466. https://doi.org/10.1191/0309133306pp487ra.

5

Synthesis

5. Synthesis

This chapter summarizes the main findings and discussions described in more detail before. It aims to outline the linkages between the different chapters and provide an overall conclusion based on the most relevant results. Finally, this section ends with some practical recommendations and an outlook on future research goals arisen during this study.

5.1. Main results and discussion

The previous chapters intended to improve the knowledge on the relationships between ecosystem condition and the capacity of ecosystems to supply ecosystem services in Europe. The framework proposed by Burkhard et al. (2018) used in this study provides a step-wise methodology for the integrated mapping and assessment of ecosystems and their services. One of the steps of this framework involves indicators selection for ecosystem condition and ecosystem services, which include those proposed by Maes et al. (2018) and additional indicators found in the literature also valuable for the assessment. This study starts with the analysis of the trends in mapping and assessment of ecosystem condition in Europe. Later, based on those findings, the following chapters focused on the relationships between ecosystem condition and ecosystem services and examined these links at different spatial scales, looking at one ecosystem type and one ecosystem service.

5.1.1. Trends in ecosystem condition mapping and assessment

Chapter 2 starts with a review of the literature on mapping and assessment of ecosystem condition in Europe. Key criteria were used to select and analyse the relevant literature. These criteria also helped to identify the main features of the studies and to make comparisons between them. The focus was on different variables ranging from the type of assessment, the methodologies used, the type of ecosystem and the measured indicators. Subsequently, the chapter analyses the main characteristics and trends of ecosystem condition research in Europe.

The results show an exponential increase in the ecosystem condition research in Europe in the last two decades. Similarly, the concept of ecosystem services is mentioned and used increasingly, especially since the release of the Millennium Ecosystem Assessment in 2005. There are, however, significant differences in the use of terms to refer to ecosystem condition over the years. Descriptors such as *health*, *quality* and *integrity* were used in the early 2000s. On the other hand, s*tatus*, *functioning*, and *state* were more often used towards the late 2010s. These differences in the use of descriptors are driven by the implementation of various environmental directives in Europe that use such concepts to describe ecosystem condition.

The distribution of the studies was unequal in Europe and in the type of ecosystems they assessed. Spain and Italy were the leading countries with the highest number of studies, whereas countries in Eastern Europe had no studies. These differences may arise from the financial and technical resources assigned for research in the countries. Another reason for this unbalanced distribution may be the relevance the countries give to their ecosystems. Regarding ecosystem types, rivers and lakes and marine inlets and transitional waters had the highest number of assessments conducted mainly at regional scales. Forests and coastal ecosystems had the second-highest number of studies primarily performed at local scales. These results possibly relate to the characteristics of the assessments in rivers that are carried out at a catchment level or in a specific site or ecosystem, such as a lake or a forest.

More than half of the literature included in the review assessed ecosystem condition through biological, physical and chemical analyses using direct measurements and monitoring data. A smaller percentage of studies used models to predict, for instance, the level of toxicity in ecosystems and organisms. These results were expected as many of the reviewed articles aimed to assess the level of compliance with environmental directives, and these directives require the measurement and report of such indicators. In contrast, the reviewed studies did not apply methods such as scenarios, workshops or interviews. Possibly because these methods are used more often to assess ecosystem services rather than ecosystems condition.

A high percentage of the reviewed studies used indicators reported under the environmental directives. These indicators inform about water and soil quality and the conservation status of species and habitats. Furthermore, the studies used different ecosystem attributes, including biotic, abiotic and socioeconomic variables as indicators of ecosystem condition. The results highlight that some of the indicators are insufficient to support management decisions and implement specific practices because of the lack of reference values or ancillary background information. Besides, most of the studies were monodisciplinary and did not use holistic methods that provide results about different aspects of the ecosystem condition. More holistic approaches would contribute to a better understanding of the socio-ecological systems.

One of the objectives of chapter 2 was to investigate to what extent ecosystem services were part of ecosystem condition assessments. The results reveal that a low percentage of the studies assessed ecosystem services and relate them to ecosystem condition. The studies evaluated regulating services, including, among others, pollination, habitat protection and maintenance. Provisioning services such as wild plants and animals, cultivated plants, reared animals, and freshwater, were broadly covered. To a lesser extent, the studies assessed cultural services such as interactions with the natural environment. Even though there were some studies linking ecosystem condition and ecosystem services, the percentage is low. The small share of assessments can be explained by the fact that research in these fields is complex and is still in an initial phase.

Another research gap identified in the review is the small number of studies mapping ecosystem condition. These maps covered woodlands, forests and grasslands, principally, and only a few covered marine and coastal ecosystems. The low number of maps in the review can be related to the insufficiency or inadequacy of spatially explicit data on the indicators used to assess ecosystem condition. A further reason for this is that mapping was out of the scope of many publications either because of the authors' disciplines or because the environmental directives do not require a spatial representation of the indicators.

5.1.2. Relationships between ecosystem condition and ecosystem services

Chapters 3 and **4** focus on the relationships between the condition of ecosystems and their capacity to provide services at different spatial scales, considering the knowledge gaps identified in chapter 2. Both chapters investigate the condition of agroecosystems and the regulating ecosystem service *control of erosion rates.* The two chapters follow the integrated operational framework proposed by Burkhard et al. (2018) and some of the indicators proposed by Maes et al. (2018) to map and assess the condition of ecosystems and their services. Chapter 3 presents a regional assessment in the Federal State of Lower Saxony in Northern Germany, whereas chapter 4 centres on the European Union and the United Kingdom.

Regional scale

Chapter 3 tests the feasibility of the indicators proposed in the 5th MAES report (Maes et al., 2018) to assess ecosystem condition at a regional level. The study focuses on agroecosystems in the State of Lower Saxony and the regulating ecosystem service *control of erosion rates*. It starts by identifying the policy objective of maintaining healthy soils. This objective is linked directly to this service and other services such as food and fibre provision, water purification and carbon sequestration. As a next step, some criteria were used to select indicators for pressures, ecosystem condition and services. These criteria are *relevancy* for the ecosystem service under study, *data availability*, *reliability*, and the possibility to *quantify* them to allow for comparisons between municipalities.

A further step was the mapping of the different pressure, ecosystem condition and ecosystem service indicators. For this, the MMU were the municipalities. Subsequently, the study integrates the results to identify the indicators' likelihood to predict the ecosystems' capacity to control soil erosion. This was done from two different angles: (1) by assessing the statistical correlations between indicators and (2) by analysing the spatial distributions and relationships based on the compiled maps.

The maps and results portray the pressure, ecosystem condition and ecosystem service distribution in Lower Saxony. The pressure indicator *change in ecosystem extent* did not have a significant variation between the years 2006 and 2012, which was expected due to the short period of the analysis. Other pressure indicators related to climate identified the lowest *temperatures* and the highest *precipitation* rates in the southern part of the state, principally in mountainous areas. In contrast, these areas recorded the shortest *growing season period*, which is in line with the prevailing low temperatures. When looking at the *soil erosion by water*, the mountainous region was the most affected, which was anticipated due to the characteristic topography and soil cover in hilly areas.

There was an uneven distribution of the condition indicators across the whole state. *Crop diversity* was higher in the north-eastern and central parts. The *share of arable land* was higher in the south-west and east, while the *density of semi-natural vegetation* was higher in the north-west. These differences align with theoretical expectations that indicate that highly cultivated areas tend to have lower shares of semi-natural vegetation. There was a relatively low percentage of *fallow land* and *permanent crops* in the study area. These small shares are probably linked to the agricultural production in Lower Saxony based on horticulture, cereals and intensive dairy farming that lead to a high value per hectare of land.

Livestock density was higher in the western part of the territory, where there is a strongly weighted industry around animal husbandry and the processing of their products. However, this indicator does not entirely reflect the agroecosystem condition because it includes indoor and outdoor systems, which have different impacts on the soil. On the other hand, the condition indicator *soil organic carbon* was relatively high in the study area, especially in the large peat areas in the northwest. As expected, *soil erodibility* was high in the mountainous region in the southwest. Also, in the northwest where soils have high silt content making them highly erodible. The condition indicator *bulk density* was relatively low in the study area, which is characteristic of sandy and sandy loam soils.

The south-eastern part of the study area had a high provision of the ecosystem service *control of erosion rates*, where the *erosion risk* was higher. However, the *actual soil erosion* was considerably lower than the calculated *soil loss potential*, resulting in a high ecosystem service provision. This was also evident in the high values of the indicator *provision capacity* in Lower Saxony, which indicates that most parts of the study area are protected against soil erosion by water. Here, it is worth noting that the study did not include wind and tillage erosion, and so the rates of soil loss might be higher than the values calculated in chapter 3.

The analysis of the relationships between the indicators for pressures, ecosystem condition and the ecosystem service control of erosion rates, mentioned before, showed positive, negative and no correlations. The positive correlations were evident for climate indicators such as *drought index*, *very high* and *extremely high precipitation*, and the *beginning of vegetation period*. *Soil erosion* and *loss of organic matter* also correlated positively with the *control of erosion rates* and *soil erodibility, bulk density, and soil organic carbon*. The indicators used to calculate the ecosystem services also showed a positive correlation between them. In contrast, the indicators *temperature, density of semi-natural areas* and *livestock density* correlated negatively with the ecosystems service.

Most of these correlations align with the framework used in the assessment that identifies *control of erosion rates* as a service that mitigates a structural impact (Steinhoff-Knopp and Burkhard, 2018). This means that in this study, the ecosystem service occurs in areas susceptible to soil erosion where *precipitations* and *loss of SOC* are high, *vegetation periods* are shorter, *temperatures* are low, and *livestock density* is high. Besides, in areas with a high *density of semi-natural vegetation*, the *erosion rates* are low and hence their control due to the vegetation cover and roots retaining the soil.

There were no evident correlations between *control of erosion rates* and indicators such as the *change in agroecosystem extent* and the climate indicators *mean annual precipitation* and *summer soil moisture*. Other indicators with no correlations were *crop diversity*, the *share of fallow land* and *arable land*. The *change in ecosystem extent* was, understandably, close to zero due to the short period of the assessment. Additionally, this indicator does not inform about the type of change, making it difficult to determine its influence on the service provision. On the other hand, the *mean precipitation* indicator does not show the occurrence and frequency of heavy rainfall events, which are determinant for soil erosion. Also, the number of crops displayed by the indicator *crop diversity* does not provide information about the type of crops. Thus, their vulnerability to soil erosion is unknown. The *share of fallow land* was relatively low in the study area, and its role in the service provision needs further research as fallow cycles and management greatly influence the soil condition.

The spatial distribution of the normalized values shows that a high percentage of the study area has medium levels of pressures, medium to high levels of environmental condition and medium to high provision capacity. The north-western, central and eastern parts of the study area had the highest condition levels and provision capacity but the lowest pressures. However, as mentioned before, it is worth noting that the assessment did not account for wind erosion, which underestimates the effects of this type of erosion, especially relevant in the north-west. When the soil erosion risk superimposed with the environmental condition and provision capacity, only a small area in the southern part seems to be "problematic" because of the high erosion risk and the low condition levels and provision capacity. Noteworthy, the study did not estimate other types of erosion, such as tillage erosion or soil erosion by harvesting, which are particularly relevant in agroecosystems. Therefore, the actual erosion risk is higher than the one calculated in the assessment.

European scale

Chapter 4 follows a similar approach as chapter 3. It quantifies and maps indicators for pressures, ecosystem condition and the ecosystem service *control of erosion rates* in agroecosystems, but this time at the European scale. The objective is to test the indicators proposed by Maes et al. (2018) and to assess the relationships between pressures, condition and ecosystem service provision. The chapter starts by applying the operational framework for integrated mapping and assessment of ecosystems and their services from Burkhard et al. (2018). As in chapter 3, the policy objective selected was *maintaining healthy soils* in the EU, framed within the Soil Thematic Strategy, and other legislation aiming to improve the condition, diversity, and resilience of the agroecosystems in the EU. The same criteria were used in both chapter 3 and 4 to select the indicators.

The following step was the mapping and assessment of the different pressure, ecosystem condition and ecosystem service, using 1 km^2 as MMU. Afterwards, the study integrates the results to identify the links between the indicators. This integration was done by statistically analysing the correlation between the indicators per environmental zones. These zones, developed by Metzger et al. (2005), were integrated into the assessment to further subdivide the agricultural areas due to the significant variations of climatic, natural conditions and management practices in Europe and, consequently, in agriculture.

The maps and results represent the distribution of pressure, ecosystem condition and *control of erosion rates* in the European agroecosystems. The pressure indicator *change in ecosystem extent* did not show a significant trend between the years 2012 and 2018, possibly because six years is a short period to identify variations in the size of the ecosystems. Other pressure indicators related to *mean daily precipitation* and *days of heavy* and *very heavy precipitation* were relatively low, with some exceptions in the north Atlantic coasts characterized by high daily precipitations and mountainous regions with heavy rainfall events. As expected, higher *temperatures* were evident in the south of the continent and the lowest in the north and mountainous regions. Correlated with *temperature*, the longer *growing seasons* occur in the south and west of the continent. On the other hand, average *summer soil moisture* was high in the Pannonian and central Atlantic zones and low in the Continental and North Atlantic zones.

Other pressure indicators include *land-use intensity*, *soil erosion* and *loss of organic matter*. In the study, the land-use intensity was calculated based on the energy inputs to the agroecosystems. These inputs cover the entire production process, from seed development to the harvest. In Europe, the zones with the highest land-use intensity were the Mediterranean mountainous and the north and central Atlantic. However, there are considerable differences in these zones regarding the sources of inputs. Irrigation adds significantly to some Mediterranean counties such as Italy and Spain, whereas most of the inputs in central Europe come from energy for cultivation and fertilizers (Pérez-Soba et al., 2015). *Soil erosion* and *loss of organic matter*, on the other hand, are closely related and both show higher rates in the Mediterranean zones in the south and decrease northwards.

Management indicators related to *tillage* and *soil cover* in winter showed marked differences throughout Europe. Due to the high variability of cultivation practices, climate, and geography. *Conventional tillage* is most commonly applied in agroecosystems, followed by *conservation* and *zero tillage* in a low percentage. The zones with the highest shares of conservation and zero tillage are Lusitanian, Atlantic central and Boreal. These high shares are especially evident in Germany and Finland, where conservation practices have spread quickly in the last years, driven principally by economic factors (Kertész and Madarász, 2014). The use of *winter crops* is the primary soil cover practice during winter around Europe. However, a large part of the agricultural soil is left *bare* during winter, principally in the north of Sweden and the Pannonian zone. The use of *plant residues* is the least common practice to protect the soil during winter. Only certain areas in Portugal and Northern Italy have high shares of crop residues.

The results show the variable environmental conditions throughout the European agroecosystems. *Crop diversity* was influenced partly by temperature, with the cool, temperate regions having the lowest number of crop types and the warmer ones having the highest. However, other technological, social, and infrastructural factors not included in the study can influence the diversity of crops and various pressure and condition indicators. Similar to chapter 3, the *share of fallow land* is relatively low. However, this percentage will probably change in the future because of the growing interest in expanding recreational areas, natural reserves, and housing. In some regions, the increase of fallow land could be due to the low incomes from farming and young people moving away from rural areas. On the other hand, although the *share of semi-natural vegetation* and *density of hedgerows* show some discrepancies because of the different data formats and methods used to calculate them, both indicators contrast the percentage of *arable land* and *permanent crops*.

Livestock density based on cattle and ovine has an uneven distribution throughout the study area with a relatively low average value for the EU. However, almost one-third of animal farms concentrate in a small number of areas, especially in Ireland, The Netherlands, Germany, Denmark, and western France. One of the limitations of the indicator of livestock density used in the study is that it does not specify the type of management, providing limited information about the condition of the agroecosystems. The impacts of the systems vary depending on the land use and extension. On the one hand, grass-based systems require more land than indoor systems. On the other hand, the use of permanent grasslands has positive impacts, whereas high animal densities negatively impact soil functions and structure. These differences make it difficult to conclude the overall positive or negative effect of livestock on soils.

Average *soil organic carbon* estimates were relatively low in the whole study area, with higher values found in the north-west and decreasing southwards. Here it is worth highlighting that it was necessary to make some assumptions and generalizations to calculate this indicator. These were somewhat inaccurate but necessary due to the difficulties to reconstruct land uses and management in such a broad territory (Lugato et al., 2016). Additionally, anthropogenic actions such as irrigation, fertilization, tillage, diversification of crops, and land use have a high influence on SOC values. Therefore, future modelling exercises and scenario development should include the effects of such actions. Besides, these models should also include climate change because it has considerable impacts on the agroecosystem's functioning and the carbon balance. These impacts include effects on pollination, the proliferation of parasites, and the change of weed communities, among others.

Soil organic carbon has a considerable impact on *soil erodibility* patterns as environmental zones with high concentrations of organic matter have low soil erodibility and vice versa. Soil erodibility depends on the complex relationships between soil properties such as soil type, texture, and susceptibility to compaction (Ballabio et al., 2016). Additionally, management practices and climate change also impact soil erodibility, although the effects of climate change are more challenging to quantify in short periods. Soil erodibility varies in time and space. Therefore, aspects such as rainfall events, freezing, roughness, vegetation cover and slope should be considered when studying erodibility (Boardman and Poesen, 2007). However, due to the spatial and temporal resolution of the available data in Europe, it is still impossible to account for these variabilities when measuring soil erodibility.

Just like with soil erodibility, *bulk density* and *soil organic carbon* negatively correlate. This correlation is evident in the average values of the environmental zones and the maps of these two indicators. High bulk density is an indicator of soil compaction that has adverse effects on soil quality, such as increase runoff and soil erosion and reduced infiltration and rates of nutrient cycling. These effects also reduce crop yield because they limit root development, and crops cannot obtain nutrients, water and air in sufficient quantities (Logsdon and Karlen, 2004). However, the impacts on crops yield are not easy to measure due to the complex and variable interactions among physical, chemical and biological factors. In addition, management measures and practices can distort the natural balance of the soil if these complex interactions are not considered.

The regulating ecosystem service *control of erosion rates* was measured using two indicators: *soil retention* and *provision capacity*. Soil retention reflects the actual ecosystem service provision by indicating the amount of soil that did not erode. Whereas, the indicator provision capacity shows the fraction of the potential soil erosion that is mitigated. These two indicators are unevenly distributed along the European agroecosystems, with environmental zones having high values of soil retention but low provision capacity, as is the case of the Mediterranean mountainous. In contrast with most of the zones that have low soil retention but high provision capacity. When comparing soil erosion rates and the supply and capacity of the ecosystem to control soil erosion, the results show that the service provision is higher than the actual soil loss. These findings would indicate that the agroecosystems are protected enough against soil erosion. However, it is worth noting that in such large-scale studies, many regional and local features are blurred and do not allow for the identification of "hot spots". In addition, the mean and median values per environmental zone are skewed to the right, meaning that the ranges within them are extensive. These distributions are also evident in the maps that show large areas with low values and small areas with high soil erosion rates and high control. Once again, these results highlight that regional and local features are necessary to identify vulnerable areas and implement the required measures to improve the condition of the ecosystems and enhance the provision of ecosystem services.

There were significant differences in the relationships between indicators of pressures and condition and *control of erosion rates* at the European and environmental zones levels. At the European level, there was hardly any correlation between the indicators. Possibly because of the varied landscapes, climate, soils, cropping patterns and management, which combined create a fuzzy image of the current pressures, ecosystem condition and provision of ecosystem services. Some negative and positive correlations occurred at the level of environmental zones. The negative correlations were more common between pressures and the provision capacity, as happens with temperature, length of the growing season, conventional and zero tillage, and share of winter crops, plant residues and bare soil. Most of these results align with theoretical expectations because areas with low temperatures, for instance, have less vegetation cover and shorter periods of vegetation growth, increasing the risk of soil erosion. However, this was not the case for management indicators such as zero tillage, winter crops and plant residues shares. These discrepancies can be associated with the spatial resolution of the management data. While the indicator provision capacity was represented in a 1 km raster cell, the management indicators were available at NUTS2 level, therefore disaggregating this indicator to a higher resolution implied some inaccurate assumptions. An alternative option would have been upscaling the indicator *control of erosion rates* to the NUTS2 level and make comparisons, but this implicates other imprecise generalizations.

Some negative correlations were also evident between condition indicators and provision capacity. Such indicators include *crop diversity*, *livestock density*, the *share of fallow land*, *arable land*, and *permanent crops*, as well as soil erodibility. In line with theoretical expectations, livestock density is negatively correlated with provision capacity because high animal density impacts the soil structure and functions. Similarly, the share of fallow land was negatively correlated with provision capacity, possibly caused by the lack of vegetation cover. However, it is important to highlight again that a more detailed study would be necessary to assess the influence of fallow cycles and management on soil condition and hence in the provision of ecosystem services. The negative correlation with the share of permanent crops could be associated with the fact that most permanent crops are grown in hilly areas where the susceptibility to soils to erode is higher. In contrast, crop diversity was negatively correlated with provision capacity, probably because a high number of crops per area not necessarily mean that the soil is protected against erosion, as this does not indicate the types of crops that are present, ignoring that some crops are more prone to soil loss than others.

Positive correlations were found with the pressure indicators *precipitation, conservation tillage*, soil covered with *intermediate crops*, and the condition indicators *density of hedgerows* and *soil organic carbon*. The positive correlations with precipitation indicators might be associated with the vegetation-precipitation interactions in which precipitation patterns affect vegetation growth. However, these effects have wide variations among ecosystem types, soils and vegetation species. The indicator *density of hedgerows*, on the other hand, was positively correlated with *control of erosion rates* as hedgerows block the sediments reducing runoff and soil loss. Nevertheless, it is worth highlighting that the study did not assess the type of hedgerow, which is crucial in controlling erosion, as some patterns are more effective than others. The results are also in line with theoretical expectations regarding *soil organic matter* because its high rates increase aggregate stability, which reduces surface runoff.

No significant correlations were evident between *control of erosion rates* and the several pressure and environmental condition indicators. In contrast to theoretical expectations, there were no positive or negative correlations with *soil moisture*, *land-use intensity*, the *share of semi-natural areas* and *bulk density*. These not significant correlations could be associated with different aspects ranging from the data resolution and temporal variability to the scale of the study that difficult the identification of regional or local patterns. Additionally, there was no significant correlation between *soil retention* and *provision capacity*. This outcome, in contrast, is in line with the framework used in the study to assess the *control of erosion rates* proposed by Guerra et al. (2014) and its application to a local case study in Portugal. Their results show a mix between areas with low soil retention and high provision capacity, similar to the results presented in chapters 3 and 4. These mismatches arise because the actual soil retention only happens when the climatic and biophysical conditions are appropriate for the occurrence of the structural impact (potential soil erosion).

In summary, chapters 3 and 4 allow for a better understanding of the relationships between pressures, ecosystem condition and the provision of ecosystem services in agroecosystems. Consequently, these studies help to improve the applicability of the MAES framework and indicators in different ecosystems and spatial scales. However, the methods applied in these two studies must be seen as a starting point. The obtained values and the analyses of correlations should be verified and complemented with additional modelling, monitoring, and measurements. Furthermore, the participation of multiple stakeholders and other disciplines would strengthen future integrated mapping and assessments of ecosystems and their services.

5.2. Summary of answers to research questions

This sub-section provides short answers to the main research questions proposed in chapter 1 used to guide this research. Additionally, these results are compared with recent literature sources.

- *How has ecosystem condition been assessed in different European countries?*

The mapping and assessment of ecosystem condition in Europe have focused on specific environmental directives such as the Water Framework Directive, Marine Strategy Framework Directive, and the Birds and Habitats Directives. Most of the studies identified in chapter 2 were oriented principally towards fulfilling the requirements of such directives, measuring specific pre-established or widely used indicators. Such indicators provide information about the ecosystem characteristics and help to make inferences on their overall condition. The results also show that most of the studies focused on one ecosystem type at a time. Marine inlets and transitional waters, rivers, and lakes are the ecosystems with the highest number of assessments. Ecosystem condition has been evaluated essentially at regional scales, which lines up with the experimental and sampling designs of studies in marine areas, rivers and lakes. There have been little efforts to map ecosystem condition, probably because the frameworks mentioned before do not specifically ask for spatial representation of the indicators. Another reason might be that the disciplines of the researchers involved in the studies do not have a strong focus on GIS.

Recent literature on ecosystem condition has shown multiple indicators used in local, regional and national case studies in Europe. Human footprint, for instance, was used as a proxy to assess the condition of rivers, coastal and marine areas in a Mediterranean biosphere reserve (Barbosa et al., 2019). Other indicators include the degree of conservation of forests in Greece (Kokkoris et al., 2018) according to the Habitats Directive, the share of vegetation cover in urban areas (Nedkov et al., 2017), and the phytosanitary status of urban vegetation (Dimitrov et al., 2018), both in Bulgaria. These recent studies show the increasing mapping and assessment of ecosystem condition through various indicators in ecosystem types not addressed in the environmental directives.

- *What indicators can be used to better define the condition of an ecosystem?*

The assessment of ecosystem condition is highly dependent on the ecosystem type and the region in which it is located. Condition is strongly affected by anthropogenic and natural pressures, and therefore the selection of indicators should take these factors into account. Other aspects to consider when choosing condition indicators are the availability of temporal and spatially explicit data to monitor and analyse their variability, and the relevance for policy. In this context, the indicators proposed by Maes et al. (2018) are a good starting point to guide the assessments. However, the results of chapters 3 and 4 evidence that data are not always available at appropriate temporal and spatial resolutions. Another limitation of some indicators is that they are insufficient to determine ecosystem condition, especially when identifying the relationships with ecosystem services. For agroecosystems, for example, indicators such as crop diversity, livestock density, and the share of fallow land lack some precisions. Additional information is needed to understand the condition of the agroecosystems, for instance, the types of crops and livestock and the duration and management of the fallow cycles.

Ecosystem condition has received some attention in the last years both in national and international initiatives. Frameworks such as the SEEA-EA (System of Environmental Economic Accounting - Ecosystem Accounting) and the MAES have presented extensive discussions about the indicators to assess and account for ecosystem condition. Some outcomes of these frameworks are the criteria to select indicators and recommendations about what type of metrics to use. Czúcz et al. (2020) proposed a typology to classify metrics used to describe ecosystem condition. This typology aims to establish a common language, improve the comparability of different studies, and provide a template to select condition indicators. The authors of this publication highlight that different ecosystem types require specific indicators to describe them, and original data are preferred over aggregated indices. In addition, data about pressures (as presented in the MAES framework (Maes et al., 2018)), natural resources management, stable environmental characteristics and extent should be regarded as ancillary data and not as part of the condition assessment itself.

- *How can the proposed indicators contribute to making policy decisions for managing ecosystems?*

The indicators assessed in chapters 3 and 4 are a good starting point to identify areas in which measures should be implemented to improve or maintain ecosystem condition. However, the identification of priority areas depends on the availability of temporally and spatially explicit data. These indicators could provide some guidance to establish overall policy targets at the EU or country level. Nevertheless, the measures needed to achieve those targets should be applied considering local characteristics. Additionally, highlighting the link between ecosystem condition and the provision of ecosystem services gives additional support to increase conservation efforts and improve sustainable management.

The EU ecosystem assessment report (Maes et al., 2020) emphasises the relevance of indicators to identify improvements or deterioration of ecosystems. The previous consultation with the Member States and EU services was an initial step for selecting the indicators and the framework proposed in the 5th MAES report (Maes et al., 2018). Therefore, they are considered appropriate for policy-making as some of these measure progress to targets under different policy frameworks. However, some additional further steps are needed to improve the mapping and assessment of good ecosystem condition. These steps include selecting a minimum set of key indicators that can be monitored over long periods, defining reference conditions, and proposing aggregation schemes that consider the complex interactions between pressures and ecosystem condition.

- *To what extent does the condition of an ecosystem determine its capacity to supply ES?*

In general, an ecosystem in better condition has a higher capacity to provide ecosystem services (in this study, control soil erosion). However, the complex interactions between pressures, ecosystem condition and services make it impossible to identify linear responses of ecosystems. The results of this study lead to thinking that the condition-service relations depend on the ecosystem service and the indicators used for the assessment. However, further studies that include bundles of services and other indicators would produce different results. Other aspects that would influence the outcomes of such studies are the spatial and temporal variability of ecosystem properties. Studies conducted on a local scale with high resolution or even field data would probably show different correlations than continental studies with data obtained from models. On the other hand, ecosystem properties are also sensitive to changes over time. Therefore, the results will vary depending on the temporal scale of the study.

According to Grizzetti et al. (2019), who assessed the relationships between the ecological status of European water bodies and multiple ecosystem services, ecosystems in good condition provide more ecosystem services. Keith et al. (2020) highlight the different perspectives in condition assessment. As mentioned in chapter 1, one perspective relates to the interdependency of all ecosystem elements, structure and functioning that maintain its integrity. Another perspective is more oriented towards the ecosystem services and their link with the required ecosystem condition to supply them. However, the latter might not include all ecosystem characteristics that interact to provide a full range of services. Additionally, it might overlook the complex, multi-dimensional, and non-linear relationships between condition and services. Therefore, the influence of ecosystem condition on services provision depends on the purpose of the assessment and intrinsic or instrumental values perspectives.

5.3. Challenges and uncertainties

Several sources of uncertainty are associated with this study due to its complexity. This sub-section discusses the main methodological challenges and uncertainties encountered during the assessment of the relationships between ecosystem condition and the provision of ecosystem services.

5.3.1. Bias in the selection of literature

A systematic review helps to identify, select and assess relevant research to a specific field, and to analyse data from the studies included in the review. In this study, a literature review was carried out to understand the trends in mapping and assessing ecosystem condition in Europe. One possible bias occurred when locating the publications at a specific geographic area (Europe), focusing on a specific language (English), and years of publication (2000 to 2017), possibly ignoring other relevant studies. Another source of bias relates to the databases used when searching for the articles. Although the major known databases were used, additional ones would have return further relevant articles. The third source of bias could happen when selecting the studies while screening them and making decisions about the eligibility of a study based on the different combinations of terms. The fourth source of bias relates to the analysis of the studies. This kind of bias may appear while selective reporting on some aspects of the studies, excluding non-significant outcomes. These biases were addressed as much as possible in the study by explaining the objectives, scope and methodology, and by acknowledging the limitations in the discussion section. However, the omission of relevant papers and results might be possible.

5.3.2. Quantification and mapping of indicators

Many sources of uncertainty emerge when quantifying and mapping indicators for ecosystem condition and services. The most noticeable uncertainty source comes from the datasets used to calculate the indicators. For instance, land use/land cover data from CORINE (European Environment Agency, 2019, 2012) have a minimum mapping unit of 25 ha for status layers and 5 ha for change layers, which could hinder the representation of small landscape features. Other data related to climate variables obtained from national (Deutscher Wetterdienst, 2018) and European (Copernicus, 2019) sources have limitations regarding the data collection methods and the changes of measurements over time. These bring some inaccuracies when comparing the girds from different years. Another source of uncertainty arises from the differences in spatial and temporal resolutions of the various datasets. Therefore, it was necessary to rescale and aggregate the data and make some generalizations to compare the indicators. Although these standard practices may introduce additional errors, they were appropriate for the aims of this study.

One source of uncertainty relates to the availability of statistical data of relevant indicators such as crop rotation and soil biodiversity. There have been simulations about crop growth to predict yields in crop rotations in some parts of Europe (Kollas et al., 2015). However, statistical and spatially explicit data on crop rotations are not freely available for the whole study area. Similarly, abundance and richness data of some species, particularly soil organisms are not yet available at a large scale. In addition, although the importance of soil biodiversity on soil condition and the *control of erosion rates* has been increasingly recognized, the models used to calculate soil erosion do not consider the role of soil-living organisms (Orgiazzi et al., 2018). This absence of data can be related to privacy concerns or lack of reporting (for crop rotations). Or insufficient measurement and monitoring of soil fauna.

Another source of uncertainty related to the quantification and mapping of indicators involves the lack of validation of the results and maps obtained in the study. Due to the absence of adequate data at a proper scale, the Universal Soil Loss Equation (USLE) was not validated for Lower Saxony. This model was used to calculate indicators such as soil loss, soil retention, and provision capacity. It has several limitations and uncertainties associated with the coupling of on-site risk erosion to runoff patterns and depositions. Some of the model variables were adapted to the study area (e.g., rainfall erosivity factor in Germany). However, due to the limited availability of measured values and lack of time and financial resources, the validation and sensitivity analysis was complicated.

5.3.3. Interpretation of statistical results

Statistical results help understand the study outcomes and identify aspects such as the variables' effects, correlations between variables, and differences among observations. However, errors can occur when interpreting the results due to the sample size or the overinterpretation of non-significant results (Makin and Orban de Xivry, 2019). In this study, it was necessary to select representative samples to make comparisons between indicators. Nevertheless, when they are small, their distribution is more likely to deviate from normality. This effect was evident in the European study where one Environmental Zone had a few data points to assess, and some extreme outliers were found. The overinterpretation of non-significant results, on the other hand, may occur when the ρ-value does not meet the established threshold, and it is assumed that it is meaningless, while in contrast, it could provide evidence against the hypothesis. For this reason, when a low ρ -value was found, the correlation was described as non-significant.

In this study, many correlations between indicators were evident, but these correlations do not prove a cause-effect relationship. Correlated incidences might indicate direct or reverse causation but could also result from a coincidence or due to an unknown cause. When significant correlations were identified, causal language was avoided to prevent result misinterpretation. Additionally, it is worth noting that complex, multi-dimensional, and non-linear relationships exist between ecosystem services and condition (Keith et al., 2020). Hence, the presumption of causal relationships is incorrect and inappropriate in these types of assessments.

5.4. Conclusions and outlook

This research aimed to identify the relationships between ecosystem condition and the provision of ecosystem services. Based on a literature review and a quantitative analysis of multiple pressures, ecosystem condition and ecosystem services indicators, it can be concluded that there is a growing interest in assessing ecosystem condition. Also, its relevance to the provision of ecosystem services is increasingly recognized. The results show that the legal requirements of environmental directives influence the assessment of the condition of multiple ecosystems in Europe. Besides, additional indicators are used to determine the condition of ecosystems that do not have a status definition under European environmental legislation.

The results of the literature review provide an overview of the existing ecosystem condition studies. They also helped to identify some knowledge gaps that guided this research. Some of these gaps associate with the lack of holistic approaches to assess multiple ecosystem components. Another gap relates to the need to define reference conditions and priority indicators that support management and decision making. Holistic approaches that include the effects of socio-economic factors on ecosystem condition would provide more robust information to managers. These approaches will allow for the implementation of adequate policy measures. However, to implement such measures and guide the definition of policy objectives, it is necessary to select appropriate indicators and reference values and develop guidelines to interpret the results and their implications. Another considerable research gap is the absence of integrated mapping and assessments of ecosystems and their services that could also contribute to design and implement better policies.

In this study, a comprehensive analysis of the relationships between ecosystem condition and services is performed based on an operational framework for ecosystems and services mapping and assessment. The aim is to help improve the applicability of the indicators proposed by the MAES working group for the evaluation of ecosystem condition and to relate those indicators with ecosystem services. A first analysis made at the regional level in Northern Germany, quantify and map the indicators for pressures and agroecosystem condition, and link them with the provision of one chosen exemplary ecosystem service, *control of erosion rates*. The results of this analysis show that not all indicators are suitable to assess condition, or the data to quantify them are insufficient. Additionally, positive, negative, and no correlations between the pressure and condition indicators and service provision were identified.

The regional assessment allowed to point out municipalities in Lower Saxony where agroecosystems have high environmental pressures, limiting conditions, and the *control of erosion rates* is relatively low. However, further spatially and temporally explicit data are needed to identify the specific areas that need interventions. With these higher-resolution data, it would be possible to superimpose raster cells and better recognise the relationships between condition and services in comparison to statistical analysis that is limited to correlations between mean values. The temporally explicit data would allow researchers to analyse the trends over time and recognise seasonal variabilities of condition and ecosystem service provision.

The study at the continental level follows the same operational framework for the integrated mapping and assessment of agroecosystems and *control of erosion rates* as the regional study, but this time applied to the EU and UK. The results show similarities with the regional analysis in the way that although positive, negative and no correlations between indicators were found, no cause-effect relationships could be identified. In addition, due to the extent of the study area and the spatial resolution of the data, many regional and local features are unidentifiable. To improve the identification of regional characteristics, the environmental stratification of Europe was included in the assessment. This stratification clusters areas with similar climatic and geomorphological patterns and allows for the comparison between indicators in smaller spatial extents. It also helps to identify correlations within the different environmental zones.

The European study also makes evident the need to improve the monitoring and reporting of data required to assess the condition of multiple ecosystems. The study calls for the identification of priority areas where conservation or restoration efforts are required by using spatially and temporally explicit data. The results also highlight the need to analyse the complex interactions between condition and services provision at a sub-European scale. This regional analysis would address the various landscapes and cropping patterns, that in this case, affect soil erosion and its control in different ways. For this purpose, a tiered approach is suggested for the implementation of measures to prevent soil erosion. Such an approach consists of proposing policy targets and guidelines on the European and country level, a regionally adapted soil conservation framework and locally implemented practices.

Another conclusion from the study is the need for transdisciplinary research. Transdisciplinary studies allow researchers to assess the condition and services provided by agroecosystems but also human health, consumption patterns and economic aspects. This more comprehensive research would better support the design and implementation of management practices that are advantageous for farmers, consumers and the environment. In addition, this research would also contribute to assessing how ecosystem condition could improve the resilience of ecosystems and the society to drivers and pressures such as climate change, increasing population, and demand for productive land and resources.

Transdisciplinary research also examines social and cultural drivers that determine the implementation of crop and soil management actions. Such an analysis would also give additional guidance in the design and execution of sustainable practices that enhance the condition of agroecosystems and their services. This analysis would take into consideration not only farmers' perceptions but also markets and consumers' behaviour. Transdisciplinary research has also the potential to contribute to the definition of reference conditions for highly managed ecosystems such as agroecosystems. It would incorporate multiple views about the optimal ecosystem condition. These reference conditions would allow for the comparisons between different regions and again identify priority areas that require actions to maintain or improve their condition.

Both the regional and continental studies assessed one exemplary ecosystem service (*control of erosion rates*). However, the analysis of additional ecosystem services would provide information about trade-offs and synergies. This analysis could help identify areas that require protection, restoration or further development. Furthermore, analysing additional ecosystem services would also involve evaluating other condition indicators used to create a comprehensive index that would be easier to communicate to managers, decision-makers and other stakeholders.

A further step to improve the research on agroecosystem condition and services is exploring different land use or land management scenarios. The analysis of various land uses or combinations (e.g., crops, livestock, and trees) would help understand the different interactions between them and the effects on ecosystem condition and services provision. Such an assessment would also identify how the diverse combinations of land uses adapt to regional socio-economic and environmental characteristics. For this assessment, it is necessary to assess first the ecosystem condition and bundles of services of the diverse land uses and combinations at a local scale. Second, identify their technical, social, and economic advantages and limitations. Third, find the appropriate actions applicable in the local context to improve ecosystem condition and ensure the sustainable provision of multiple services.

References

Ballabio, C., Panagos, P., Monatanarella, L., 2016. Mapping topsoil physical properties at European scale using the LUCAS database. Geoderma 261, 110–123. https://doi.org/10.1016/j.geoderma.2015.07.006

Barbosa, A., Martín, B., Hermoso, V., Arévalo-Torres, J., Barbière, J., Martínez-López, J., Domisch, S., Langhans, S.D., Balbi, S., Villa, F., Delacámara, G., Teixeira, H., Nogueira, A.J.A., Lillebø, A.I., Gil-Jiménez, Y., McDonald, H., Iglesias-Campos, A., 2019. Cost-effective restoration and conservation planning in Green and Blue Infrastructure designs. A case study on the Intercontinental Biosphere Reserve of the Mediterranean: Andalusia (Spain) – Morocco. Sci. Total Environ. 652, 1463–1473. https://doi.org/10.1016/j.scitotenv.2018.10.416

Boardman, J., Poesen, J., 2007. Soil erosion processes, in: Boardman, J., Poesen, J. (Eds.), Soil Erosion in Europe. John Wiley and Sons Ltd.

Burkhard, B., Santos-Martín, F., Nedkov, S., Maes, J., 2018. An operational framework for integrated Mapping and Assessment of Ecosystems and their Services (MAES). One Ecosyst. 3, e22831. https://doi.org/10.3897/oneeco.3.e22831

Copernicus, 2019. Copernicus Climate Change Service [WWW Document]. URL https://climate.copernicus.eu/index.php/ (accessed 7.31.19).

Czúcz, B., Keith, H., Maes, J., Driver, A., Jackson, B., Nicholson, E., 2021. A common typology for ecosystem characteristics and ecosystem condition variables. One Ecosyst. 6. https://doi.org/10.3897/oneeco.6.e58218

Deutscher Wetterdienst, 2018. Climate Data Center (CDC OpenData) [WWW Document]. URL https://opendata.dwd.de/climate_environment/CDC/ (accessed 10.8.18).

Dimitrov, S., Georgiev, G., Georgieva, M., Gluschkova, M., Chepisheva, V., Mirchev, P., Zhiyanski, M., 2018. Integrated assessment of urban green infrastructure condition in Karlovo urban area by in-situ observations and remote sensing. One Ecosyst. 3. https://doi.org/10.3897/oneeco.3.e21610

European Environment Agency, 2019. Land Cover Change (LCC) 2012-2018 [WWW Document]. URL https://land.copernicus.eu/pan-european/corine-land-cover/lcc-2006-2012?tab=metadata (accessed 7.31.19).

European Environment Agency, 2012. CLC 2012 [WWW Document]. URL https://land.copernicus.eu/pan-european/corine-land-cover/clc-2012?tab=metadata (accessed 8.23.19).

Grizzetti, B., Liquete, C., Pistocchi, A., Vigiak, O., Zulian, G., Bouraoui, F., De Roo, A., Cardoso, A.C., 2019. Relationship between ecological condition and ecosystem services in European rivers, lakes and coastal waters. Sci. Total Environ. 671, 452–465. https://doi.org/10.1016/j.scitotenv.2019.03.155

Guerra, C., Pinto-Correia, T., Metzger, M.J., 2014. Mapping Soil Erosion Prevention Using an

Ecosystem Service Modeling Framework for Integrated Land Management and Policy. Ecosystems 17, 878–889. https://doi.org/10.1007/s10021-014-9766-4

Keith, H., Czúcz, B., Jackson, B., Driver, A., Nicholson, E., Maes, J., 2020. A conceptual framework and practical structure for implementing ecosystem condition accounts. One Ecosyst. 5, 1–54. https://doi.org/10.3897/oneeco.5.e58216

Kertész, Madarász, B., 2014. Conservation Agriculture in Europe. Int. Soil Water Conserv. Res. 2, 91–96. https://doi.org/10.1016/S2095-6339(15)30016-2

Kokkoris, I.P., Dimopoulos, P., Xystrakis, F., Tsiripidis, I., 2018. National scale ecosystem condition assessment with emphasis on forest types in Greece. One Ecosyst. 3, e25434. https://doi.org/10.3897/oneeco.3.e25434

Kollas, C., Kersebaum, K.C., Nendel, C., Manevski, K., Müller, C., Palosuo, T., Armas-Herrera, C.M., Beaudoin, N., Bindi, M., Charfeddine, M., Conradt, T., Constantin, J., Eitzinger, J., Ewert, F., Ferrise, R., Gaiser, T., Cortazar-Atauri, I.G. de, Giglio, L., Hlavinka, P., Hoffmann, H., Hoffmann, M.P., Launay, M., Manderscheid, R., Mary, B., Mirschel, W., Moriondo, M., Olesen, J.E., Öztürk, I., Pacholski, A., Ripoche-Wachter, D., Roggero, P.P., Roncossek, S., Rötter, R.P., Ruget, F., Sharif, B., Trnka, M., Ventrella, D., Waha, K., Wegehenkel, M., Weigel, H.J., Wu, L., 2015. Crop rotation modelling-A European model intercomparison. Eur. J. Agron. 70, 98–111. https://doi.org/10.1016/j.eja.2015.06.007

Logsdon, S.D., Karlen, D.L., 2004. Bulk density as a soil quality indicator during conversion to no-tillage. Soil Tillage Res. 78, 143–149. https://doi.org/10.1016/j.still.2004.02.003

Lugato, E., Paustian, K., Panagos, P., Jones, A., Borrelli, P., 2016. Quantifying the erosion effect on current carbon budget of European agricultural soils at high spatial resolution. Glob. Chang. Biol. 22, 1976–1984. https://doi.org/10.1111/gcb.13198

Maes, J., Teller, A., Erhard, M., Grizzetti, B., Barredo, J.I., Paracchini, M.L., Condé, S., Somma, F., Orgiazzi, A., Jones, A., Zulian, G., Petersen, J.-E., Marquardt, D., Kovacevic, V., Abdul Malak, D., Marin, A., Czúcz, B., Mauri, A., Loffer, P., Bastrup-Birk, A., Biala, K., Christiansen, T., Werner, B., 2018. Mapping and Assessment of Ecosystems and their Services: An analytical framework for ecosystem condition. Luxembourg. https://doi.org/10.2779/12398

Maes, J., Teller, A., Nessi, S., Bulgheroni, C., Konti, A., Sinkko, T., Tonini, D., Pant, R., 2020. Mapping and Assessment of Ecosystems and their Services: An EU ecosystem assessment, JRC Science for Policy Reports. European Commission. https://doi.org/10.2760/757183

Makin, T., Orban de Xivry, J.-J., 2019. Science Forum: Ten common statistical mistakes to watch out for when writing or reviewing a manuscript. Elife 8, e48175. https://doi.org/10.1016/j.enfi.2020.09.002

Metzger, M.J., Bunce, R.G.H., Jongman, R.H.G., Mücher, C.A., Watkins, J.W., 2005. A climatic stratification of the environment of Europe. Glob. Ecol. Biogeogr. 14, 549–563. https://doi.org/10.1111/j.1466-822X.2005.00190.x

Nedkov, S., Zhiyanski, M., Dimitrov, S., Borisova, B., Popov, A., Ihtimanski, I., Yaneva, R., Nikolov, P., Bratanova-Doncheva, S., 2017. Mapping and assessment of urban ecosystem condition and services using integrated index of spatial structure. One Ecosyst. 2. https://doi.org/10.3897/oneeco.2.e14499

Orgiazzi, A., Ballabio, C., Panagos, P., Jones, A., Fernández-Ugalde, O., 2018. LUCAS Soil, the largest expandable soil dataset for Europe: a review. Eur. J. Soil Sci. 69, 140–153. https://doi.org/10.1111/ejss.12499

Pérez-Soba, M., Elbersen, B., Kempen, M., Braat, L., Staristky, I., Kaphengst, T., Andersen, E., Germer, L., Smith, L., Rega, C., Paracchini, M.L., 2015. Agricultural biomass as provisioning ecosystem service: quantification of energy flows. Publications Office of the European Union, Luxembourg.

Steinhoff-Knopp, B., Burkhard, B., 2018. Mapping Control of Erosion Rates: Comparing Model and Monitoring Data for Croplands in Northern Germany. One Ecosyst. 3, e26382. https://doi.org/10.3897/oneeco.3.e26382

Acknowledgements

I want to express my gratitude to all those who contributed to the completion of this thesis.

First of all, I want to thank my supervisor Benjamin Burkhard for his supervision, patience, trust, motivation, inspiration and great support beyond the academic and work fields. I also like to thank Emily Poppenborg and Dolf de Groot for agreeing to serve as reviewers of my dissertation.

Some parts of this work were possible thanks to my visit to the JRC in 2019. I especially want to thank Joachim Maes for welcoming me into his team for the three months I spent there. I also want to say thank you to Maria Luisa Paracchini for her guidance and support, Carlo Rega, Sara Vallecillo and Balin Czúcz for the knowledge and experience they shared with me. I thank other staff members of the JRC that made my stay there an enjoyable experience.

Thanks to Bastian Steinhoff-Knopp for his contributions and support, especially with GIS and soil erosion. His advice and critical review of the manuscripts. Thanks to Angie Faust for her help when settling up in Hannover and for proofreading my texts.

Thanks to all my colleagues at Phygeo for their support and welcoming atmosphere while I have been here. I really enjoyed our coffee and lunch breaks, our staff outings and other activities we did together in and outside the University. Special thanks to Kremena for making me feel welcomed in Hannover and her home and for her kind words. I will definitely miss you all!

Thanks to Monica, Alejandro, Oscar, Yuly, Lina, Sylvie and Grissel for their friendship, support and the great times we could spend together despite the distance and the circumstances.

Special thanks to my family: To my parents, Consuelo and Alfonso, my brother Jorge, my grandmothers Cecilia and Rubiela, my aunts, uncles, and cousins who always support me, inspire me and encourage me to follow my dreams. To my mother-in-law Stefania, my sister-in-law Alice and nonno Gigi for welcoming me into their family, and for their support during all this time. Most importantly, I am grateful to my husband Ivan for his unconditional love, patience, motivation and support. Thank you for encouraging me to continue the journey despite my doubts and insecurities.

Thank you!

Curriculum vitae

Paula Mercedes Rendon Cardona

Date and place of birth: March 31st, 1988 Bogota, Colombia

Education

Since 2017	**Doctoral studies in geography**
	Leibniz Universität Hannover, Germany
	Topic of dissertation: “Relations between ecosystem condition and ecosystem services at different spatial scales”
	Promoter: Prof. Dr Benjamin Burkhard
	Co-promoter: Prof. Dr Emily Poppenborg Martin
2014 – 2016	**Master studies in environmental sciences**
	Wageningen University, The Netherlands. Degree: Master of Sciences
	Thesis: “The role of natural capital to enhance the resilience to drought in the Coorong lagoons, Lower Lakes and Murray Mouth, South Australia”
2005 – 2010	**Bachelor’s degree in environmental engineering**
	El Bosque University, Bogota, Colombia. Degree: Environmental Engineer
2004	High School, María Auxiliadora School Chia, Colombia

Professional background

Since 2017	**Research assistant**
	Institute of Physical Geography and Landscape Ecology, Leibniz Universität Hannover, Germany
06/2019-08/2019	**Visiting researcher**
	Joint Research Centre of the European Commission, Ispra, Italy

08/2016-12/2016	**Member of the organizing committee**
	First Regional Conference of the Ecosystem Services Partnership in Latin America and the Caribbean (18-21 October 2016)
	FSD (Foundation for Sustainable Development) – ESP (Ecosystem Services Partnership) Wageningen, The Netherlands/ Cali, Colombia
01/2016-06/2016	**Postgraduate student**
	CSIRO (Commonwealth Scientific and Industrial Research Organisation), Adelaide, Australia
09/2015-12/2015	**Intern**
	FSD (Foundation for Sustainable Development) – ESP (Ecosystem Services Partnership) Wageningen, The Netherlands
2011-2014	**HSEQ Coordinator**
	Elite Training Ltda, Bogota, Colombia
2011-2014	**Responsible Care Coordinator**
	UPL Colombia SAS, Bogota, Colombia
2010-2011	**Junior consultant**
	Fundación Instituto Excelencia, Bogota, Colombia
06/2009-11/2009	**Intern**
	The Elite Flower Ltda, Facatativá, Colombia

Paula Mercedes Rendon Cardona

January 2022

Publications

Rendon, P., Erhard, M., Maes, J., & Burkhard, B. (2019). Analysis of trends in mapping and assessment of ecosystem condition in Europe. *Ecosystems and People*, *15*(1), 156-172.

Rendon, P., Steinhoff-Knopp, B., Saggau, P., & Burkhard, B. (2020). Assessment of the relationships between agroecosystem condition and the ecosystem service soil erosion regulation in Northern Germany. *PloS one*, *15*(12), e0234288.

Rendon, P., Steinhoff-Knopp, B., & Burkhard, B. (2022). Linking ecosystem condition and ecosystem services: A methodological approach applied to European agroecosystems. Ecosystem Services, 53, 101387.

Scientific communications

Rendon, P., Erhard, M., Maes, J., & Burkhard, B. Analysis of trends in mapping and assessment of ecosystem condition in Europe. 9th ESP World Conference in Shenzhen, China 11-15 Dec 2017. Poster presentation – Winner of 2nd place best poster award.

Rendon, P., Steinhoff-Knopp, B., & Burkhard, B. Assessment of the relationships between agroecosystem condition and soil erosion regulating ecosystem service in Northern Germany. ESP Regional conference Europe in San Sebastian, Spain 15-19 Oct 2018. Poster presentation.

Rendon, P., Steinhoff-Knopp, B., & Burkhard, B. Integrated mapping and assessment of agroecosystems in Northern Germany. 10th IALE World Congress in Milan, Italy 1-5 July 2019. Oral presentation.

Rendon, P., Steinhoff-Knopp, B., & Burkhard, B. Application of an operational framework for the integrated mapping and assessment of agroecosystems' condition and services at different spatial scales. 10th ESP World Conference in Hannover, Germany 21-25 Oct 2019. Oral presentation.

Rendon, P., Steinhoff-Knopp, B., & Burkhard, B. Linking ecosystem condition and ecosystem services: A MAES framework applied to European agroecosystems. 3rd ESP Europe Conference in Tartu, Estonia 7-10 June 2021. Oral presentation.