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In copertina
Olive grove in Corfù (Greece) (source: Mark Cartwright, in https://www.worldhistory.org/image/5644/olive-grove/*, used under licence CC BY-NC-SA 4.0 / Cropped from original*

Note biografiche

Cristian Soldati is a PhD student at the Department of Agriculture of the Mediterranean University of Reggio Calabria. He obtained a Bachelor's degree in Environmental Sciences at the University of Bologna - Ravenna Campus in 2017 and then a Master's degree in Environmental Analysis and Management at the same university in 2020.

He currently carries out research activities in the field of ecosystem services assessment through a Life Cycle approach, with a focus on terrestrial ecosystems. In the course of his PhD he has participated in several conferences and contributed to scientific publications.

His main interests concern i) ecosystem services and biodiversity, ii) environmental, economic and social assessments using Life Cycle Thinking approaches, iii) the inclusion of organic carbon in life cycle assessment processes and iv) the analysis of ecosystem services in support of current agricultural policies.

The research area of his PhD thesis focuses on the environmental and economic valuation of ecosystem services from a Life Cycle perspective. The study integrates tools such as the Geographic Information System for spatial analysis and applies various methodologies to assess both environmental and economic outcomes, considering several methodologies like the cost of restoration and willingness to pay.

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Integrating Ecosystem Services and Life Cycle Methodologies in Mediterranean olive growing

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Ai miei nonni e in particolare a mia nonna che
non ha potuto vedere la fine di questo percorso

SIC PARVIS MAGNA
DALLE PICCOLE COSE NASCONO LE GRANDI

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MOTIVATION AND STRUCTURE OF THE THESIS

Twenty years have passed since the first classification proposed by Millennium Ecosystem Assessment. During this period several works were produced in the attempt to measure Ecosystem Services driving also by policy maker with decisions more and more focused on the environmental health. On one side, the field of Ecosystem Services modelling has advanced, but on the other side, the world is experiencing the first dangerous consequences of neglecting these important functions, as observed in the previous century. Some examples include extreme climatic events such as the Emilia-Romagna floods in 2023 and 2024, cyclones and hurricanes in different parts of the world (mainly in the U.S.A.), as well as biological issues like the spillover that led to the global pandemic.

Although models to quantify ESs exist, some gaps persist, such as the neglect of certain parameters related to anthropogenic activities. One of the most important human activities, in particular in the Mediterranean basin, is the olive growing. Olive growing is a complex and multi-functional system that provides several ESs, from olives to climate regulation, from water filtration to aesthetic service. In this particular context, the SustainOlive is inserted, aiming to enhance the sustainability along the entire supply chain and in particular in the olive groves. This project evaluated, in qualitative terms, the ESs produced by certain management practices, such as the use of cover crops to reduce soil erosion and the integration of livestock to increase organic matter. However, a quantitative appraisal is missing to determine the magnitude of these benefits.

In this work an evaluation to understand the magnitude of ESs is performed considering the olive farms of SustainOlive project. The structure of this thesis is reported below.

Chapter 1 provides the introduction, discussing the definitions and classifications of ESs, as well as the structure of Life Cycle Assessment (LCA) and Life Cycle Costing (LCC) and the core concepts regarding how ESs are considered in Life Cycle (LC) perspective.

An overview of how ESs are integrated in LC methodologies is provided in Chapter 2, focusing on general aspects such as the year of publication or journal, methodological aspects like the functional unit applied or the system boundaries used, and ESs-related aspects such as the services considered, and the classification applied.

The Chapter 3 shows an empirical classification of methods and models to assess ESs in LC methods valuing for example the data required, or the output produced.

Materials and Methods are detailed in Chapter 4, with a comprehensive explanation of SustainOlive and the methods used. The results are presented and discussed in Chapter 5 and Chapter 6. The conclusion, along with identified gaps and future perspectives, is provided in Chapter 7.

Research Problem and Objectives

Agricultural systems provide multiple ESs (Swinton et al. 2007), yet conventional LC methodologies have limitations in fully incorporating these services into sustainability assessments (Blanco et al. 2018).

Existing models focus primarily on provisioning and regulating services, often neglecting the influence of land-use dynamics, biodiversity shifts, and anthropogenic interactions (D'Amato et al. 2020). This lack of a comprehensive assessment framework limits the ability to make informed policy and management decisions regarding agricultural sustainability (Moreno-Miranda and Dries 2022).

Current LC models predominantly assess provisioning and regulating services but often fail to incorporate broader ecological dynamics, such as biodiversity loss and long-term land-use changes (Chaudhary et al. 2015; Rugani et al. 2019). Addressing these limitations requires an approach that not only refines the conceptual integration of ES in LC methodologies but also enhances its methodological robustness for sustainability assessments. This research builds upon existing methodologies while addressing key methodological gaps by proposing a structured approach that integrates multi-method assessment techniques. The methodological inconsistencies in current frameworks highlight the need for a structured approach that ensures reproducibility and applicability across diverse agricultural contexts. This research seeks to address these limitations by proposing an enhanced framework for ESs integration.

By addressing these research questions, this thesis aims to contribute to the advancement of LC methodologies, ensuring a more comprehensive assessment of ESs in sustainability studies. This work is expected to provide relevant insights for both academic research and policymaking, particularly in the context of sustainable agricultural systems and European environmental policies. While previous studies have attempted to include ESs within LCA frameworks (e.g., impact categories, inventory modelling, etc.; see Chapter 2 for further details), inconsistencies remain in the definition, quantification, and valuation of ES. This research builds upon existing frameworks but aims to bridge key methodological gaps by proposing a structured approach that integrates multi-method assessment techniques. By addressing these limitations, the study contributes to the refinement of LC methodologies, ensuring a more holistic assessment of ESs in agricultural systems (Hein et al. 2020; Taelman et al. 2023). Given this general goal, this study is guided by the following research questions:

1. What are the strengths and weaknesses of current LC methodologies in assessing ecosystem services?

2. How can multiple methods be applied to improve the integration of ecosystem services into LC perspective?
3. To what extent can the selected methods be applied to other agricultural systems beyond olive cultivation?
4. How can the results of this research contribute to European policy frameworks related to ecosystem services and sustainable agriculture?

The methodological approach adopted in this thesis aligns with recent advancements in impact assessment methodologies that emphasise integrating ESs in sustainability evaluation frameworks (Teixeira et al. 2016). By systematically applying these methods to Mediterranean agricultural systems, this research aims to provide a replicable framework applicable to other agricultural contexts. These research questions define the analytical scope of this study and guide the selection of methodologies for evaluating ESs within LCA frameworks. Building upon these research questions, the study formulates specific objectives aimed at addressing these methodological challenges and advancing the integration of ESs into LCA:

- Objective 1: Identify methodological inconsistencies in existing frameworks for the integration of ESs in LC methods for agricultural systems and assess their applicability to Mediterranean olive cultivation.
- Objective 2: Integrate multiple existing methodologies and tools (e.g., Environmental Priority Strategies in product design (EPS 2015), Land Use Indicator Value Calculation in Life Cycle Assessment (LANCA), i-Tree Canopy) to improve the assessment of ecosystem services within LCA for olive farming systems.
- Objective 3: Evaluate the scalability of the proposed methodology to other cropping systems, particularly perennial and mixed agroecosystems.
- Objective 4: Investigate how anthropogenic activities (e.g., irrigation, soil management, biodiversity conservation) influence the provision of ES and refine LC methodologies accordingly.

By addressing these challenges, this research contributes to a more robust and holistic approach to sustainability assessment. The findings are expected to support both methodological advancements and policy-making strategies, particularly in the context of European environmental policies.

This research is based on the hypothesis that a multi-method integration of ecosystem services into LC perspective can provide a more comprehensive sustainability assessment for agricultural systems. By addressing current methodological gaps, the proposed framework is expected to improve the applicability and robustness of LC methods in evaluating ESs.

The research questions and objectives outlined in this chapter form the foundation for the subsequent sections of the thesis. Chapter 2 provides a state-of-the-art review of ES and LC methods integration, while Chapter 3 builds on this foundation to assess the current methodological frameworks. The following chapters progressively refine these approaches, culminating in a novel methodology tested within the SustainOlive project.

Abbreviations

EPS 2015	Environmental Priority Strategy in product design 2015
ESs	Ecosystem Services
LANCA	Land Use Indicator Value Calculation in Life Cycle Assessment
LC	Life Cycle
LCA	Life Cycle Assessment
LCC	Life Cycle Costing

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Chapter 1.
INTRODUCTION

1.1 **Introduction**

1.1.1 *Ecosystem services: classification and importance in policy-making*

The expression “Ecosystem Services” was first mentioned in the text “Extinction: The Causes and Consequences of the Disappearance of Species” by Ehrlich and Ehrlich (1981) and it is usually referred to in studies that deal with the subject. The concept of Ecosystem Services (ESs) emerged in the 1960s and 1970s, with key works such as “Silent Spring” (Rachel 1962) and “Limits to Growth” (Meadows et al. 1972) influencing the valuation of human-nature interactions. Two study threads can be recognised as having laid the ground for later studies on ESs: one within the natural sciences, namely ecology, and the other in economic sciences (Braat and de Groot 2012). According to Mooney and Ehrlich (1997), George Perkins Marsh's book entitled Man and Nature, which was published in 1864, is an important and enlightening work that laid the foundations of a new scientific approach to the human-nature relationship. As early as a century later, new studies came to break the traditional perspective in ecology and questioned the study of natural ecosystems as isolated entities in which the human role was excluded. The authors highlight that society must rely on the balance of ecosystems and their healthy maintenance, largely threatened by human activities (Ehrlich 1968) and in general links between nature and social structures (Odum 1970). The term “ecosystem services” in a more contemporary definition can be traced back to the report to MIT by Wilson, C. L. and Matthews, W. H (1970) entitled “Study of Critical Environmental Problems”. The work made mention of how human activities affected global climate and ecological balances and listed a range of “environmental services” that would be lost if ecosystem functions were severely compromised (Mooney and Ehrlich 1997). At this point, the term 'function' transitions from solely denoting physiological processes related to ecosystem balance to a more explicitly 'utilitarian' definition, which focuses on the impact of these processes on sustaining living conditions for humans (Hueting et al. 1998; Jax 2016). Fisher et al. (2009) emphasise the importance of terminology in the context of ecosystem studies discussing the subtle difference between “ecosystem function” and “ecosystem functioning,” which leads into a debate about anthropocentrism and implicit goals.

In the 1970s, various studies enhanced this field; following “environmental services,” other definitions emerged: “public service functions of the global environment” (Holdren and Ehrlich 1974), “service functions of the global ecosystem” (Ehrlich et al. 1977), “nature's services” and “service of an ecosystem” (Westman 1977), “functions of the natural environment” (Braat et al. 1979) and lastly “ecosystem services” in the text of Ehrlich and Ehrlich (1981) where they were identified as vital since they are unable to be replaced. The work of Westman (1977) is particularly interesting because it is among the first to clearly and critically raise this dependence of society upon ecosystems from an ecological point of view, underlining the necessity of their economic evaluation. Giving a quantitative value to nature is an issue in itself controversial; the author gives examples of economic evaluation from a previous research project distinguishing between good and services provided by nature. The former is essential for the physical survival of the community in the form of food, minerals, soil, and water; the latter are ecosystem dynamics that provide all kinds of benefits to mankind. These ideas connect to the second branch of studies that laid the foundation for research on ESs, specifically connecting to economic science. In 1970, United States scholars established the Association of Environmental Economics (Spash 1999). Its scholars aimed to include environmental implications as integral parts of economic assessments. The approach that opposed the neoclassical economics strategy, which revolved around trade value, did not include natural resources as determining factors because they were perceived to be free and arguably unlimited. Environmental Economics brings another approach to economic debates by adding together the influence of human's economic activities on the environment, their contribution to maintaining the standard of life of a society, and the economic consequences for the environment policy. In the 1980s, a split within the discipline resulted in the establishment of Ecological Economics. This branch offered a more critical perspective on neoclassical methods and a broader perspective on the relationship between humans and the biosphere (Gómez-Baggethun et al. 2010). Environmental Economics and Ecological Economics differ in their views on natural capital. Environmental Economics sees natural capital as partially replaceable by technological capital and human effort, while Ecological Economics views the two as complementary but considers natural capital to be essentially irreplaceable. Both fields concentrate on interpreting the value of natural elements, which is crucial from an economic perspective (Farber et al. 2002). Therefore, a key area of study in these disciplines is the development of methods to measure the value of these services, as they are not traditional market commodities.

The most commonly used method in this context is known as “willingness to pay” (WTP) or “willingness to accept” (WTA). Willingness to pay is a concept that assigns economic value to goods or services based on the amount of money individuals are willing to spend to benefit from them, in the absence of market prices. This trait is crucial to highlight, as it has significantly influenced the establishment of the valuation of ESs at a scientific level. This method of quantifying economic value highlights the human-centred and practical aspect of commodities or services associated with nature: only those that provide a benefit to the individuals being studied are considered valuable in calculating by WTP. Willingness to pay is often determined by empirical data collection using quantitative or qualitative methods. Willingness to accept (WTA) is a related concept, representing the compensation consumers would accept for the loss of a good or service. Several inquiry procedures are well-established and documented in numerous texts and guides. The natural sciences and economic disciplines were the first fields to explore the economic value of environmental components in the 1970s. The concept of ESs gained popularity and was defined in the 1980s and 1990s and then was solidified with the publication of the Global Biodiversity Assessment in 1995 (Watson 1995), which was commissioned by UNEP (United Nations Environment Programme). This paper aligns with the Convention on Biological Diversity (UN 1992), which was signed by 196 countries committing to preserve natural resources and safeguard biodiversity. The Global Biodiversity Assessment is the initial report that showcases the current scientific understanding of the connections between humans and nature. It includes two chapters focused on ecosystem functioning processes and ESs, as well as one chapter on the economic worth of biodiversity. Currently, there is no consensus among scientists on a clear definition of ESs (Boyd and Banzhaf 2007). Various researchers provide different definitions based on the specific focus of their research. The definitional issue is crucial since it goes beyond a mere debate about terminology and instead focuses on how this concept is comprehended and acknowledged in practice. The use of a common definition can facilitate meaningful international comparisons and enhance the quality of scientific analyses (Fisher et al. 2009). Two prominent definitions in the scientific field are those provided by Daily (1997) and Costanza et al. (1997). The definition proposed by Daily (1997, pag. 3) is as follows:

"Ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life. [...] In addition to the production of goods, ecosystem services are the actual life-support functions, such as cleansing, recycling and renewal, and they confer many intangible aesthetic and cultural benefits as well".

In "The value of the world's ecosystem services and natural capital" by Costanza et al. (1997, pagg. 253-254), one of the pilar work in the literature on ESs, titled, it states:

"Ecosystem functions refer variously to the habitat, biological or system properties or processes of ecosystems. Ecosystem goods (such as food) and services (such as waste assimilation) represent the benefits human populations derive, directly or indirectly, from ecosystem functions. […] Ecosystem services consist of flows of materials, energy, and information from natural capital stocks which combine with manufactured and human capital services to produce human welfare".

The two definitions differ significantly. The first focuses on the conditions and processes of ecosystems that support human life and provide benefits. The second defines ESs as the benefits themselves, distinguishing between goods and services and mentioning the flows of materials and energies that connect natural capital to human-produced resources. Costanza et al.'s article is significant as it provides the first experimental monetary valuation of ESs on a global scale. The authors calculated an average total value of approximately US$ 33 trillion per year, which is about three times the global GDP at the time. The valuation is derived from an analysis of over a hundred previous publications, supplemented with particular calculations, predictions, and modifications detailed in the article. The study by Costanza et al. does not aim to establish a precise monetary value for natural components. Instead, it seeks to demonstrate and bring attention to a value that is often overlooked or not well-defined, even within the scientific community.

Costanza et al.'s approach falls within the initial phase of spreading the concept of ESs, identified by Gómez-Baggethun et al. (2010) as more "pedagogical," focused on raising awareness of the contribution of ESs to society rather than strictly evaluating their monetary value.

The Millennium Ecosystem Assessment (MA 2005) has significantly influenced current research by becoming the first scientific study to analyse the worldwide status of ecosystems. The definition of MA, "the ecosystem services are the benefits people obtain from ecosystems," partly derived from Costanza et al. (1997), and is surely the most concise and therefore easier to remember and widely used even in non-scientific contexts. Ecosystem services, being constituted of "benefits," fall squarely within the ambit of a strongly anthropocentric approach that fully meets the objectives of the MA. Its ultimate objective is to evaluate the implications of recent modifications in ecosystem balances and their repercussions on human health and well-being, and from this evaluation, to define the scientific basis necessary to foster ecosystem conservation policies. The MA developed the first and most well-known classification system for services, which are categorised into four groups:

- Provisioning services, which supply material goods like food, water, fuel, and raw materials of all kinds.
- regulating services that control other services which are imperative for the well-being of humans, such as climate regulation, air and water quality regulation, soil formation among others.
- Cultural services include non-material benefits, such as cultural identity, spiritual and intellectual enrichment, and aesthetic and recreational values.
- Supporting services, which include habitat creation, genetic biodiversity conservation, and making sure development of the above three categories end.

The MA analysis found that approximately 60% of the world's ecosystems were in poor conservation status or irreversibly degraded. This degradation was found to be due to 5 direct drivers of climate change, pollution, overexploitation, habitat transformation, and invasion of alien species. Even indirect drivers have helped decrease ESs and biodiversity. These include the population change-growth and migration-economic fluctuations, including economic growth and maldistribution of wealth and trade patterns-sociopolitical factors, including conflict, public involvement to decision-making, cultural impacts, and technological change.

The categorisation by the MA has encountered severe criticism. One of the main criticisms against this classification is that it does not distinguish between the ways or means used to produce goods or services and the goods and services produced. For example, water supply is a method through which drinking water is obtained; therefore, water supply and access to drinking water are two different services that should not be mixed up. Boyd and Banzhaf (2007) explicitly refute the connection between benefits and ESs, which they consider to be the natural components that are directly consumed or used to produce well-being. Ecological processes are not, by definition, services or benefits. Fisher et al. (2009) recognise three kinds of components: “intermediate” services, direct services, and benefits (or goods, which generally require human or technological input to be appropriated). Clean water is a natural nutrient cycling product, yet also a contributor to the supply of drinking water for human consumption. Drinking water is a beneficial product for humans where human intervention is needed in order for it to be accessible. The production of clean water is a direct service, whereas the nutrient cycling is an ecological process independent of human influence.

In 2007, a study group called TEEB or The Economics of Ecosystems and Biodiversity was formed with the support of UNEP and the EU in order to focus on systems and biodiversity in terms of economics. The primary outcome of this research came out in the year 2010 and is really a first step forward by the model created by MA (TEEB 2010). According to TEEB, ESs were defined as all the direct and indirect benefits that ecosystems provide for human wellbeing. The difference in this case from MEA is that here the ecosystems are not considered as benefits by themselves but are acknowledged for their contribution to human wellbeing. TEEB uses a cascade model as suggested originally by Haines-Young and Potschin (2010), which has been developed further for the research of TEEB by De Groot et al. (2010). The cascade model is a schema showing how ESs are transferred from the natural environment to the human sphere. This model explicitly defines the flow of services and distinguishes them into five subcategories. The first category refers to “biophysical processes and structures,” by which is meant processes not serving human needs directly, such as natural habitats. The second involves the functions or capacities of those processes, for example, water flow or biomass production. The third category involves more anthropocentric services, for example, food production or flood protection. The fourth sub-category involves very benefits which humans get from the ecosystem functions in the form of health, food security, and general well-being. The last category consisted of the economic values, both monetary and non-monetary, which people attribute to the services and the techniques developed for their valuation, for instance, through willingness to pay or accept. The sub-categories form parts of larger categories, namely: the first two under "Ecosystems and Biodiversity" under the green rectangle while the last two are under "Human wellbeing socio-cultural context" under the pink rectangle. This category "services" in the centre is an interface between two sectors in blue (Figure 1.1). This scheme thus shows that the category of ESs is indeed an interpretative paradigm, linking the ecosystem processes that are not driven by human beings but are affected by them, and benefits and values determined through human judgment or interpretation.

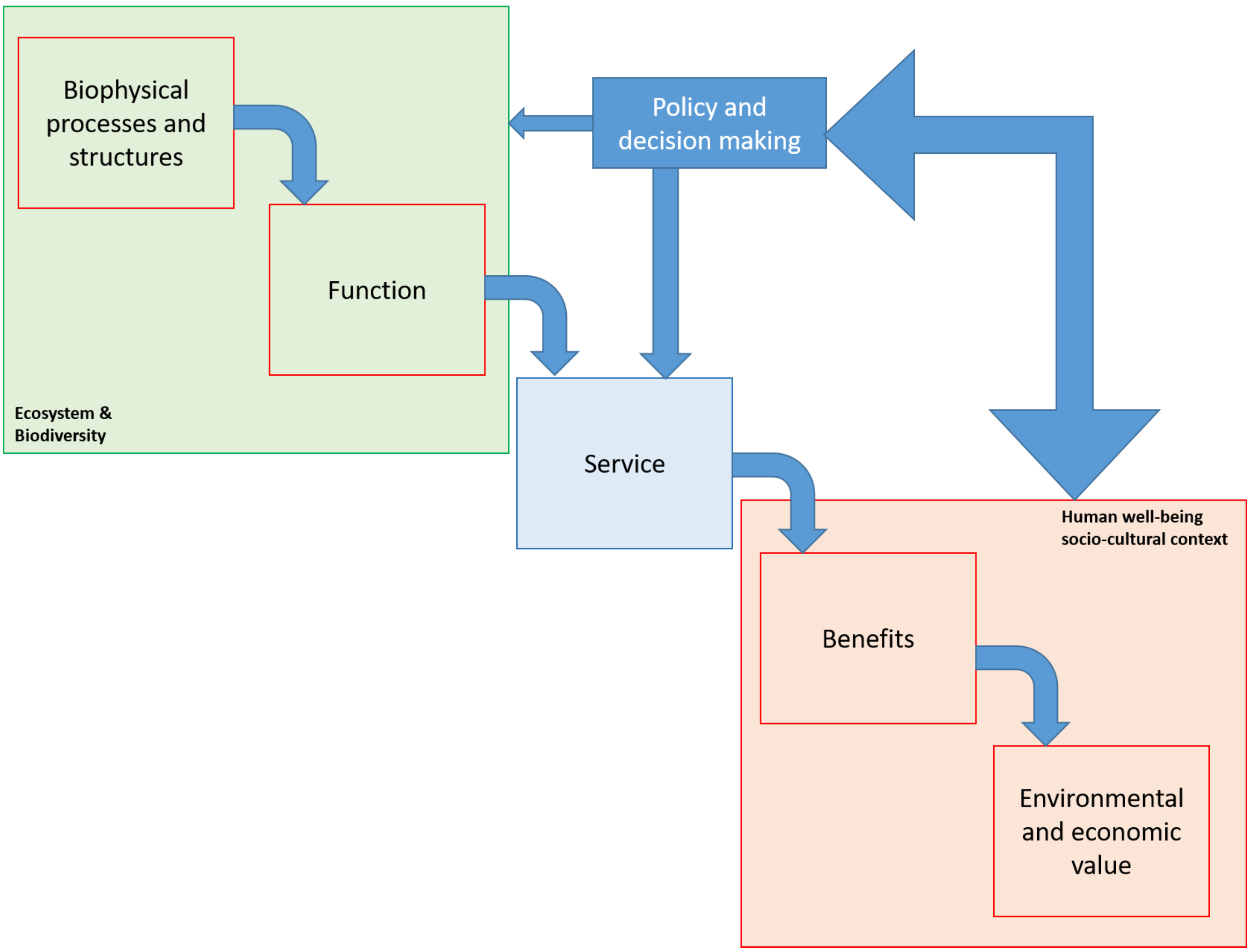

Figure 1.1 Cascade model (source: personal elaboration based on TEEB (2010)).

The cascade model has been a significant development in expanding and clarifying the concept of ESs. While there is ongoing discussion about the definition of ESs, many authors now incorporate, modify, or integrate the cascade model to suit various research studies (Potschin and Haines-Young 2016). An evolution of the MA categorisation was proposed Haines-Young and Potschin (2011). The European Environmental Agency (EEA) introduced the Common International Classification of Ecosystem Services (CICES) to establish standardised criteria for assessing ecosystems uniformly across Europe. The CICES definition of ESs is as follows (Haines-Young and Potschin 2011):

"For the purposes of CICES, ecosystem services are defined as the contributions that ecosystems make to human well-being. They are seen as arising from the interaction of biotic and abiotic processes and refer specifically to the 'final' outputs or products from ecological systems".

This categorisation has been borrowed from MA model and TEEB, with a certain modification and improvement directed mainly towards distinguishing between services and ecosystem functions for clear identification of the end services. It categorised ESs into three categories: Provisioning, Regulating & Maintenance, and Cultural. The authors exclude the category Supporting because they find it irrelevant to the assessment of end service and group it under the ecosystem "functions" instead of ecosystem "services".

The categorisation under each one of these fields is hierarchical: ESs are categorised into Division, Group, Class. The hierarchy is static and fairly rigid yet intended to sharply distinguish ESs; it is to be generic foundation for persons doing economic valuations. Although the effort of standardisation by CICES is constructive, some scholars put forward the argument that it has mainly utilised the terminology and methods of natural science, which provides little applicability to social studies and decision-making in the field of ESs. According to Schwilch et al. (2016), CICES is a regularly updated document, which has been done with several evaluations of ESs at the national levels in various European nations and the actual version is V4.3 (No 2012).

There are similar characteristics that can be identified in the various definitions provided in the literature:

i) Ecosystem services are a concept that highlights the human-centred view that emphasises the importance of natural processes for human well-being.

ii) these services are an interpretative tool that demonstrates how human societies rely on natural processes, which would exist regardless of human presence but are significantly impacted by it in modern times.

iii) the ecosystem services paradigm aims to quantify the value of natural processes in supporting human living conditions, not just in economic terms, with the goal of developing practical tools to safeguard ecological processes.

The features serve as the theoretical foundation for comprehending the idea of ESs in this research, interpreting the acquired data, and forming its findings.

Reflection on ESs has expanded and intensified in the past decade, focusing on a more accurate assessment of their ecological and economic value, as well as their application in political decision-making. Two main areas of focus are the assessment methods (monetary and non-monetary) and the geographical mapping of these approaches in relation to population demand. The economic assessment of ESs is crucial in defining the concept of these services. According to this perspective, economic judgement is not optional; the decision is in whether to articulate the values and evaluations that influence of the societal choices. In this perspective, society and individuals consistently assign a value to environmental processes. When these values are not clearly articulated, it leads to a lack of recognition and distorts decision-making processes, ultimately impacting environmental services that benefit mankind. Clearly defining the economic value and the boundaries of scientific understanding regarding environmental processes and ESs is crucial for developing policies that fully consider the value of these processes and services beyond just their economic aspects. Evaluation is seen as an essential instrument in ensuring the protection of the environment and its ecosystems, as well as maintaining good living conditions for humans. Commonly used and approved evaluation methods can be categorised into monetary and non-monetary approaches. Monetary methods quantify the value of ESs in monetary terms, while non-monetary methods focus on aspects like quantity, quality, and values beyond monetary considerations (such as spiritual or collective identity values). Monetary approaches are prevalent because their outcomes are readily applicable in decision-making as they communicate effectively to a wide audience. Non-monetary incentives are less common yet offer a wider range of tactics and outcomes. Many scholars have critiqued or pointed out potential dangers regarding the topic of evaluation. Cowell and Lennon (2014) have raised doubts about the belief that increased scientific expertise invariably results in improved political choices. They view the non-linearity of the knowledge-action relationship and the involvement of several subjects and conditions as crucial theoretical assumptions for comprehending the issue. Additional scholars have pointed out an increasing focus on monetary economic assessments in literature and practices related to ESs, cautioning against the risk of nature being treated as a commodity (Norgaard 2010). They warn that the concept of ESs could become a methodological pitfall, potentially masking the diverse characteristics of ecosystems. Gómez-Baggethun and Ruiz-Pérez (2011) conducted a detailed investigation of the distinction between evaluation and commodification. To explain this difference is necessary to differentiate between use value and trade value. The use value is the aspect of a good that allows for the differentiation of its individual and societal dimensions, whereas the exchange value is what categorises the good as a commodity. Since an economic valuation of ESs does not determine their market value, interpreting nature only in economic terms is not sufficient to turn it into a commodity. However, while the distinction may make sense theoretically, in reality, the line between assessing a product and its prospective transferability is often unclear, particularly in today's neoliberal environment.

Ecosystem services' monetary valuation methods aim to convert the value individuals place on these services into a monetary amount, facilitating their consideration in decision-making processes, where natural services are often perceived as free. These strategies are mostly founded on the concept of willingness to pay or willingness to accept. It is an economic analysis tool primarily used to estimate non-market values; it represents the subjective value consumers attribute to a good or service, indicating the price they would pay for a change in well-being. Willingness to accept is a related concept, representing the compensation consumers would accept for the loss of a good or service. Willingness to pay is often determined by empirical data collection using quantitative or qualitative methods. Several inquiry procedures are well-established and documented in numerous texts and guides. Farber et al. (2002) provide a clear picture and describe six of them: Avoided Cost, Replacement Cost, Factor Income, Travel Cost, Hedonic Pricing, and Contingent Valuation are several methods used in economic analysis. It is vital to note that each service being analysed may require one or more evaluation procedures, and no single technique can provide a comprehensive assessment on its own. The literature discussing application case studies of ESs evaluation also mentions an alternative methodology known as the benefit transfer method or value transfer method. This approach relies on utilising secondary data to provide extensive assessments when constraints related to resources or time prevent the collection of empirical data (Wilson and Hoehn 2006; Richardson et al. 2015). This method is commonly employed in assessing ESs because of the challenges and time involved in locating primary data. It has even been the focus of a dedicated manual published by UNEP (Brander 2013).

The key focus in monetary valuation approaches, particularly in urban and territorial planning, lies in the theoretical assumptions rather than the tools used. This evaluation is founded on the concept of assigning value at an individual level, as emphasised by Farber et al. (2002) and Kenter (2016). The total value, considered to be “social,” is actually the combination of the values that individuals assign to a specific commodity or service. The neoclassical idea of value pertains to rational, predefined, individual, and utilitarian values. It is thought that individuals can communicate their logical values by recognising how items or services contribute to their well-being during data collecting to determine willingness to pay. Yet, this rational process does not accommodate the communication of values associated with a social or really collective aspect. This is particularly noticeable in the context of cultural ESs, such as historical significance or community identity traits associated with certain locations. It also extends to regulatory services, where individuals may not always recognise the urban value of a wooded area in reducing CO_2 emissions or the flood regulation function of a river area, among other service types. Kenter (2016) further points out that conventional economic research emphasises the importance of what is valued, and the value people place on it, but often neglects to explore the reasons behind why specific values are assigned to certain goods or services. Monetary methods are inadequate for capturing the whole spectrum of social values inherent in human perception of ecosystem products and services. Another theme related to this is the importance of recognising a diverse dimension of the concept of value, as emphasised by Gómez-Baggethun et al. (2016) using the term value pluralism since ESs encompass a wide range of values including economic, social, and ecological aspects, attracting various stakeholders who assign value to them for different reasons. The authors believe that acknowledging and accepting the diverse perspectives within a range of values is essential for achieving a comprehensive and realistic evaluation and they suggest that once the concept of value pluralism is generally embraced in the ESs community, non-monetary valuing can be replaced by more positive terms.

Before concluding this brief introduction to ESs, it might be relevant to underscore how these have been integrated more and more in recent years into major European and international policies. The European Taxonomy is, for example, a classification system elaborated for classifying those economic activities that are environmentally friendly (European Commission 2020a). For example, the European Taxonomy Regulation identifies the criteria and conditions an activity must satisfy to be labelled “green”. It precludes, for instance, any activity that will not contribute to one of the six stated purposes, such as “protection and restoration of biodiversity and ecosystems”.

In this respect, the EU Biodiversity Strategy for 2030 is an example of translation into legislation, programs, and policies of the work of ESs in relation to the policy-making process (European Commission 2020b). It should be a broad, ambitious, long-term strategy for nature conservation and balancing ecosystem degradation. Therefore, it shall allow the restoration of biodiversity in Europe by 2030 simply with the proper implementation of concrete actions and commitments.

Nowadays, ESs have gained international policies; the Kunming-Montreal agreement, better known as the Global Biodiversity Framework (UN 1992) is one of the major documents adopted at the Conference of the Parties for Biodiversity in 2022. The agreement was adopted by 190 countries aiming to establish 4 goals with 23 targets to be achieved by 2030 in order to halt and reverse the decline of biodiversity and the ecological services underpinning it.

Over time, the scientific understanding of ESs has evolved, shifting from a purely conceptual framework to a structured approach integrated into sustainability assessments. This transition has been particularly significant in environmental policy and decision-making, emphasising the role of ESs in maintaining ecological balance and human well-being.

1.1.2 *Life Cycle Assessment – LCA*

Guinée et al. (2011) published a comprehensive summary of the history of the Life Cycle Assessment (LCA) approach.

In the late 1960s and early 1970s, two significant environmental challenges arose, prompting the development of LCA. The 1973 oil crisis and subsequent energy debate, together with concerns about resource conservation, were the primary factors that increased the popularity of LCA as a methodology (Meadows et al. 1972).

In 1969, the first recorded LCA was conducted in the United States for Coca-Cola by the Midwest Research Institute using a methodology that was later called Resources and Environmental Profile Analysis (REPA) (Hunt 1974; Hunt and Franklin 1996). The organisation was examining several packaging-related issues, such as different types of beverage containers such as plastic bottles, reusable glass bottles and disposable containers, along with the environmental consequences of producing the packaging. The study's most surprising finding was the company's switch from glass to plastic bottles, a bold decision given the unfavourable perception of plastic at the time. Other independent projects conducted during this period include Ian Boustead's LCA investigation of milk packaging in the UK (Boustead 1996), a research project initiated by the Federal Ministry of Education and Science in Germany to examine the impact of plastic in packaging (Oberbacher et al. 1996), and LCA research in Sweden triggered by Tetra Pak's proposal to launch a PVC bottle, which was carefully examined for its recyclability and its substantial contribution of acidifying substances when burned.

Until the early 1980s, LCA started as an approach initially confined to internal corporate decision-making, which up to that time has now been extended to public discourses. Large environmental catastrophes, such as the Bhopal chemical accident in 1984, the Chernobyl nuclear reactor explosion in 1986, and the Exxon Valdez oil spill in 1989, raised public interest in environmental issues. Initially, environmental impacts were measured in terms of energy and material consumption and amounts of waste, with a limited focus on emissions. In the 1970s and 1980s, LCAs were executed with different approaches and without a consistent theoretical framework. Businesses have also frequently abused LCA forjustifying marketing claims. Even with the same study objects, results showed significant deviations, which obstructed the wider acceptance and usage of LCA as an analytical tool (Guinée et al. 1993).

The increasing number of LCA-studies has generated skepticism and uncertainty about the method. Many studies have been accused of being biased and used by product manufacturers whose investment in research was used to advance their products (London SustainAbility Ltd. 1993). In that time LCA was an attractive method since it was product-related, measurable and systematic which gave an impression of neutrality. The major setback was that it lacked a standard, hence it was subjective to certain interpretations on the ground during its implementation. The major effort placed in the 1990s was merely for improving and further developing this LCA technique. This activity culminated in scientific and coordinating activities mounted globally in seminars and forums like those held by the World Wildlife Fund and The Conservation Foundation in Washington DC in 1990 and the Society of Environmental Toxicology and Chemistry (SETAC) – Europe in Brussels in 1992. Other LCA guides and manuals were also published, for example "Environmental Assessment of Products". Volume 1 "Methodology, Tools and Case Studies in Product Development" (Wenzel et al. 1997) and "Volume 2 Environmental Assessment of Products. Scientific Background" (Hauschild and Wenzel 1998). The early scientific publications were originally published in scientific journals like the Journal of Cleaner Production, Resources, Conservation and Recycling, Environmental Science & Technology and the Journal of Industrial Ecology. It has produced a number of its own publications, the most important of which is the International Journal of Life Cycle Assessment.

During this time, the first scientific conferences on LCA were organised by SETAC. The early development of the LCA methodology through definition and framework, with initial guidelines within a Code of Practice (Consoli and SETAC 1993) was under the auspices of the critical role of SETAC. The development of the method led to the standardisation of LCA through International Organisation for Standardization (ISO) between 1993 and 2002 with the publishing of the 14040 series. The actual standards are:

- UNI EN ISO 14040:2021
- UNI EN ISO 14044:2021

The work invested in developing the methodology has also been acknowledged in the academic arena with the awarding of LCA.

With this, ISO never wanted to detail standardisation of LCA methodologies, which resulted in larger subjectivity and freedom of interpretation in other areas of methodologies. This Joint Research Center via the European Commission and the Directorate-General for Environment worked in conjunction with the development of the International reference of Life Cycle Data system handbooks - ILCD. These manuals give support for an objective application of the LCA methodology to be able to compare results with reliability and consistency.

The ISO standards and ILCD Handbook give the framework of LCA, comprising of four phases that are executed consecutively, as presented below (Baldo et al. 2008) and in Figure 1.2.

1. Goal and scope definition. This phase defines the objectives of the study, its target audience, and the functional assumptions for the subsequent analysis stages. It involves the definition of functional unit, system boundaries, methods, cut-off criteria, and data quality requirements.

2. Life Cycle Inventory: It involves quantification of input and output flows of different processes within the system.

3. Life Cycle Impact Assessment: Input and output streams entering the system are changed into factors that can affect the environment. The step in the process is segregated into categorisation, characterisation, normalisation, and weighting. These last two stages in the process-that is to say, normalisation and weighting-are taken as optional within the LCA methodology.

4. Interpretation: One evaluates at this stage whether or not the aims of the study have been reached if the assumptions used in it are congruent. Recommendations are made about improvements to be carried out in environmental performance of the studied system.

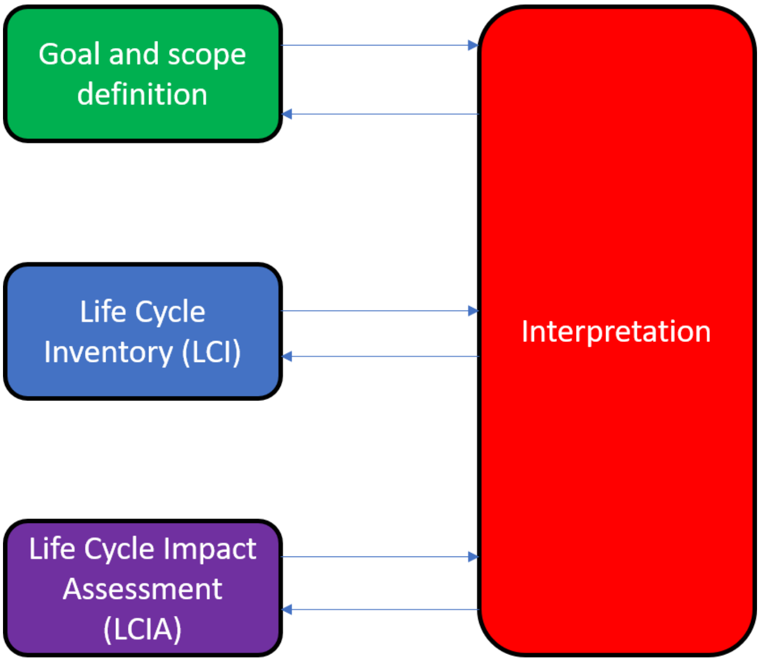

Figure 1.2 Framework of LCA (Source: personal elaboration based on UNI EN ISO 14040:2021).

1.1.2.1 Goal and scope definition

The study's purpose must be clearly specified from the beginning, including the motivation for doing it, its intended use, and the target audience. If decision support is the main purpose, the study's intended decision to inform must also be specified. The research must clearly define the product's functional specifications, quantified by the functional unit, and outline the activities and operations to be included inside the system boundary.

The aim and scope definition must include the principles for allocation, data quality standards, and selection of effect categories and technique for impact evaluation.

The functional unit (FU) measures the performance of the product being analysed. The main goal is to establish a standard reference point for normalising input and output data; hence it is crucial to explicitly describe and quantify the FU. LCA studies are typically conducted to compare different methods of providing a specific function, with the FU serving as the foundation for this comparison. The outcomes of an LCA study are closely tied to the chosen FU, hence it is important not to isolate FUs from the results.

The system boundaries delineate the technical system, comprising all operations involved in or influenced by the product's life cycle, from the external environment (Figure 1.3).

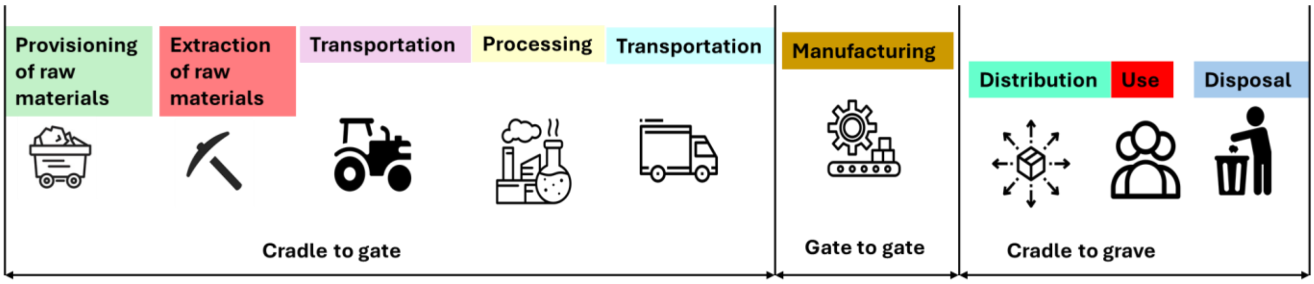

Figure 1.3 System boundaries (source: authors' elaboration based on (Wolf et al. 2010)).

Technical flows refer to the transfer of material or energy between processes, while elementary flows are the transfers that traverse system boundaries between processes and the surrounding environment. An elementary flow refers to the direct entry of material or energy from the environment into a system without any prior human modification, or the exit of material or energy from the system without any subsequent human manipulation. The technological system should ideally be represented so that all inputs and outputs are basic flows. However, this is impractical due to time and other constraints, and often unfeasible due to lack of data; hence, decisions must be made regarding which processes to include.

The processes can be grouped into three categories, that are upstream processes which represent the processes that involve the provisioning of material and energy, the core processes which the represent the main processes that involve the product or the service under examination and the downstream processes which involve the processes for the distribution and the disposal of the product or the service (Figure 1.4).

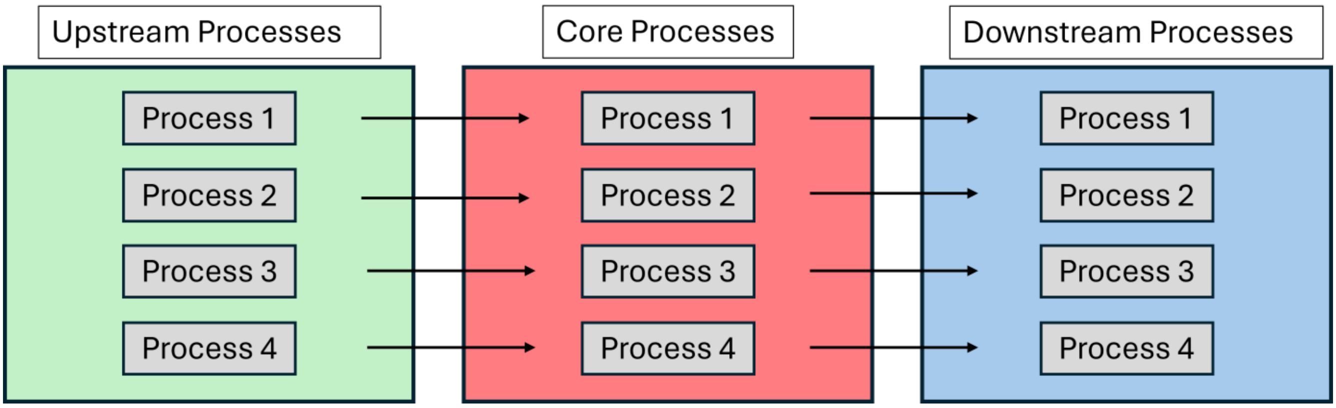

Figure 1.4 Upstream, Core and downstream processes (source: authors' elaboration based on (Wolf et al. 2010)).

The principles were developed in 1999 by the SETAC working group on inventory methodology enhancement. A further classification is based on the type of processes that can be divided into background and foreground processes. The foreground system includes processes directly affected by study-based decisions, while the background system consists of all other processes that interact with the foreground, typically by providing or receiving material or energy. The differentiation between foreground and background does not suggest any distinction based on the significance of the burden associated with those systems - the environmental burdens of either one can be substantial. Clift et al. (2000) suggest using primary data for the foreground and secondary data for the background when dividing the technological system.

1.1.2.2 Life Cycle Inventory – LCI

The UNI EN ISO 14040:2021 defines inventory analysis as the process of collecting data and doing calculations to measure the inputs and outputs of a product system. Through inventory analysis, it is able to predict in advance the significance of resource utilisation and emissions to the ecosystem.

The inventory analysis begins with collecting data about the foreground system to be utilised as input for the modelling phase. The final inventory should correspond to the reference flow that meets the previously determined FU. Life Cycle Inventory (LCI) typically requires a significant amount of time and effort due to challenges in obtaining high-quality data for all processes. Not all data may be necessary for achieving the study's purpose. In some cases, cut-offs, i.e., the practice of neglecting certain processes through a criterion (mass, energy or environmental relevance) and a threshold of non-relevance, generally 1% can be applied if they do not affect the results of the impact assessment phase (LCIA). LCA practitioners often face challenges related to managing the multi-functionality of processes, which occurs when a process offers many services or produces multiple outputs. The ISO offers a hierarchy to be followed in order to tackle this challenge. The final stage of the Life Cycle

Inventory (LCI) investigation involves developing models and computing LCI data, which are essential for assessing environmental impacts during the Life Cycle Impact Assessment (LCIA) phase.

Prior to commencing data collection, it is essential to identify all processes within the system being studied. Begin by identifying the unit process that produces the reference flow, which fulfils the function quantified by the FU. Then, analyse the processes both before and after the foreground system. The data utilised in the model should be as representative as possible of the system being studied, including its potential variations. It is advised that the data cover the entire production cycle, encompassing not only production but also waste generated during operations, as well as activities such as heating, calibration, cleaning, and maintenance over a specific timeframe. Data is typically gathered throughout a one-year production cycle and then adjusted to the chosen reference flow quantity. Collect data for every unit process inside the specified system boundaries. The ISO classifies this data as:

- Energy inputs, raw materials, ancillary inputs, other physical inputs.
- Products, by-products, and waste.
- Emissions to the atmosphere, releases into water and soil.
- Other environmental factors.

It is advisable to prioritise high-quality primary data obtained from direct measurements at a given location or derived from measurements at a specific site, notwithstanding the significant time and resources required to obtain them, over medium and low-quality primary data.

Companies may lack knowledge about the input and output flows of certain production units. In these instances, material and energy flows can be approximated by utilising information from similar processes occurring at different locations, technical reports, scientific literature, and LCI databases (secondary data sources).

During the system boundaries definition or inventory compilation, it may be discovered that certain operations have several outputs that serve as inputs for various supply chains. It might be challenging to determine the specific environmental effects alone linked to the desired outcome and even more challenging to design a standardised approach for practitioners to follow.

The UNI EN ISO 14044:2021 standard establishes a hierarchy of potential ways for managing multifunctionality in order to address this issue.

1. According to ISO, subdividing into subprocesses should be the primary method for addressing multifunctionality issues. Subdivision involves enhancing the amount of

information in a process to determine if it may be segmented into 2 or more sub-processes and adjusting the outputs accordingly.

2. The second strategy recommended by ISO is system expansion and accreditation. These two approaches to handling multifunctionality are conceptually different methods, although mathematically they are equivalent. In the first scenario, when comparing two or more systems and one offers an extra function compared to the other, this additional function is incorporated into the product lacking it to create two comparable systems. The credit technique involves removing the impacts associated with a monofunctional activity that serves a secondary purpose from a multifunctional system.
3. If the initial two procedures are not feasible, the ISO 14044 standard recommends utilising allocation. It refers to the mathematical allocation of inputs and outputs among products or functions. ISO suggests use allocation based on
   - Physical criterion: for example, based on the weight or volume of the various products
   - Economic criterion of each product.

1.1.2.3 Life Cycle Impact Assessment – LCIA

The Life Cycle Inventory's inputs are transformed into environmental impact indicators during the Life Cycle Impact Assessment (LCIA) phase of an LCA study. The assessment determines the extent of contribution of each flow (such as emissions or resource utilisation of a product system) to an environmental impact, providing indications for the next step, i.e., the interpretation phase. The goal is to render the outcomes more ecologically pertinent, understandable, and simpler to convey. This is accomplished by utilising impact categories and category indicators linked to the outcome of the inventory analysis. The number of inventory results factors might vary from a few dozen to hundreds, which can complicate the interpretation of the study's findings. Life Cycle Impact Assessment can decrease the number of parameters by categorising the environmental loads of the LCI into environmental impact categories. An environmental effect is a collection of alterations in the environment caused by human activity. The consequences are determined by a variety of qualitative and quantitative approaches that convert the results of the Life Cycle Inventory (LCI) into an understandable environmental outcome. 1 kilogramme of methane has a greater influence on climate change compared to 1 kg of CO_2, even when they are emitted in the same quantities, due to methane being a more potent greenhouse gas. Life Cycle Impact Assessment characterisation approaches model the environmental cause-effect chain for each impact category, starting from the environmental intervention to its impact. The outcomes of a LCIA should not be regarded as real environmental impacts. They merely indicate a possible influence. The UNI EN ISO 14040:2021 standard classifies sub-stages into mandatory and optional (Figure 1.5) and in particular they are:

- Selection of impact categories, category indicators and characterisation models
- Classification
- Characterisation,
- Normalisation,
- Weighting
- Grouping.

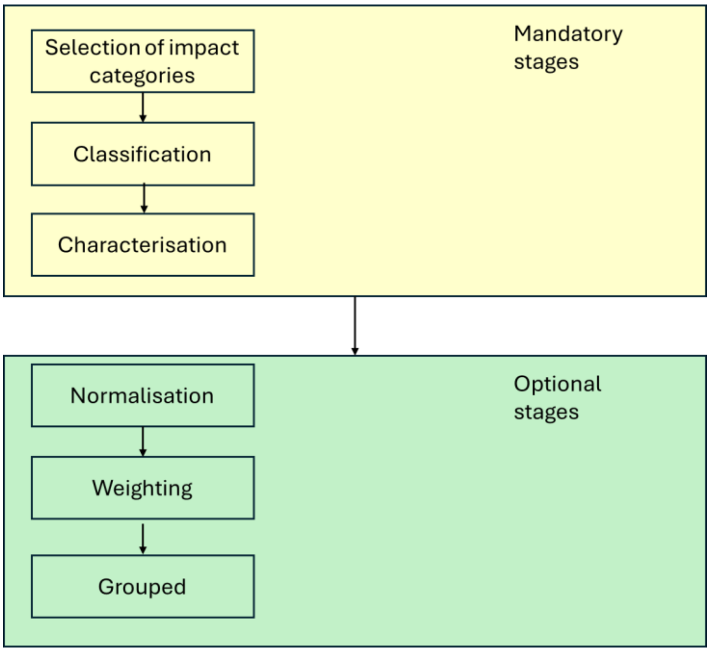

Figure 1.5 LCIA stages (source: author's elaboration based on (Wolf et al. 2010)).

ISO requires the first three sub-phases, which are the indicator results for various impact categories. Normalisation, weighting and grouping are optional and should be utilised based on the research LCA's purpose and field of application.

1.1.2.3.1 Selection of impact categories

The selection of impact categories, category indicators, and characterisation models is aimed at identifying the most effective and practical ones for the study's objective. The choice of effect categories should align with the study's objective to accurately guide the gathering of information on the pertinent elementary flow in the Life Cycle Inventory (LCI). This is completed during the scope definition step prior to gathering inventory data to ensure that the data collected is focused on what will be evaluated in the end. Determining the criterion that specifies what is essential for the study is frequently not simple. The UNI EN ISO 14044:2021 provides criteria in the form of requirements and recommendations. The UNI EN ISO 14044:2021 specifies that impact categories must be non-redundant, avoid double counting, avoid hiding important effects, be finite and allow traceability.

Furthermore, this list is complemented with mandatory criteria. The selection of impact categories, category indicators, and characterisation models must be consistent and justified with the study's purpose and scope based on these criteria. The analysis must encompass all environmental challenges associated with the product system to avoid moving problems from one impact category to another. The document is thoroughly referenced with all information and sources cited, typically by including the name and version number of the LCIA method utilised along with references used to develop the method. During a typical LCA procedure, a collection of category indicators derived from specific characterisation models are included into sets or methodologies known as LCIA approaches. The sets such as ReCiPe, CML, TRACI, EDIP, LIME, IMPACT 2002+, etc., are commonly used in LCA software (Hauschild et al. 2011; Hauschild et al. 2013). With the growing availability of LCIA sets and indicators, selecting the most appropriate one for a practitioner's study necessitates a thorough awareness of the key properties of these methodologies. It is important to consider the evolution of these methods, as they undergo constant upgrades and merging.

1.1.2.3.2 Classification

During this step, the Life Cycle Inventory (LCI) results are organised and allocated to the impact categories they influence. Understanding the environmental consequences of pollution and resource utilisation is necessary. Typically, this process is carried out using pre-compiled LCA software, which relies on categorisation tables. It necessitates a profound comprehension of the characteristics of the chemical being analysed and the various pathways and changes it undergoes post-emission.

1.1.2.3.3 Characterisation

During the characterisation phase, the amount of a certain substance is multiplied by its relative effect factor for a particular environmental category that takes the name of Characterisation Factor (CF) (Equation 1.1).

Equation 1.1 Impact score formula.

$$IS_i = \sum m_j \cdot CF_{i,j}$$

Where:

IS_i is the impact score for the category i

m_j is the mass of the substance j

$CF_{i,j}$ is the characterisation factor of the substance i for the category j

The characterisation can be conducted at either the mid-point or end-point level. Characterisation at midpoint level involves identifying impact indicators that collect elementary flows from a life cycle inventory based on their potential to contribute to a specific environmental consequence. This characterisation results in the creation of an impact profile for the product system, which can be presented as the outcome of the LCA research or utilised as a foundation for evaluating the impact at an endpoint level. Additional modelling is needed to extend a midpoint indicator to an endpoint indicator. This stage of modelling is commonly referred to as damage or severity modelling. An endpoint indicator interprets the output of midpoint indicators into several subjects or Areas of Protection (AoP) related to human health, natural environment, and natural resources. The endpoint indicators represent the final stage of the cause-effect chain in the environmental mechanism triggered by the elementary flows analysed in the LCI, whereas midpoint indicators are situated in the middle (refer to Figure 2.4).

All endpoint indicators, associated with a certain AoP, have a common unit. Their combined contribution can be aggregated to provide a comprehensive impact score per AoP, with or without weighting. Midpoint indicators allow for aggregate and contribution analysis of several effect categories only after normalisation and weighting.

The most of environmental categories considered in an LCA study are

- Abiotic Depletion Potential (ADP), that allows the assessment the potential depletion of non-living resources such as minerals and fossil fuels.
- Global Warming Potential (GWP), which quantifies the warming potential caused by greenhouse gases.
- Ozone Depletion Potential, which considers the change in the stratospheric ozone layer.
- Acidification Potential (AP), that calculates the potential for acidification in aquatic and terrestrial habitats.
- Photochemical Ozone Creation Potential, which takes into account the substances that contribute to the increasing of tropospheric ozone layer.
- Eutrophication Potential (EP), that determines the impacts due an increasing of nutrients in both aquatic and terrestrial environments.
- Human toxicity, which consider the negative impact for human.
- Eco-toxicity, which consider the negative impact for ecosystems and can be divided in Freshwater ecotoxicity, marine ecotoxicity and terrestrial ecotoxicity.

1.1.2.3.4 Normalisation

During the normalisation process, the characterisation outcomes are associated with the scale of influence within a specified geographic area or industry for each effect category. This stage aims to enable the comparison of the magnitude of the system's impact across several impact categories. It can be conducted at both the midpoint and endpoint levels. Post normalisation, the LCIA output reveals the magnitude of the impact of the product system being studied in relation to the reference system.

The Normalised Impact (NI) is determined by multiplying the product system's impact by the normalisation factor (NF) (Equation 1.2).

Equation 1.2 Normalisation formula.

$$NI_i = IS_i \cdot NF_i$$

Where:

i is the i-th impact category.

1.1.2.3.5 Weighting

Weighting is used to equalise the significance of impacts from the normalisation process. It can only be utilised when the effects have been standardised. The effects are prioritised by providing varying or equal weighting factors to each impact type. The criteria for weighting lack scientific foundations, making this phase purely subjective. This stage is beneficial for comparing impact categories or presenting results in a way that supports the prioritising of ethical principles.

1.1.2.3.6 Grouping

This stage aims to group impact categories to enhance the readability and comprehensibility of the analysis output. Groups or clusters can include consequences at global, regional, or local levels, or impacts categorised by high, medium, or low priority. The grouping often involves implementing two methods: Sorting and clustering midpoints affect categories geographically. Ranking the impact categories based on a specified priority hierarchy.

1.1.2.4 Interpretation

After receiving the LCIA results, practitioners must analyse and explain them, considering the study's objective, key assumptions, potential errors and uncertain parameters, and the modelling approaches used. The LCIA results are often analysed to conduct a hotspot analysis across various life cycle stages, followed by a detailed examination of each process within those stages. The hot spot analysis will identify the primary factors influencing the outcomes in terms of life cycle phase, processes, and flows.

Prior to quantifying errors, it is advisable to do a completeness check to ensure that all necessary information and data are present for accurate interpretation. According to the ISO guidelines, if information or data is absent or incomplete, it should be determined whether it is essential to meet the objectives and scope of the LCA. If needed, recover the information or modify and alter the goal and scope accordingly. If the information is deemed unnecessary, it should be justified and documented. After verifying completeness, the ISO suggests conducting a sensitivity analysis to assess the impact of input uncertainties on the outcomes, although it does not specify a particular technique for this purpose. Uncertainty analysis is used to determine the confidence interval of the results.

Both analyses can be used for both the LCI and LCIA phases.

1.1.3 *Life Cycle Costing – LCC*

Conventional life cycle costing (LCC) is a widely accepted methodology used to determine the total expenses incurred across the entire lifespan of a product or service. The origins of the LCC are exposed in "Environmental Life Cycle Costing" by Hunkeler et al. (2008) and they may be traced back to 1933 when the General Accounting Office (GAO) initially used it to apply for the purchase of tractors, which included both operating and maintenance costs. During the 1970s, the US government required LCC to buy expensive weaponry and military equipment such as planes and tanks. This mandate also applied to construction projects for public institutions in various American states (Society of Automotive (SAE) 1992). During that time, Europe also developed an inclination for LCC, especially in relation to the public sector. The rationale behind the LCC's interest was that procurement decisions should consider not just the upfront purchase cost, but also the expenses associated with operation, maintenance, and, to a lesser degree, disposal.
Another reason for using traditional LCC analysis is to ensure the most efficient allocation of funds over the lifespan of a system, product, or item, and to optimise business performance. Expanding upon this established practice, LCC has primarily been utilised for decisions pertaining to the procurement of durable goods or items with significant individual investment expenses. The initial domains of implementation encompassed (Sherif and Kolarik 1981):

- Structures, primarily for commercial or public utilisation.
- Energy generation and consumption.
- High-cost transportation vehicles (mostly derived from the aerospace industry).
- Primary military equipment and weaponry systems.

In the early 1970s, various principles were established for LCC and cost design. These principles take into account the expenses associated with a product or system over its entire life cycle, starting with the research and development phase (Office of the Secretary of Defense Washington DC 1973). In addition, the United States has implemented many policies that mandate the computation of life cycle costs when acquiring public buildings (Zehbold 1996). These two instances demonstrate that LCC has typically been restricted to applications that are specialised to industries or products. Sherif and Kolarik (1981) present a thorough examination of the applications listed earlier, the cost models employed, and the relevant literature. The authors observe that the development of LCC was driven more by practical applications rather than theoretical models. This conclusion remains essentially valid in the present day. Nevertheless, there is currently no universally applicable methodological framework or model that has been produced, despite some emerging tendencies in this area (Rebitzer 2005).

The most similar approach to a comprehensive LCC method was first introduced by Blanchard (1978) and later improved by Blanchard and Fabrycky (1998). Additional examples can be found in the ISO 15663 standards (International Standard for Organization (ISO) 2000), the IEC 60300-3-3 standards (International Electrotechnical Commission (IEC) 1999), and the AS/NZS 4536 standards (International Standard Organization 1999). These methodologies originate from systems engineering and prioritise the assessment and comparison of technology options. The life cycle of a product or system is organised into several phases: research and development, manufacture and construction, operation and support, decommissioning and disposal. The framework of this methodology, along with the cost categories and systems view, closely resembles the life cycle approach in life cycle management (LCM). As a result, it provides a solid foundation for the creation of a LCC method for LCM. Blanchard and Fabrycky (1998) and the standards do not offer a specific method for calculating and comparing costs. Instead, they focus on the concept of “life cycle thinking” and emphasise the significance of taking systems view.

In summary, it can be stated that traditional LCC has not been clearly developed into a comprehensive and universally applicable technique. It was created using the life cycle vision, with the intention of implementing customised procedures for use in particular sectors. Given these circumstances, one may question why this straightforward concept has not been widely adopted in industries and the public sector, including quality management. In recent times, certain industrial sectors, like the railway sector, have started to acknowledge the growing significance of LCC in their procurement and maintenance decision-making processes. A common reason for not implementing a conventional LCC method in typical commercial applications is the lack of compatibility between the cost system of the firm or government organisation doing the LCC research and the specific needs of the buyer, such as a railway buyer. The LCC approach must align with the cost data used by a company. Broadening the scope may reduce efficiency, and developing a more comprehensive LCC approach requires converting company-specific cost data into standardised data.

As the emphasis on systems thinking increases, LCC is expected to receive more recognition in the future, especially in the areas of process planning and control. It is also considered one of the key components of sustainability analysis, along with LCA and societal evaluations.

Conventional LCC methodologies, or cost management practices, are typically inadequate for evaluating the economic consequences of a product's life cycle within a comprehensive sustainability framework. For instance, these systems may be deficient in certain aspects, such as when there is a lack of management for end-of-life (EoL) processes and when the environmental consequences or impacts cannot be consistently connected to the same system specifications used for environmental analysis. The LCC must ensure that all costs associated with a product or system throughout its full life cycle are taken into account in the decision-making process. This approach enhances transparency in decision-making and helps prevent early environmental degradation and financial imbalances. Therefore, it may be inferred that traditional LCC methods must encompass the full life cycle and should be broadened to establish stronger links with other dimensions of sustainability, such as environmental and, if desired, social factors. The primary inquiry revolves around the effective integration of expenses and environmental factors.

The SETAC-Europe working group on LCC has established the definition based on three typologies (Hunkeler et al. 2008):

• Conventional LCC represents the summation of all costs associated with the entire lifetime of a product, assuming those costs which the manufacturer/user directly incurs. The actual internal costs are majorly considered for evaluation, which sometimes exclude end-of-life or usage costs perhaps the external bodies might cover. Conventional LCC studies do not typically incorporate stand-alone LCA findings. From this perspective, the main focus is on the participant of the market, either a producer, user, or consumer.

• Environmental LCC: The calculation of all costs associated with the whole life cycle of a product that are directly borne by any actor along the product's life cycle (supplier, producer, user or consumer and/or other relevant actor) considering also externalities that are expected to be internalised in the decision-making process in the future (according to the definition given by Rebitzer and Hunkeler (2003)).

In that, the environmental LCC goes beyond the conventional LCC in insisting that all life stages and all costs must be included in the decision-making process, upfront costs, and future consequences, and non-monetary LCAs must be excluded. Both are to be product system-based according to ISO 14040/44 (2006). The perspective is that of one or more market participants, mainly producers. If applicable, the environmental LCC includes subsidies and taxes.

•Societal LCC refers to the comprehensive assessment of all expenses related to the entire lifespan of a product, which are borne collectively by the members of society, both presently and in the far future. The social LCC encompasses all aspects of the environmental LCC and also include an evaluation of supplementary external expenses, typically measured in monetary terms (e.g., using willingness-to-pay techniques). The perspective involves the whole of society, both nationally and internationally, including governments. Unlike the environmental LCC, subsidies and taxes cancel each other out in terms of net cost and are therefore excluded in social LCC.

All three types of LCC have a system perspective that is focused on functions, indicating the presence of a life cycle approach.

Many techniques for doing conventional LCC analysis have been documented in the literature, including studies by Dhillon (1989), Ellram (1993, 1994, 1995), Fuller and Petersen (1996), Riezler (1996), Zehbold (1996), and the Australian Department of Defence (1998).

Two significant conventional costing methods that are strongly associated with LCC are total cost of ownership (TCO) and activity-based costing (ABC). Total Cost of Ownership (TCO) assists customers and business managers in assessing the complete expenses associated with utilising a certain item Ellram (1993). The user or consumer TCO and LCC share common elements, since TCO takes a comprehensive user viewpoint that particularly considers the purchase and usage phase, including investment, maintenance, operation, support, and other related factors. Activity-based costing (ABC) assists producers in accurately determining the total expenses associated with a certain item. Nevertheless, neither approach incorporates an environmental evaluation nor considers external costs. Additionally, ABC typically fails to adopt a life cycle viewpoint, therefore it is not qualified as an LCC.

1.1.3.1 Environmental Life cycle costing

An argument supporting the use of environmental LCC is that evaluation methodologies like LCA are frequently perceived as hindrances to business growth, especially in the immediate future (Rebitzer and Hunkeler 2003). A new methodology that provides a robust combination of the environmental and economic performance of a product can help drive technological development and managerial decisions in a more rational direction, identifying advantageous situations for all and optimising the trade-offs between the environmental, economic, and entrepreneurial perspectives.

The environmental LCC is encompassed by a conceptual framework that elucidates the connection between LCC and LCA, as well as social evaluations such as working conditions, unemployment rates, and overall social consequences on communities in the LCM.

The environmental LCC framework is founded on the physical life cycle of the product, which encompasses five distinct phases that can be further detailed if required: research and development, material or component production, product manufacturing and development, product use and maintenance, and EoL management.

There are two distinct categories of expenses that can be identified (Rebitzer and Hunkeler 2003):

Internal costs refer to the expenses incurred during the whole lifespan of a product. These costs are borne by individuals or entities directly involved in the manufacture, transportation, consumption, or other aspects of the product's life cycle. As a result, internal costs can be directly associated with the overall business expenses. This notion of cost encompasses all expenses and income inside the economic system. Internal costs can be categorised as either costs that are internal or external to an organisation, depending on one's perspective.

External costs, also known as externalities, are already quantified in monetary terms and are considered in decision-making for the future. These costs are internalised and do not require translation between environmental and monetary measures. Externalities should not be counted twice in the LCC and complementary LCA.

1.1.3.2 Societal LCC

The societal LCC assesses the expenses related to the whole lifespan of a product that are borne by the entire society. In a social LCC, the analysis takes into account not only the direct financial transactions but also the externalities, which refer to the changes in value resulting from a business transaction but not yet reflected in its price or benefit. The environmental LCC analysis may already incorporate the quantification of external costs in a future that is relevant for decision-making (such as forthcoming expenses for CO_2 emissions trading certificates or expenses related to adapting to global warming). Social LCC extends beyond this notion by taking into account all external factors that can be quantified in monetary terms, such as the costs of damages evaluated in the ExternE project (Bickel et al. 2005), as well as those that are challenging to quantify monetarily and can only be assessed qualitatively, such as public spending on health and social welfare, standards and rights pertaining to work quality, and personal and family life. The limits are evident due to the presence of several externalities that may be quantified monetarily. Additionally, an externality can only be considered if it can be identified and if a participant in the product life cycle is aware of it and demonstrates interest and concern.

The research above demonstrates that social LCC is employed to measure the monetary value of environmental impacts on society. It can be seen as a valuable idea for connecting environmental life cycle approaches to corporate social responsibility or decision making. However, this connection is subjective and dependent on personal preferences and viewpoints. The benefit of having a solitary score in relation to wellbeing must be balanced against the considerable uncertainty surrounding the assessment of social impacts, if they are ever known, as well as practicality and transparency. Therefore, it may be more preferable to keep the social effect assessment score distinct, so that it can be compared with other expenses. The latter would be in line with the Brundtland definition of sustainable development (Brudtland Commission 1987), i.e., the analysis and balancing of sustainability implications can only be done if the social, environmental, and economic dimensions are considered separately. It is advisable to give separate results for distinct effect categories instead of a single data, and to conduct sensitivity analyses, as is also done for the LCA. This proposal proposes that a government organisation, which is the primary audience for social LCC, should take into account all the advantages and disadvantages that a decision, legislation, or policy may have on society. As an illustration, the European Commission applies the 3-pillar concept in its policy impact assessments. These assessments are divided into independent evaluations of the economic, environmental, and social impacts (European Commission 2005).

Out of the three categories mentioned, the framework following will be based on the environmental LCC. This choice is made because the LCC is not considered as a separate technique, but rather as an analysis that complements the environmental LCA. The subsequent procedures may be pertinent in attaining a uniform environmental life cycle cost, but the precise analytical procedures may differ depending on the circumstances:

1) Goal and scope definition.
2) Information gathering.
3) Interpretation and identification of hotspots.
4) Sensitivity analysis and discussion.

1.1.3.3 Environmental LCC framework

1.1.3.3.1 Goal and scope definition

Prior to conducting a study, it is necessary to establish the purpose and extent of the environmental LCC. It is crucial to accurately establish the limits of the system and the unit of measurement for its functionality. After examining multiple expert viewpoints, the following potential supplementary goals for LCC have been identified:

- Determine the overall expenses incurred by the actor, such as a company or a consumer.
- Assess the level of competitiveness of the product by considering the costs associated with its consumption, including the cost of ownership.
- Implement reporting, monitoring, and proactive measures to control costs within companies.
- Obtain management-level consensus on the development and selection of the product portfolio and establish the connection between the LCC and the product portfolio.
- Identify potential alternatives for the development or commercialisation of the product.

- Identify the economic and environmental trade-offs and opportunities for mutually beneficial outcomes. Discuss the concept of corporate social responsibility (CSR), specifically when it is carried out in conjunction with social evaluations.
- Identify a viable business case and analyse the long-term costs, while also assessing the potential economic rewards for consumers and the potential dangers in terms of economics, environment, or society at the conclusion of the product's life cycle.
- Evaluating trade-offs between different criteria, such as balancing future expenses with current expenditures or considering internal costs versus external costs.
- Identifying lifetime optimisation concerns, such as determining the need to modify maintenance schedules for a purchased product.

Life cycle costing (LCC), similar to LCA, typically involves comparing several options. These options can include fundamental alternatives, or they can arise from variations in activities within a particular process in an otherwise identical life cycle. In the second scenario, the process of gathering data is made easier as it only requires specifying the variations in cost.

It is crucial to emphasise that the choice of alternatives must align with the FU specified in the LCA as described in UNI EN ISO 14040:2021 and UNI EN ISO 14044:2021.

1.1.3.3.2 Information gathering

If there is a lack of essential data, it may be necessary to employ scenario development, forecasting, or other estimation techniques, such as those used in LCA. Thankfully, the discipline of cost estimating is highly established (Dhillon 1989). Moreover, thresholds can be implemented, as exemplified by the use of LCA. Typically, a process-specific threshold of less than 5% related to the mass of input materials can be disregarded (i.e. used as a limit) in this scenario, since it would normally have little impact on the LCA outcome (Rebitzer 2005). Currently, LCC has not assessed these thresholds. However, a cautious estimate for a specific lifecycle stage (such as manufacture, transportation, or usage) could be 1% of the entire cost of that stage. Nevertheless, if this cost estimation has been conducted, it may be incorporated as a fixed amount without any additional breakdown.

A challenge in data collection occurs when allocating costs to products that are produced alongside co-products or by-products. Within the LCC framework, it is necessary to distribute the expenses related to staff, capital, and purchased products and services among the many outputs involved, taking into account their respective market values. The cost allocation method commonly used is the gross sales value method, which involves allocating costs to a designated “split point” depending on revenue share and adjusting all upstream costs accordingly (Huppes 1993).

1.1.3.3.3 Interpretation and identification of hotspots

Both LCC and LCA aim to identify hotspots as a significant result. These critical areas typically become apparent as a consequence of the evaluation, particularly when a sensitivity analysis is conducted. The analysis of these areas of intense activity might be measured in terms of quantity or described in terms of quality. Quantitative analysis in investment valuation can utilise classical approaches such as net present value, returns, internal rate of return, and payback time.

The evaluation of different systems is often influenced by non-financial (qualitative) factors. An alternative that appears ideal based on quantitative analysis may be rejected due to factors such as insufficient market appeal, leading to inadequate sales volume. Only alternatives that outperform others in both quantitative LCC and qualitative evaluations are considered truly favourable. When scores differ, the decision-maker must evaluate each option separately. Several decision support systems have been designed for LCM, integrating both quantitative and qualitative factors.

Sensitivity analyses are not mandatory for LCA, although they are recommended during the interpretation phase and rarely used in conventional LCC. It could be hoped, and perhaps expected, that a definitive code of conduct or LCC standard would enforce sensitivity studies for environmental and societal LCC.

The relationships between uncertain parameters in LCC (e.g., project lifespan, lifecycle costs, revenues, sales volume, and discount rates) and the calculated outcomes (e.g., net present value) should be highlighted using sensitivity analysis. The central question is how variations in input parameters affect the results. To analyse this, the uncertain input parameters are varied ceteris paribus by a certain percentage, and their effects on the output parameters are observed. As a result, output variations caused by input changes can be determined. Sensitivity analysis also provides an answer to the question “To what extent can input values vary without affecting the conclusions regarding the comparison of different options or causing the output values to deviate from a certain value?” A key limitation of sensitivity analysis is that only one input can be modified at a time. The Monte Carlo simulation addresses this issue, but it is significantly more complex and less transparent. Sensitivity analysis should underpin the final discussion and guide recommendations.

1.2 **Concepts**

Among the various aspects of sustainability assessment, one critical challenge is the integration of ESs within quantitative evaluation frameworks. Despite the extensive application of LCA and LCC in environmental and economic assessments, these methodologies often fail to consider the direct and indirect contributions of natural ecosystems to human activities. This limitation has led to an increasing focus on incorporating ecosystem service assessments within LCA to provide a more comprehensive perspective on sustainability.

1.2.1 *Terrestrial ecosystems*

Before concluding this introduction and exploring the integration of ESs in Life Cycle Thinking, along with the clarification of techniques and their application, it is advantageous to present an overview of various ideas intended to define soil in LCA.

Life cycle inventories and impact assessments generally consider two categories of land use interventions: land transformation and land occupation (Lindeijer 2000; Baitz 2002; Lindeijer et al. 2002; Milà i Canals et al. 2007; Beck et al. 2010; Koellner et al. 2013b; Bos et al. 2016; Milà i Canals et al. 2007a). Land transformation, or land use change (LUC), entails modifying the attributes of a parcel of land to render it suitable for a designated purpose, such as deforesting or draining areas to establish agricultural fields. The changeover period is short, and the temporal aspect is overlooked. During land occupancy, the land is employed for its initial productive function, such as serving as an agricultural field. Furthermore, actions are implemented to maintain the distinct attributes of the land, including the prevention of forest regrowth on agricultural fields.

Land use interventions impact the quality of ecosystems, referred to as Q, over a designated period. Q denotes the capacity of an ecosystem, or a collective of ecosystems at a landscape scale, to maintain biodiversity and provide services to humanity. Community. The phrase “protected natural environment” denotes an area that has been conserved to maintain the intrinsic worth of nature, encompassing ecosystems and species, together with the economic benefits gained from the services that sustain life. This notion was articulated by (Udo de Haes et al. 1999). Diverse metrics can assess the impact of land use and ecological degradation on biodiversity and natural environments. These indicators can convey the intrinsic value of biodiversity and natural landscapes, alongside the functional value of ecosystems regarding assets (e.g., timber or food) and services (e.g., climate regulation or erosion control). This theme aligns with (Milà i Canals 2007). These effects stem from both the occupation and alteration of the region. The employment impacts the quality of the environment, causing it to diverge from its natural condition. The deliberate alteration of the landscape changes the attributes of the ecosystems.

Figure 1.6 illustrates how land use is considered in Life Cycle Methodologies. It is assumed that at the time t1, the Ecosystem Quality Q changes from Q_{ref}, i.e., the ecosystem quality of a reference situation, to Q_{LU1}, i.e., the ecosystem quality with a certain land use that increase the Q of reference. If no occupation process would take place, the ecosystem quality would change toward the initial situation and the impact of land transformation would calculate as the integral of the difference between the two ecosystem qualities by the time. The temporal dynamics of ecosystem quality are unknow and they are assumed to be linear. As consequence, the impact of land use transformation ($TI_{ref \rightarrow LU1}$) will be (Equation 1.3):

Equation 1.3 Transformation impact formula.

$$TI_{ref \rightarrow LU1} = 0.5 \cdot (Q_{ref} - Q_{LU1}) \cdot (t_3 - t_2) \cdot A$$

The difference between t_3 and t_2 is called "time of regeneration" and it is the time necessary for the complete regeneration of ecosystem quality; it is not equal for all land use change and depends on the land use. As it can be observed, the time for change the Q is not considered because the land change is assumed to be fast.

When there is occupation, some activities that do not allow the complete restoration from the previous Q take place. This is, for example, the case of an arable camp that plough and does not allow the reforestation of trees. The occupation can take place for a certain time before allowing to nature forces to repristinate the Q_{ref}. In this case the impact of land occupation (OI_{LU1}) will be (Equation 1.4):

Equation 1.4 Occupation impact formula.

$$OI_{LU1} = (Q_{ref} - Q_{LU1}) \cdot (t_2 - t_1) \cdot A$$

Equation 1.3 and Equation 1.4 can be applied also for the other part of the graph. When transformation and occupation produce a Q higher than reference situation, the impacts are expressed in negative values, while when they are lower, the impacts are positive. In the graph the time of modelling finish in t7 when there is the complete restoration of ecosystem quality. If the time of modelling ($t_{modelling}$) finished before the restoration of the initial ecosystem quality ($Q_{ref,\ i}$ at t_7), there would be a new Q_{ref} ($Q_{ref,\ f}$) and the difference between the two Q_{ref} ($Q_{ref,\ i}$ and $Q_{ref,\ f}$) is called "permanent impacts". This term does not indicate the impossibility of restoring the initial Q but rather a longer recovery period that is not accounted for.

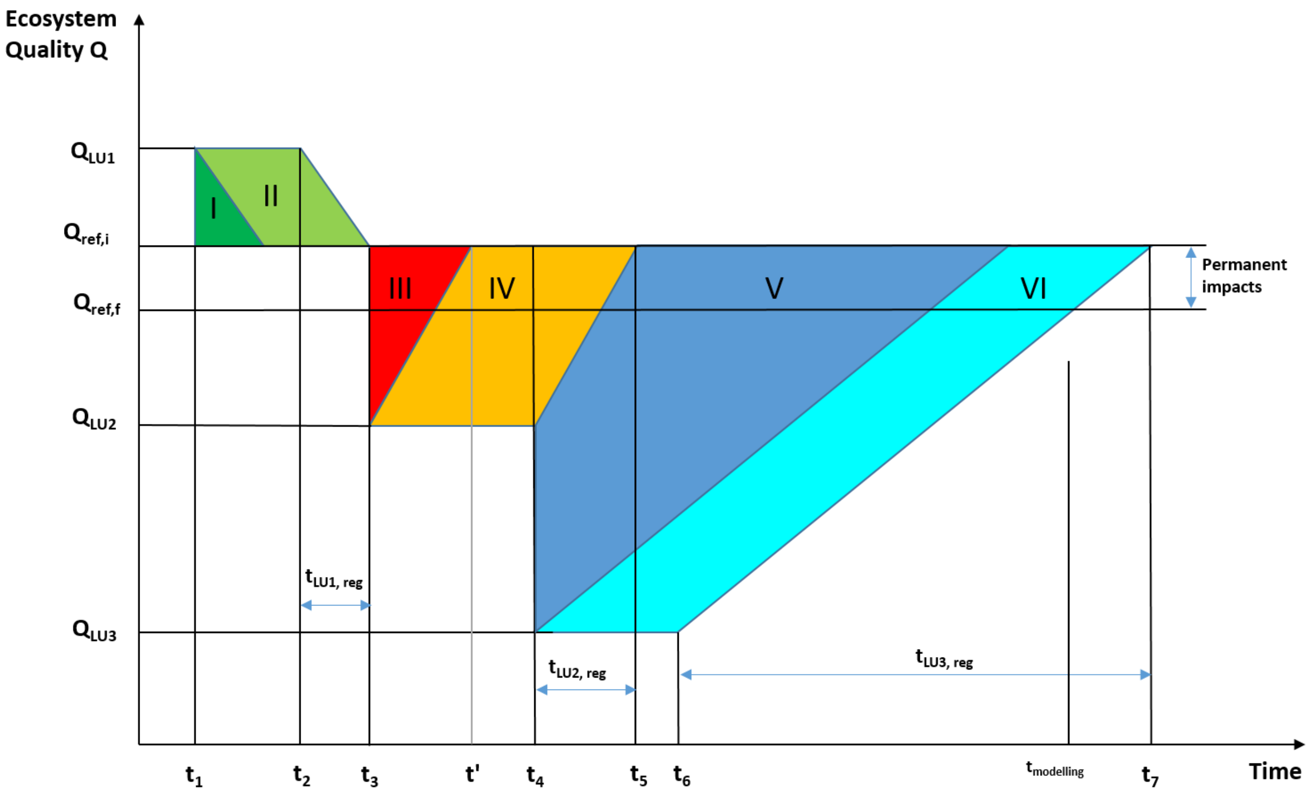

Figure 1.6 Explanation of how land use impact are taken into account in LCA (source: personal elaboration based on Koellner et al. (2013)).

Considering the figure above, it is necessary to understand what is meant by a reference situation. Koellner et al. (2013) suggest using one of the lists below:

1) Potentially natural vegetation refers to the anticipated condition of the vegetation in the absence of human interventions, as defined by Chiarucci et al. (2010).
2) The quasi-natural condition in each biome refers to the composition of natural vegetation found in that biome, as described by Koellner et al. (2013).
3) Present condition: existing distribution of land use categories Koellner and Scholz (2008).

Options 1 and 2 are sufficiently similar for the purpose of determining the lowest common ancestor (LCA). Global data on the characteristics of potential "natural" land in various biogeographical regions are available with varying degrees of quality. This allows for a global assessment of the effects of human activities on land use, compared to the idealised "natural" land cover. However, the current combination of land uses is constantly changing and would not be particularly feasible to define precisely.

To summarise, it is advisable, in line with the findings of Milà i Canals et al. (2007), to consider the prevailing (almost) natural land cover in worldwide biomes and ecoregions as a benchmark for evaluating the influence of land use on a global level. However, the process of establishing a reference situation is an aspect that requires additional investigation and is acknowledged as a valuable decision. The choice between options 1 or 2 compared to option 3 facilitates different sorts of decision-making (refer to Milà i Canals et al. (2013)).

The significance of selecting a reference condition has been extensively illustrated in several research investigating the effects on land use. These studies include the works of (Milà i Canals et al. 2007; Koellner et al. 2013b; Arana Benitez and Bos 2015; Bos et al. 2016; Cao et al. 2017). Arana Benitez and Bos (2015) and Cao et al. (2017) discovered that the result is highly dependent on the selection of the reference case.

Milà i Canals et al. (2007) and Koellner et al. (2013) concur that the quality of the ecosystem can vary over time during the occupation of the land. The erosion vulnerability in a recently formed plantation is higher compared to its vulnerability after five years. Moreover, the reference scenario may not be inherently fixed. Natural regions undergo continuous and evolving evolution (Bork 1998). The incorporation of dynamic professions and reference scenarios gets quite intricate. Thus, the authors first assume static conditions (Milà i Canals et al. 2007; Koellner et al. 2013b). Currently, there are no characterisation criteria that consider the changing situations of reference. The land use inventory flow for land occupation is measured in $m^2 \cdot y$ while for the land transformation is measured respectively in m^2. The methods used may vary in terms of the level of differentiation, including land use type and spatial scale. However, it is important to record both land use type and biogeographical information (Koellner et al. 2013a). The classification of land use type and refinement of ecosystem type in a LCA study depends on the study's scope and whether it is conducted in the foreground or background system (Milà i Canals et al. 2013)

The length of this period of regeneration, that as mentioned before is an important variable in the quantification of land use impact, depends on:

1. The effect pathway: It is possible that it will take a longer time to restore biodiversity on a certain area compared to restoring the capacity for biotic output.
2. The nature of land transformation affects the time it takes for a forest to regenerate. If the land was transformed into sealed urban areas, it will take longer for the forest to regenerate compared to when the land was transformed into agricultural areas.
3. The biogeographical characteristics of a location significantly impact regeneration rates, with hot and humid climates often promoting faster regeneration compared to cold or dry climates.

There are several studies that try to measure the regeneration time for each ecosystem. Müller-Wenk and Brandão (2010) established the regeneration time (called in their work "relaxation time") for the category "Carbon Storage Potential". Koellner and Scholz (2007) quantify the regeneration time for the biodiversity, using (Bastian and Schreiber 1999) information.

The regeneration time depends on latitude (increasing toward poles), altitude, specific impact and ecosystem function (for ESs) or taxonomic group (for biodiversity). to occur tends to grow as one moves closer to the Earth's poles and at higher altitudes. The extent of the impact is further influenced by the specific sort of impact, as well as the particular ecological function or taxonomic group being studied (Jones and Schmitz 2009).

1.2.2 *Biodiversity*

According to Conventional Biological Diversity (CBD), biodiversity is composed by three elements:

- genetic, which refers to the variety inside a single gene pool
- species, which refers to the variety in an ecosystem
- ecosystem diversity, which refers to the variety of ecosystem in a defined region.

The integration of biodiversity in the Life Cycle Methodologies has been ongoing for over 20 years; the first attempts included the land use impacts analysis as a pressure on biodiversity (Koellner 2000; Lindeijer 2000). Despite the multitude of scientific work on the integration of biodiversity in LCA, some gaps persist such as the focus on species diversity and ecosystem diversity, and the failure to take genetic diversity into account, except in rare articles. The pressure that can influence biodiversity are increased since the first publication of Millennium Ecosystem Assessment (MA 2005). However, some of these pressures are not linked to a one or more impact categories (like light pollution) or the methods are not clear in the connection between pressure and impact on biodiversity (like noise pollution).

The CBD meeting on the Aichi Targets, where more than 100 distinct indicators were submitted, also demonstrated the wide range of indicators available to measure biodiversity. The framework of fundamental biodiversity variables aids in the selection of acceptable indicators by prioritising those that can capture the central aspect of biodiversity change. The six key categories of ecosystem structure, ecosystem functions, community composition, species populations, species attributes, and genetic composition have been highlighted as being crucial for evaluating biodiversity (Pereira et al. 2013; Dasgupta 2021). Three already existing indicators were proposed by Mace et al. (2018), which collectively address the key aspects of biodiversity:

1) The IUCN Red List Index, which evaluates extinction and vulnerability threats.
2) The Living Planet index, which assesses the species abundance.
3) The Biodiversity Intactness Index (BII), which measure abundance.

The most often used measure to determine the effect of land use is species richness (SR), an indicator that counts the number of species found in a certain area. Relative species richness (SR_{rel}), a measure that the SR indicator is frequently used for compares species richness to that of a reference region (SR_{ref}). The reference area is selected to match the region's natural state. The quantity of species in this "natural area" is then connected with all land-use practices. This means that (relative) species richness in a particular region j is defined as (Delft CE 2023) (Equation 1.5):

Equation 1.5 Relative species richness formula.

$$SR_{rel,j} = SR_j - SR_{ref}$$

Where:

$SR_{rel,j}$ is the relative species richness of region j

SR_j is the species richness of region j

SR_{ref} is the reference species richness.

In general, two important factors should be considered when evaluating biodiversity i.e., ecological model and metric, along with the previously listed ones such as vulnerability, endemism, rarity, and irreplaceability.

1.2.2.1 Ecological relationships

Ecological relationships are often used in LCA for the estimation of species loss due to certain human activities. Ecological relationships evolved in the course of time starting from the first model called classic species-area relationship (classical SAR or simply SAR) to species-habitat relationship. In the following paragraphs are reported a short description of main relationships including also strength and weakness points

1.2.2.1.1 Classical Species-Area Relationship

The first formulation of SAR is due to Arrhenius (Arrhenius 1920a; Arrhenius 1920b; Arrhenius 1920c; Arrhenius 1921). He proposed a potential and empirical relationship between the number of species and the size of the area where the species were present (Equation 1.6):

Equation 1.6 Species-Area Relationship formula.

$$S = cA^z$$

Where:

S is the number of species

A is the area considered

c is a constant related to the taxon type and unit of measure of geographical area, i.e., the population density of all species of the area considered

z is a constant related to the area type, and it varies from island to island or from island to continent in independent way from specific density. It can assume values between 0.2 and 0.3.

A version slightly modified of SAR is the Endemic Area Relationship (EAR) which the focus is a small part of all species, i.e., the endemic species. A critical issue that common SAR and EAR is that both relationships overestimate the extinction rate of species. This is caused by the assumption that the areas where are present human activities (e.g., agriculture) become completely hostile to host biodiversity. However, it is recognised and accepted that also modified habitats allow an important role in biodiversity conservation. In fact, there are some species that have a high sensibility to the loss habitat, other ones that are partially or totally tolerant to the modified habitats and others that have benefits from these modifications.

To try to overcome this problem, several modifications to SAR are proposed.

1.2.2.1.2 Matrix-calibrated Species-Area Relationship

Matrix-calibrated Species-Area Relationship (well-known as matrix SAR) is a first modification of classical SAR, proposed by Koh and Ghazoul (2010), which the focus is for z-parameter. In fact, the authors proposed to consider two components to z-parameter as reported in this relation (Equation 1.7):

Equation 1.7 z-parameters formula.

$$z = \gamma \cdot \sigma$$

Where:

γ is a constant

σ represents the sensibility of taxon to transformed habitat

However, this relationship suffers the same problem of classical SAR, considering a no adaption of species to modified habitat.

1.2.2.1.3 Countryside Species-Area Relationship

Because the problem of the previous relationships is the no adaptation to the anthropised habitat, Pereira and Daily (2006) proposed a new relation, known as countryside SAR which overcome this issue. The formula of countryside SAR is the following (Equation 1.8):

Equation 1.8 Countryside Species-Area Relationship.

$$S_i = c_i \left(\sum_j h_{i,j} A_j \right)^z$$

Where:

A_j is the covered area by habitat j

$h_{i,j}$ is the affinity of species group i to the habitat j

1.2.2.1.4 Species-Habitat Relationship

A further develop of SAR is Species-Habitat Relationship (SHR) proposed by Kuipers et al. (2021). This relationship considers the conversion and the fragmentation of habitat through the assessment of the probability from a species to pass from a fragment to another one. The following equation shows this relationship (Equation 1.9):

Equation 1.9 Species-Habitat Relationship

$$S = 1 - \left(\frac{\sum_i h_i ECA_i}{\sum_i h_i ECA_{i,ref}}\right)^z$$

Where:

h_i is the suitable of soil type i

$ECA_{i,\,ref}$ is the ECA of soil type i of the reference landscape

ECA_i is the equivalent connected area of a soil type i. The equivalent connected area, a concept proposed by Garcia-Ulloa et al. (2016), is weighted measure of area for its connectivity based on the dimensions of fragments and the probability of dispersal of species between the fragments. This probability derives from the distance of the fragments, the permeability of the landscape matrix and the ability of dispersal of species. ECA is equivalent to the total area if all fragments are connected while it is close to the size of the biggest single fragment for fragments more isolated.

1.2.2.2 Indicators

1.2.2.2.1 Potentially Disappeared Fraction (PDF)

The Potentially Disappearing Fraction (PDF) is the amount of species loss attributed to human activity within a certain region over a specific time period. Both terrestrial and marine species are included in this. The PDF for an area j compares the loss of species there to the baseline condition (Müller-Wenk 1998) (Equation 1.10):

Equation 1.10 PDF formula.

$$PDF_i = 1 - \frac{SR_i}{SR_{ref}}$$

PDFs are frequently created for a certain place and time. A $PDF \cdot m^2 \cdot yr$ of 1 denotes that all species become extinct for one year in one m2. This is equivalent to 10% of species disappearing in a 10 m^2 region for a year or 10% disappearing in a decade. The PDF is sometimes compared to the rate of extinction of a specific biome. Similar to PDF is global PDF but this indicator represent the extinction probability at global scale instead of the simple PDF that measures its effects at local scale (Verones et al. 2020). The metric to quantify global PDF is represented by Global Extinction Probability (GEP) which measures the value that a local extinction of a species reflects itself at global level (Kuipers et al. 2019).

1.2.2.2.2 Potentially Affected Fraction (PAF)

In ecotoxicological models, the potentially affected fraction (PAF) is a more often employed indicator. It describes the percentage of species that a drug affects. The PAF is frequently calculated using the no-observed-effect concentration (NOEC) as a threshold concentration (Klepper and van de Meent 1997). The PAF uses concentration-effect relationships to describe the harm done to species without reference to extinction. Ecotoxicity effects are calculated from concentration-response curves developed in the lab that relate to the percentage of the test group impacted. The impact could be related to many health conditions, such mortality or morbidity. The EC50 factor, which affects 50% of the population above background, or the LC50 factor, which kills 50% of the population, are typically utilised. These elements can be used to build models that describe how the entire ecosystem reacts to a particular stressor.

A mid-point PDF can be created from the PAF. The conversion is difficult, though, because the PAF relates to the percentage of species that are (somewhat) affected by a stressor whereas the PDF represents the loss of species owing to background changes. According to studies, the PAF_{EC50} to PDF conversion factor should be between 1 and 10. Impact 2002+ (Jolliet et al. 2003) offers a conversion factor of 2 (Aquatic PDF=Aquatic PAF/(2*h^w), with h^w the mean depth of freshwater expressed in m), whilst (Goedkoop & Spriensma, 2001) advocate a factor of 10 (PDF=PAF/10). The ReCiPe 2008 makes the assumption that PAF_{EC50} and PDF (Goedkoop et al. 2008) are equivalent, whereas LC-Impacts makes use of a factor of 2 for conversion (Verones et al. 2020) (PDF=PAF*2).

1.2.2.2.3 Ecosystem and biodiversity damage potential (EDP and BDP)

The Ecosystem Damage Potential (EDP) (Koellner 2003; Koellner and Scholz 2007) has a similar format to the PDF but measures damage in hectares rather than square meters. Based on the Corine Plus land use classification method, a specific EDP value was given to each type of land use. According to the UNEP-SETAC land use assessment framework (Koellner et al. 2013b) and based on the EDP indicator, (de Baan et al. 2013) created a new characterisation factor (CF) termed Biodiversity Damage Potential (BDP). By distinguishing the effects for nine important biomes, this factor assesses the consequences of land use on terrestrial ecosystems.

When compared to a reference semi-natural habitat, the derived BDP indicator assesses relative changes in species composition (relative species richness, or SR_{rel}). A specific region's land use type is compared to a reference condition using the characterisation factor $CF_{occLUij}$. The late habitat stage, which is frequently utilised in restoration ecology, served as the reference. To calculate the CF, the median SR_{rel} is removed from 1 (de Baan et al. 2013) (Equation 1.11):

Equation 1.11 CF of BDP formula

$$CF_{occ,LU,i,j} = 1 - SR_{rel,LU,i,j}$$

1.2.2.2.4 Functional diversity

Functional Diversity (FD) is an indicator considered from several academics (Díaz and Cabido 2001; Petchey and Gaston 2006; Flynn et al. 2009; Mouchet et al. 2010) more appropriated than metrics that used taxonomic or traits characteristics. The calculation of FD is based on a group of morphological, physiological or behavioural characteristics of organisms that are considered in relation with environmental pressure or ecosystem process effects (Maia de Souza et al. 2013). Hooper et al. (2002) state that ecosystem processes are more influenced from FD between species than taxonomic composition: in other words, the loss of one or more species can be compensated from other species with similar functional roles (Maia de Souza et al. 2013).

The metrics above exposed are only few examples of how metrics e methods exist to quantify biodiversity in LCA. Several works (Curran et al. 2011; Winter et al. 2017; Damiani et al. 2023) also reviewed these indicators to understand the methodology in deep, such as which taxonomy group they take into account or if a metrics is suitable to be implemented in LCA or not.

1.3 **Conclusions**

Ecosystem Services (ESs) have evolved over time, influenced by early reflections on the interactions between humans and nature. The growing awareness of their importance has led to their progressive recognition in decision-making processes, both at local and international levels, with the aim of integrating their assessment into economic and political models.

Several methodological approaches have attempted to define and categorize Ecosystem Services, contributing to their integration in decision-making processes and providing tools for their evaluation. Among these, a widely recognized classification divides ESs into four main categories: provisioning services, regulating services, cultural services, and supporting services. This model has helped clarify the role of ESs in different areas of human society, showing how they are interconnected with quality of life and environmental sustainability. However, this classification is not free from limitations, particularly regarding the distinction between ecological functions and direct benefits perceived by society. To address these limitations, new approaches have been developed, introducing a more detailed and operational vision for the analysis of ESs. Some argue that these newer approaches are mainly based on natural science terminology, limiting their applicability to social studies and policy-making.

The need to attribute an economic value to ESs has led to the adoption of specific tools to quantify their financial value, with the aim of facilitating their integration into economic and political processes. These tools allow the estimation of the value of ecosystems and the services they provide. However, the monetization of ESs has raised debates and criticisms, particularly regarding the difficulty of translating complex and interdependent ecological processes into financial terms.

Another key aspect concerns the application of ESs in sustainability models and environmental analysis tools, such as Life Cycle (LC) methodologies. The integration of ESs in these tools has highlighted the importance of considering environmental impacts throughout the life cycle of products, processes, and human activities. LCA has become an essential method for sustainability assessment, as it allows the analysis of natural resource consumption and environmental impacts at all stages of production and consumption.

However, several methodological challenges remain, including the difficulty of obtaining accurate data, the need to develop standardized models, and the complexity of effectively integrating ESs into environmental impact assessment processes.

Land management and land use change represent a further critical element in ES conservation, as the transformation of ecosystems can significantly alter the capacity of a territory to maintain its ecosystem functions. Land modification affects biodiversity, biogeochemical cycles, and the capacity of ecosystems to provide essential services, such as climate regulation and air quality. Measuring the effects of land use on ES is therefore crucial to understand the long-term environmental consequences and develop more effective management strategies. However, modelling the impacts of land use on ES remains a challenge.

Recent international environmental policies have recognized the key role of ESs in biodiversity conservation and climate change mitigation. These regulatory instruments aim to promote more effective protection measures, ensuring that ecosystem management is integrated into decision-making processes and sustainable development strategies. However, their practical application will depend on the ability to overcome economic and institutional barriers, as well as on the availability of more precise and reliable data to support sustainability policies.

In conclusion, the assessment and management of ESs are a central aspect for environmental and economic sustainability. The continued development of analysis methodologies and the greater integration of Ecosystem Services into public policies and economic models will be crucial to ensure the protection of biodiversity and the maintenance of ecosystem functions essential for human well-being. The main challenge remains to refine measurement tools, improve their applicability at a global level, and ensure that the value of ecosystems is fully recognized in future decision-making processes.

Abbreviations

ABC	Activity-Based Costing
CICES	Common International Classification of Ecosystem Services
EC50	Effective Concentration 50%
ECA	Equivalent Connected Area
EoL	End of Life
EPS 2015	Environmental Priority Strategy in product design 2015
ESs	Ecosystem Services
FU	Functional Unit
GEP	Global Extinction Probability

ILCD	International reference of Life Cycle Data system handbooks
ISO	International Standard Organization
LANCA	Land Use Indicator Value Calculation in Life Cycle Assessment
LC	Life Cycle
LC50	Lethal Concentration 50%
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MA	Millennium Ecosystem Assessment
NOEC	No Observed Effect Concentration
PAF	Potentially Affected Fraction
SAE	Society of Automotive Engineers
SAR	Species-Area Relationship
SETAC	Society of Environmental Toxicology and Chemistry
SHR	Species-Habitat Relationship
TCO	Total Cost of Ownership
TEEB	The Economics of Ecosystems and Biodiversity

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Chapter 2.
APPLICATIONS: STATE-OF-ART[1]

While Chapter 1 provided an in-depth review of ecosystem services (ESs) and their integration within Life Cycle Assessment (LCA), this chapter aims to explore their practical implementation. The focus is on case studies where ESs have been assessed using LCA tools, identifying common methodologies and challenges. In particular, this chapter highlights how ESs are considered within LCA frameworks and evaluates their effectiveness in capturing environmental trade-offs.

2.1 **Material and methods**

Given the interdisciplinary nature of this research, a well-defined selection process was necessary to ensure that the reviewed literature accurately reflects current advancements in this field. The following section outlines the criteria and procedures adopted for selecting relevant articles, including the construction of search queries, database selection, and screening methods used to refine the dataset for analysis. This procedure aims to identify the key characteristics of ecosystem services analysis applied within a Life Cycle Perspective. The subsequent sections detail the selection process, data extraction, and the obtained results, followed by a discussion of the findings.

To ensure a structured and comprehensive review, the following research questions were formulated:

1. How have ecosystem services (ESs) been assessed within Life Cycle Assessment (LCA) frameworks?
2. What methodological approaches have been adopted to integrate ESs into LCA, and what are their main advantages and limitations?
3. Which types of ESs are most frequently analysed in agricultural contexts through LCA methodologies?

[1] This chapter is based on:
1. the following scientific article: Soldati, C., De Luca, A.I., Iofrida, N. et al. Ecosystem services and biodiversity appraisals by means of life cycle tools: state-of-art in agri-food and forestry field. Agric & Food Secur 12, 33 (2023). https://doi.org/10.1186/s40066-023-00438-0. Personal contribution to the article: setting up the methodology, conduction of literature research, data analysis and writing of the first draft. These activities were conducted in collaboration with the other co-author
2. the following contribution in oral poster: Soldati, C., Falcone, G., Iofrida, N., Spada, E., Gulisano, G., De Luca, A. I. Ecosystem services through the lens of Life Cycle Methodologies: state-of-art of their application in the agriculture field. In Conference Procedeeings “La sostenibilità nel contesto del PNRR: il contributo della Life Cycle Assessment”. June 22-24 2022. Palermo. Personal contribution to the article: setting up the methodology, conduction of literature research, data analysis and writing of the first draft. These activities were conducted in collaboration with the other co-author

These research questions guided the literature selection process and the subsequent analysis, ensuring a thorough evaluation of how ESs are incorporated into LCA frameworks.

2.1.1 *Article selection and screening*

The Scopus and Web of Science (WoS) databases were used to search for relevant articles in June 2022. The syntax used for the relevant literature consisted of three parts: “ecosystem services” which are the subject of this review, the applied methodology, e.g., “life cycle assessment” and the field of application “agr*” which includes words like agriculture, agroforestry, agroecosystem, etc. Each part of the syntax is connected through Boolean operators, e.g., AND/OR. The complete query strings used for the research were the following:

- (TITLE-ABS-KEY (ecosystem AND services) AND TITLE-ABS-KEY (life AND cycle AND assessment) AND TITLE-ABS-KEY (agr*).
- (TITLE-ABS-KEY (ecosystem AND services) AND TITLE-ABS-KEY (life AND cycle AND costing) AND TITLE-ABS-KEY (agr*).
- (TITLE-ABS-KEY (ecosystem AND services) AND TITLE-ABS-KEY (social AND life AND cycle AND assessment) AND TITLE-ABS-KEY (agr*).

Alternative keywords such as ‘life cycle analysis’ and ‘ecological functions’ were excluded to maintain alignment with the terminology predominantly used in LCA-based studies. In addition, broader terms such as ‘environmental impact assessment’ were avoided to prevent retrieving studies unrelated to LCA frameworks. The chosen syntax is in line with previous systematic reviews in this field, ensuring methodological consistency and replicability. This approach ensures a focused dataset, facilitating a comprehensive analysis of how ESs are incorporated into LCA in agricultural contexts. While this selection may exclude studies using non-standard terminology (e.g., ‘Life Cycle Analysis’ instead of ‘Life Cycle Assessment’), it ensures that the retrieved literature aligns with established LCA methodologies and avoids terminological inconsistencies that could affect the comparability of results.

The search in the Scopus and WoS databases yielded 77 and 142 articles, respectively, for a total of 219 papers. The selection process followed the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) statement (Moher et al. 2009), a widely recognised guideline for systematic reviews. PRISMA provides a peer-accepted methodology that enhances the transparency, quality, and replicability of the review process. The PRISMA methodology was rigorously applied to ensure a structured and reproducible selection process. Screening phases were conducted in multiple iterations to refine the dataset and minimise potential biases. The selection criteria were carefully established to ensure methodological robustness, prioritising studies that explicitly integrate ecosystem services into Life Cycle Assessment frameworks. Although no pre-existing systematic review was directly followed as a model, the approach was structured to adhere to PRISMA guidelines and best practices for literature screening. Duplicate papers were excluded, resulting in 155 documents, which underwent a screening process. An initial selection was made using the “Refine Results” tool of the databases used to exclude reviews and editorial material and include only English-language articles. Thus, only indexed references to applied case studies were considered. A second screening was carried out through a thorough reading of the full text. Studies that did not directly focus on measuring ecosystem services through LC Methodologies were discarded. From the initial searches, 35 articles were found that adhered to the aim of the current review, so they were analysed in depth according to review parameters. The final screening produced a matrix with all information deemed relevant to answer the questions of this review. Figure 2.1 illustrates the complete selection of the literature search using the PRISMA model.

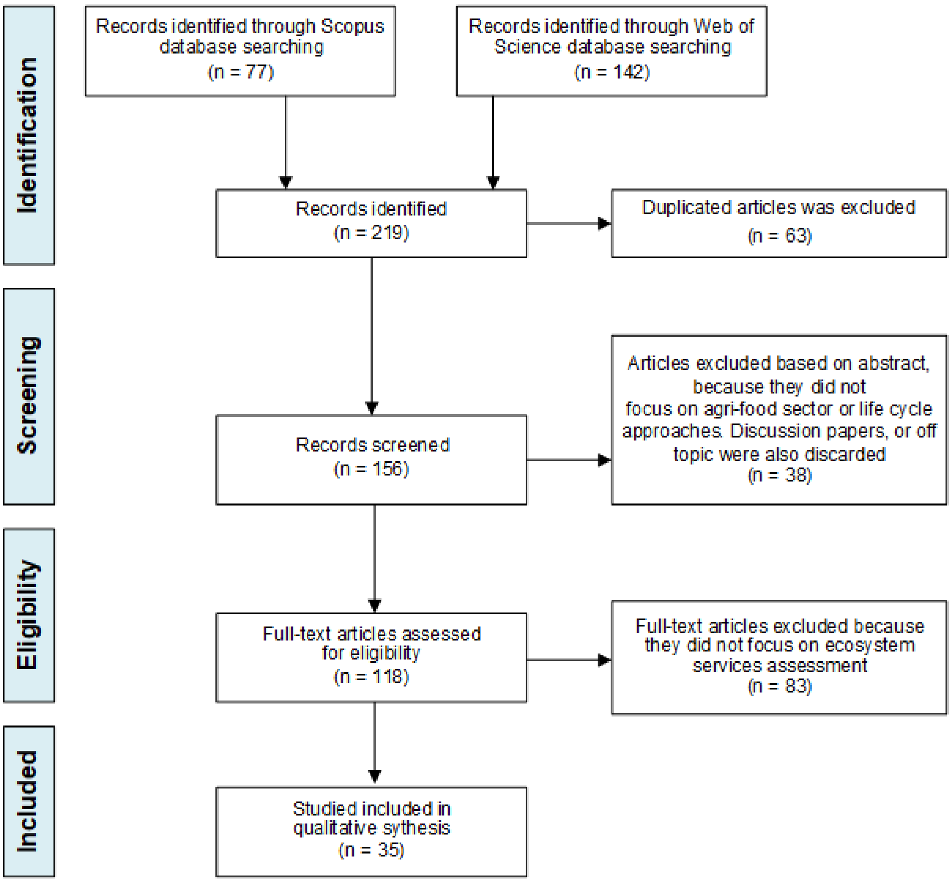

Figure 2.1 Methodological steps of the literature search process using PRISMA flow diagram (rework based on Moher et al. (2009)).

2.1.2 *Data extraction*

The matrix consists of four parts: the first concerns general information about the articles, i.e., authors, title, year of publication, journal, and country. The second part concerns ecosystem services and biodiversity and which methods and units were used to assess them. The third part focuses on methodological aspects of LCA, considering e.g., reasons for carrying out the study, the functional unit (FU), the scale of analysis, etc. There is also another section on the methodological aspects of LCC, covering the type of approach used, type of data and type of costs. Table 2.1 below provides a complete overview of all information sought in all documents.

Table 2.1 Matrix criteria for the critical review of the selected papers.

Section	Criteria
General information	Authors, Year, Title, Source, Place, Data duration gathering, Field of application, Main reference products
Ecosystem Services (ESs) information	Provisioning services, Regulating services, Cultural services, Supporting services and Biodiversity, Method and unit used for the analysis
LCA details	Reasons for carrying out the study, Functional units, System boundaries, Scale of analysis, Data source, Allocation procedures, Impact categories selected, Methodology of impact assessment
LCC details	Approach used, Type of cost, Data

The section "General information" considers all information related to the publication of a paper such as titles, authors, year of publication and journal. This section also includes information on the field of application and products discussed in the article.

Information on ESs provides information related to classification groups, ESs evaluated, and their units of measure. In this review, all ESs have been classified according to the MA classification because it is the most established, and some impact categories that will be discussed below are based on it. In relation to this classification, biodiversity assessment is considered separately because it cuts across all groups of ESs. Each ES column also considers the method and the unit of measurement of the ES group.

In the section on LCA details, all methodological aspects are discussed. The reasons for carrying out the study are grouped into similar objectives. The functional unit was classified into four groups, based on the object of the FU. Thus, FUs have been classified into "mass-related," "land-related," "energy-related," and "economic value-related". Similarly, system boundaries have been classified into three groups: "cradle to gate," "cradle to grave," and "gate to gate". The "well to tank" system boundary considered in some papers, was interpreted as a "cradle to gate" system boundary.

Regarding the scale of analysis, it concerns the spatial scale on which the study was conducted. Four levels were detected: local level (for studies that involve a farm/company or a small group of them), regional level (for studies that involve a part of a country or a large group of farms or companies), national level (for studies that involve a country) and continental level (for studies that involve a whole continent).

Also, the approaches used in the LCA applications were analysed. A typical LCA study can be performed following two main approaches: attributional and consequential. Attributional LCA (ALCA) represent the potential environmental impacts that can be retrieved from a system product throughout its life cycle. To model LCA following an attributional approach, average or generic data can be used when the system product comes from several producers or technologies. Consequential LCA (CLCA) studies how a decision in foreground processes influence the other process or the economy. To model LCA following a consequential approach, marginal data can be used that allow evaluation of the consequences of a decision or a series of decisions (Joint Research Centre and Institute for Environment and Sustainability 2010).

Data sources are classified according to their gathering modality: if data are collected for a specific site using interviews, questionnaires, or direct measurements, they are classified as primary data. Secondary data have been classified according to three subcategories: secondary data from databases (e.g., Ecoinvent), secondary data from the literature (e.g., from previous scientific studies in the same field), and secondary data from other bibliography sources (grey literature such as reports, statistics, theses, non-indexed journals, etc.). The remaining types of data, e.g., calculated data, are classified as tertiary data.

For the allocation procedures, some specific criteria were identified and classified, but not the approach applied. The term “criteria” refers to how the allocation was made, e.g., through physical criteria (e.g., mass or energy) or economic criteria. The term “approach” refers to how the environmental load is allocated in the study, e.g., the cut-off approach (100:0 approach). Moreover, for the purpose of this review, a specific selection of terms and definitions was proposed (Table 2.2), to describe ESs, and inspect their use and life cycle inclusion in the articles examined.

Table 2.2 Definition used in this review.

Term	Definition	Reference
Biotic Production Potential (BPP)	BPP is the condition of the land to support biomass production in the short, medium and long term. Given the definition, BPP is considered a supporting service in this revision. In the LCA, BPP is a mid-point category, and the unit of measurement is kg C · yr FU^{-1}	Brandão and Milà i Canals (2013)

Climate Regulation Potential (CRP)	CRP is the lack of carbon sequestration due to land use, i.e., non-stored carbon, compared to a reference land use. Given the definition, CRP is considered a regulating service in this revision. In the LCA, CRP is a mid-point category, and the unit of measurement is kg C transferred to air FU^{-1}	Müller-Wenk and Brandão (2010)
Erosion Regulation Potential (ERP)	ERP is the capacity of a terrestrial ecosystem to resist soil loss through erosion. Given the definition, ERP is considered a regulating service in this revision. In the LCA, ERP is a mid-point category, and the unit of measurement is kg of soil potentially eroded FU^{-1}	Saad et al. (2013)
Water Purification Potential related to physicochemical filtration (WPP-PCF)	WPP-PCF is the ability of soil to act as an absorption matrix and adsorb dissolved substances. Given the definition, WPP-PCF is considered a regulating service in this revision. In the LCA, WPP-PCF is a mid-point category, and the unit of measurement is centimoles of cation fixed per kilogram of soil per kg of soil ($cmol_c$ kg_{soil}^{-1} FU^{-1})	Saad et al. (2013)
Water Purification Potential related to mechanical filtration (WPP-MF)	WPP-MF is the ability of soil to mechanically clarify a suspension through soil infiltration and provide a cleaning action to ensure groundwater protection. Given the definition, WPP-MF is considered a regulating service in this revision. In the LCA, WPP-MF is a mid-point category, and the unit of measurement is centimetre per day (cm day^{-1} FU^{-1}).	Saad et al. (2013)
Net primary production (NPP)	NPP is the quantity of carbon assimilated through photosynthesis by vegetation in a certain period. In LCA, NPP can be both a mid-point category and an end-point category for ecosystem	Taelman et al. (2016)

	quality. The unit of measure of NPP is MJ_{se} and when it is used as an end-point category, it can be converted into the most common unit of ecosystem quality, i.e., Potentially Disappeared Fraction (PDF).	
Human appropriation net primary production (HANPP)	HANPP is an indicator that measures the difference between NPP of potential vegetation without any human intervention and NPP remaining due to human intervention. In LCA, it can be considered a mid-point category.	Haberl et al. (2007) and Mattila et al. (2011)
Emergy	Emergy is the quantity of direct and indirect solar energy used for delivering a product or a service. The common unit of measure is Solar Equivalent Joule (J_{se}). In LCA, it can be a proxy to consider and measure ESs for example for the measuring of soil erosion (Núñez et al. 2013).	Perrotti (2020)
Land use	Land use is the change in the use or management of land by humans, which can lead to a change in land cover. It is part of the driver “habitat change,” one of the five drivers of ESs and biodiversity loss. In the LCA, land use is a mid-point category, and the units of measure are $m^2 \cdot yr$ or $PDF \cdot m^2 \cdot yr$.	MA (2005) and Mattila et al. (2011)
Land occupation	Land occupation is the continuous use of an area for a certain purpose controlled by humans, e.g., agriculture, forestry or construction. It is part of the driver “habitat change,” one of the five drivers of ESs and biodiversity loss. In the LCA, land occupation is a mid-point category, and the unit of measure is $m^2 \cdot yr$.	MA (2005) and Mattila et al. (2011)
Climate change	Climate change is the change in climate induced by increasing temperature and CO_2 concentration. It is one of the five factors driving	MA (2005)

	the loss of ESs and biodiversity. In LCA this can be both a mid-point category and an end-point category.	
Overexploitation	Overexploitation is the exploitation of natural resources and wildlife for human activities. It is one of the five drivers of ESs and biodiversity loss. In LCA does not exist counterpart in the mid-category, so this should be a new category for impact for ESs and biodiversity assessment.	MA (2005)
Exotic species	Exotic species are indigenous species that can disturb the ecological functions of a natural ecosystem. It is one of the five factors that determine the loss of ESs and biodiversity. In LCA does not exist counterpart in the mid-category, so this should be a new category for impact for ESs and biodiversity assessment.	MA (2005)
Pollution	Pollution is a change in the composition of soil, atmosphere and water caused by chemicals. It is one of the five drivers of ESs and biodiversity loss. This driver is represented by several categories, e.g., acidification or eutrophication. However other impact categories can be added, e.g., emission of noise or emission of light	MA (2005)

2.1.3 *General information*

As shown in Figure 2.2a, countries are divided into groups by macro-areas. The category “Others” includes comparisons between different countries or missing information. Most of the studies took place in Europe, followed by North America. According to D’Amato et al. (2020) the reason why Europe and North America have, in general, the largest number of case studies, is related to the history of LCA (Bjørn et al. 2018). Moreover, as it will see later, most of the methods used for analysis in papers were developed in Europe and North America.

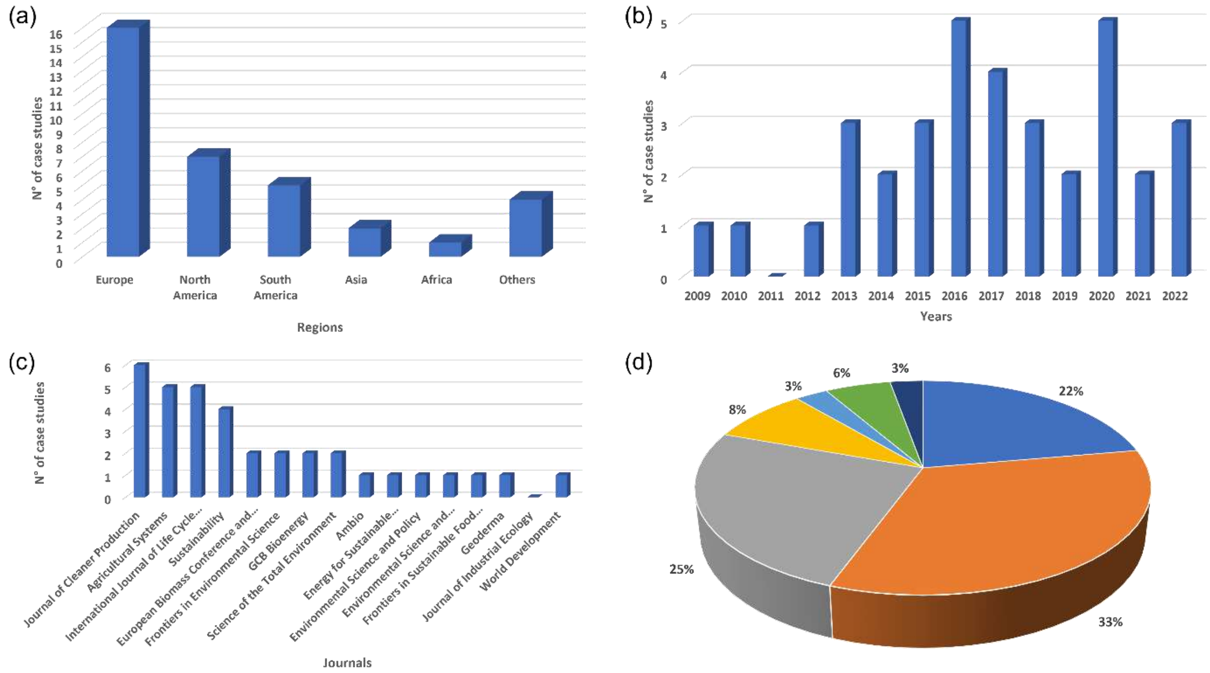

Figure 2.2 Graphs related to country (a), years of publication (b) journal of publication (c) and field of application (d). In graph (d): livestock (blue), agriculture (orange), bioenergy/biofuels (grey), agroforestry (yellow), orchard (light blue), food product (green), biomass production (dark blue) and aquaculture (brown).

Interest in LCA of ESs increased in 2013 and subsequent years as shown in Figure 2.2b. This may be related to the publication of UNEP-SETAC guidelines (Koellner et al. 2013) that create five new categories to assess ESs and two new categories to assess biodiversity. Indeed, several papers have applied these categories, and Lathuillière et al. (2017) assessed the robustness of these guidelines. The results have made it possible to highlight some criticism in the area of study due to ESs problems, e.g., climate sequestration and soil mechanical filtration. To explain the reason for the critical issues, it is necessary to have knowledge of biophysics processes and the uncertainty due to the regionalisation of characterisation factors.

Most of the articles are published in the Journal of Cleaner Production with eight case studies. This may be related to its multidisciplinary nature, as the topics cover environmental science, economics, engineering, and energy. The same number of articles were published in the International Journal of Life Cycle Assessment and Agricultural Systems (Figure 2.2c).

According to the search syntax, most of the papers concern agriculture, with thirteen articles (Figure 2.2d). Other important fields of application are Bioenergy/Biofuels with nine papers, and livestock with eight papers. Sectors less explored are agroforestry with three papers, food production with two papers, and orchard and biomass production with one paper each. Only two papers explored two fields jointly: agriculture and livestock (Glendining et al. 2009) and agriculture and agroforestry (Brandão et al. 2010).

2.2 **Results**

2.2.1 *Ecosystem services information*

In this section, the information on ESs, as well as how they are considered, was interpreted, and the respective unit of measure considered in the articles was discussed. As mentioned above, this information was classified according to the MA classification. In addition, biodiversity was analysed as an independent category because it cuts across all categories.

2.2.1.1 Provisioning services

Since each article reviewed focuses on food or crops for fuel/energy production, these have been considered as provisioning services. To take these services into account, if the method and unit of measurement were not expressed, they were considered within the LCA methodology itself and the unit of measurement was considered to be the functional unit of the LCA. Most of the articles (26) analyse food supply, followed by biomass production for energy (7), biomass for fuel production (5) and water (4). The category "Other" considers the supply of other materials, e.g., lithological material (Baral et al. 2012) or genetic resources (Styles et al. 2015) (Figure 2.3).

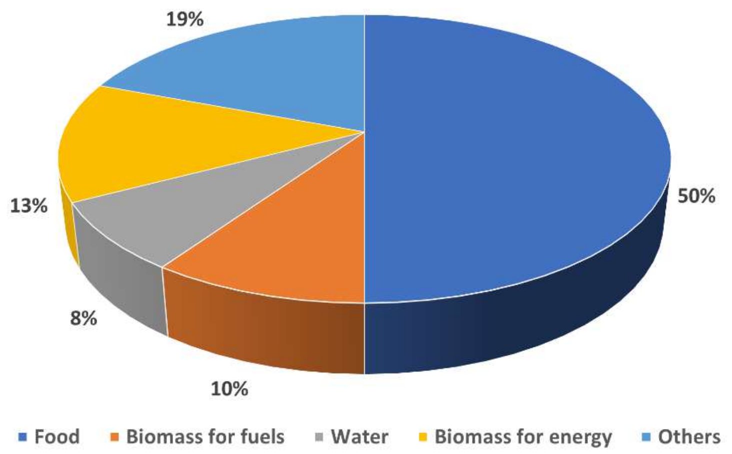

Figure 2.3 Provisioning ESs in reviewed papers.

2.2.1.2 Regulating services

Regulatory services are all those services that allow part of ecosystems to be regulated. This category includes, for example, pollination or erosion regulation. Regulating services consider many categories, as can be seen from the Figure 2.4. The category "Others" considers regulating services that are not considered more than once, such as Hazard regulation (Styles et al. 2015) and Nitrogen mineralisation (Baral et al. 2012). The most analysed regulating services are climate regulation (10), followed by erosion (9) where both wind-caused and water-caused erosion are jointly considered, carbon sequestration (8) and water purification (6) where both mechanical and physical–chemical purification are jointly considered. The category "N/A" (8) means that in this case, the paper does not analyse any regulating services (Figure 2.4). For regulating services that are taken from UNEP-SETAC Guidelines (Koellner et al. 2013) the unit of measure is the same as Table 2.2, e.g., CRP or ERP. Other units of measure are MJSE with emergy modelling (Núñez et al. 2013) or t ha^{-1} y^{-1} with RUSLE equation (Cecchin et al. 2021) for the measuring of soil erosion or t ha^{-1} for soil carbon sequestration through SOC quantification (Nguyen et al. 2022).

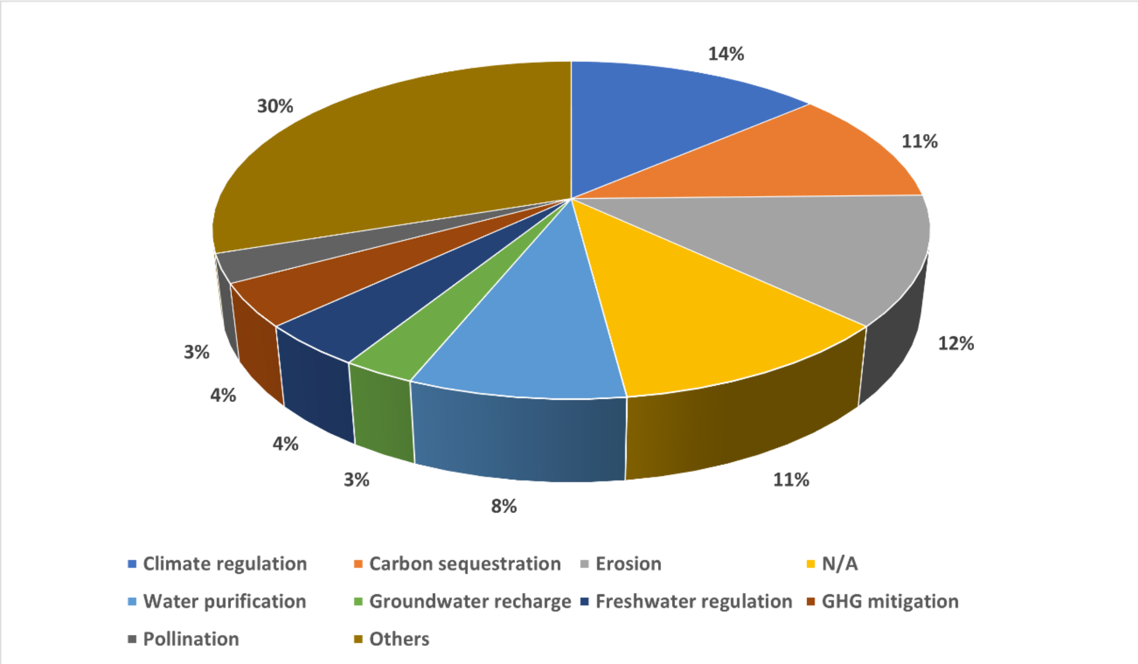

Figure 2.4 Regulating ESs in reviewed papers.

2.2.1.3 Cultural services

Cultural services are the least treated category for the evaluation of ESs. Few articles consider them, and the methods for calculating them are too weak compared to methods for calculating other services. Of the five articles that consider this type of ESs, the most reliable is probably the application of an economic value because cultural services are based on the intrinsic value that people attach to, for example, a landscape (aesthetic, social or cultural). Furthermore, Zhang et al. (2010) assert that economic valuation "is the most appropriate way to account for cultural services since these services are truly anthropocentric in nature".

2.2.1.4 Supporting services

Supporting services are a particular category because they are the only category that is in a double relationship with the others. Due to this peculiarity, this category is not often considered in papers.

Supporting services are services that guarantee the functionalities of the other services, e.g., without good soil quality, it would be impossible to guarantee food, fibres, nitrogen cycle, etc. Other services of this category can reflect the propriety of a system, e.g., soil quality does not consider other proprieties of the supporting system except for the quality. Other services in this category are biotic production (6), whose definition is reported in Table 2.2, primary production (1), soil conservation (1) and soil quality (3) (Figure 2.5). For supporting services that are taken from UNEP-SETAC Guidelines (Koellner et al. 2013), e.g., BPP, the units of measure are the same as in Table 2.2. Other units of measure are, for example, kg C per year using the LANCA method (Jeswani et al. 2018) or t C ha^{-1} y^{-1} for measuring soil quality using SOC quantification (Cecchin et al. 2021).

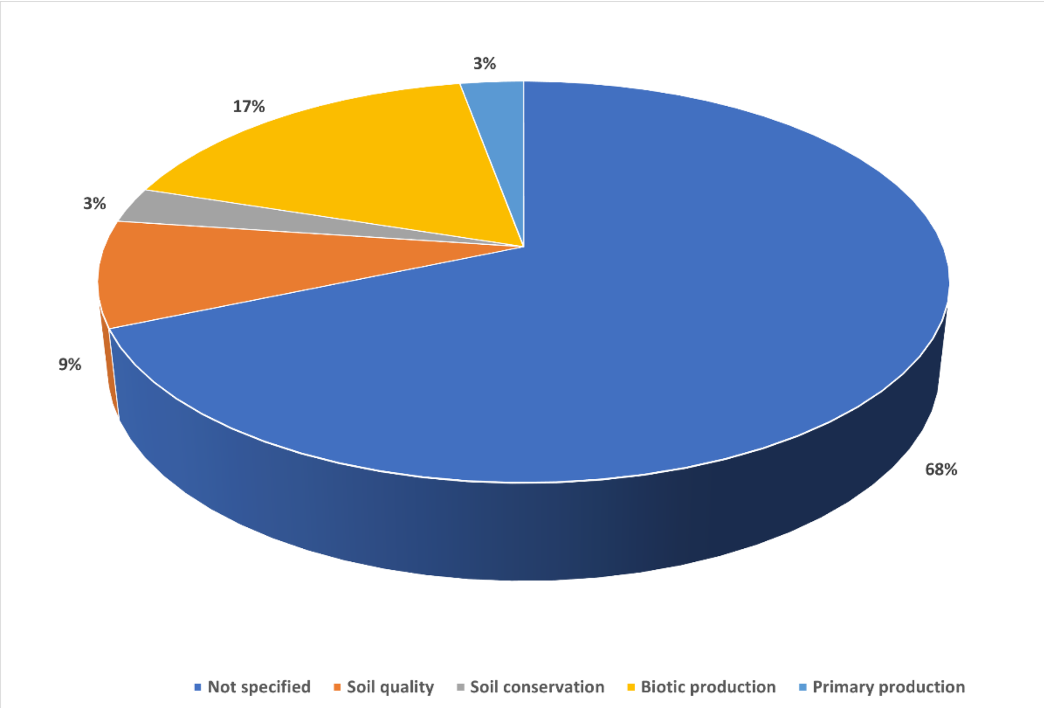

Figure 2.5 Supporting ESs in reviewed papers.

2.2.1.5 Biodiversity

Biodiversity is a special category for ESs assessment because it can influence all ESs categories. Almost 50% (17) of papers analysed biodiversity with different methods: a paper (Dick et al. 2022)used a ranking system to classify, for example biodiversity conservation, while other papers used Biodiversity Damage Potential (BDP) as proposed in UNEP-SETAC guidelines (Koellner et al. 2013). One paper (Jeswani et al. 2018) used the LANCA model (Bos et al. 2016) and in this case, it specified the type of relationship to calculated biodiversity through countryside Species – Area Relationship (SAR).

2.2.2 *LCA details*

2.2.2.1 Reasons for carrying out the study, functional unit, system boundaries and scale of analysis

All the reasons for carrying out the study were classified into the following six categories, grouping them for the same objective (Figure 2.6a):

- comparison between two or more products/supply chains, which concerns 13 papers and is the second most common category in this review.
- trade-off assessment, which concerns only one paper.
- environmental, economic and/or social assessment, which concerns 20 papers, and is the most common category in this review.
- comparison of different management practices, which concerns 4 papers.
- develop and/or test a methodology, which concerns 5 papers.
- identify environmental, economic and/or social hotspots, which contain 3 papers.

As already mentioned, FUs were classified into four categories, as shown in Figure 2.6b:

- mass-based: this first category is the most used FU in papers reviewed with 18 articles. In general, environmental impacts are related to 1 kg of product or 1 tonne of production. However, there are several cases which use several mass-based FUs, i.e., the annual production or human food intak.
- land based: this category is the second more use in papers reviewed used by 16 papers. This type of FU is the second more use of FUs in this review.
- energy based: this type of FU is used by 5 papers and is used when the focus of the paper is on biomass production for energy or fuels.
- economic value based: this type of FU is used in only one case and for a comparison to a land-based FU.

According to Kim and Dale (Kim and Dale 2006), LCA outcomes can have a strong influence on the final results, generating very different or conflicting upshots for what concerning impact assessment, and, consequently, also for impacts on ESs. Seda et al. (2010) suggest working with multiple FUs to ensure a complete overview of a product system analysed from different points of view. In this review, only six papers operate using double FUs. As reported in "Material and methods" section, the system boundaries were classified into three categories (Figure 2.6c). The most commonly used system boundaries are cradle-to-gate (27 papers), followed by gate-to-gate (7) and cradle-to-grave (2). In cradle-to-gate, both well-to-tank system boundaries and papers that consider the distribution of a product are considered. The gate-to-gate system boundaries focus on a single phase: in this review, the phase analysed is the agricultural phase because of the initial research setting and because it is assumed that the agricultural phase is the one that can generate the greatest impacts for ESs and biodiversity.

The scale of analysis was evaluated considering the spatial scale of the evaluation. In this review, scales of analysis were classified into four categories (Figure 2.6d):

- the local level is where the focus of assessment is a specific farm or a company. This assessment is quite common in literature, with 14 papers that used this scale.
- the regional level is where the focus of assessment is an area of a country or a group of farms or companies in a different part. This assessment is applied to 9 papers.
- the national level is where the focus of assessment is the whole country, i.e., the whole United Kingdom. This assessment is performed by 12 papers and is the second most used scale of analysis.
- the continental level is where the focus of assessment is a continent. There is only a single paper in which the assessment was performed, considering the whole of Europe.

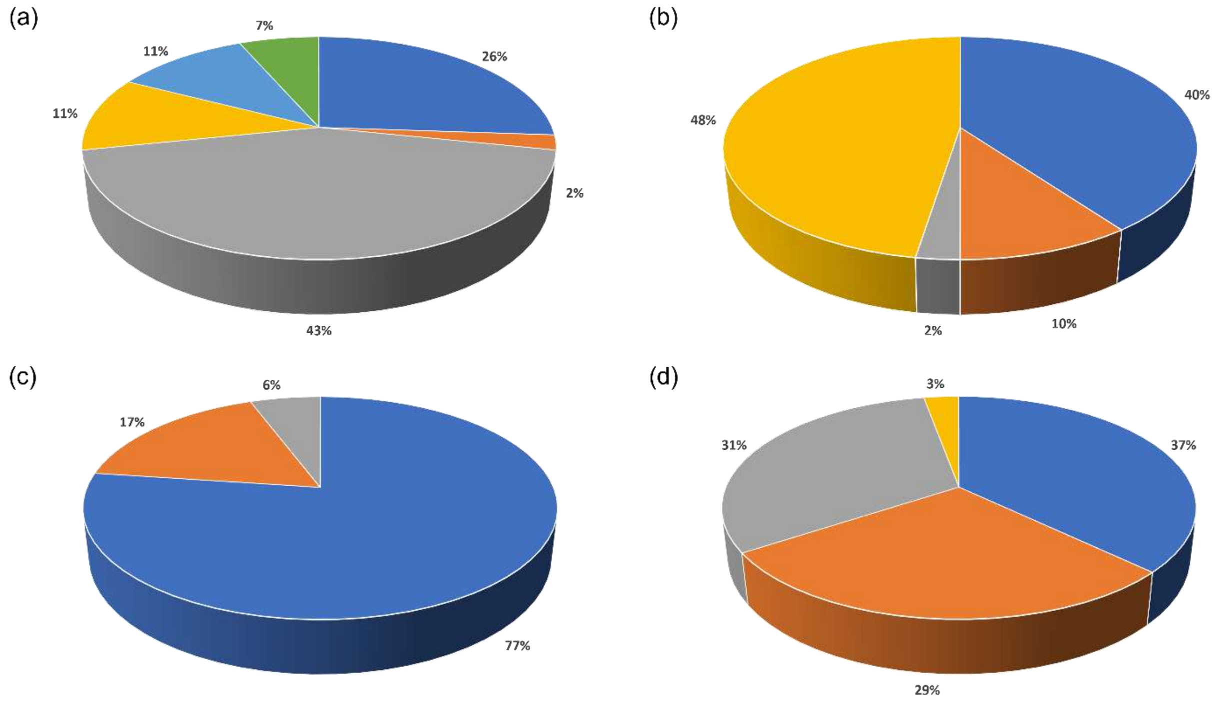

Figure 2.6 Graphs related to goals (a) FUs (b), system boundaries (c) and scale of analysis (d). In graph (a): comparison of two or more products/supply chains (blue), assessment of trade-offs (orange), environmental, economic, and social assessment (grey), comparison of different management practices (yellow), development and/or testing a methodology (light blue) and identification of environmental, economic and/or social hotspots (green). In graph (b): mass-based FU (yellow), land-based FU (blue), energy-based FU (orange) and economic value-based (grey). In graph (c): cradle-to-gate (blue), cradle-to-grave (grey) and gate-to-gate (orange). In graph (d): local level (blue), regional level (orange), national level (grey and continental level (yellow).

2.2.2.2 Data source and allocation procedures

The most frequently used approach is ALCA, applied in 34 papers. Two papers (Styles et al. 2015, 2016) applied both ALCA and CLCA approaches to assess the difference between their applications. This approach is not widely applied: it requires the estimation of consequences through market data or economic models, and this can produce high uncertainty. Styles et al. (2015) used both ALCA and CLCA to assess eight possible bioenergy scenarios. The results showed how to manage a scenario to pursue a purpose, e.g., maintain the production for food crops instead of energetic crops, while also considering ESs from a qualitative point of view, i.e., an increasing or decreasing trend.

Styles et al. (2016) analysed three scenarios for assessing three ESs. CLCA allows capturing some negative effects related to the expansion or intensification of agricultural production to compensate for the food loss.

Most of the studies are considered secondary data, and as mentioned before, they are classified according to their source. Secondary data from the literature (34) is more widely used than secondary data from databases (22). Some papers use secondary data from other literature sources, i.e., statistics or almanacks (grey literature). Fifteen of the reviewed studies analysed primary data obtained from interviews or questionnaires. The last category considered in this review is tertiary data: in this category were grouped data obtained from experts' judgement, data obtained from simulations, and data obtained from calculations and assumptions (Figure 2.7a).

A multifunctionality problem, and thus of allocation, occurs when a system produces two or more products or services. In this review, no paper has been found that has applied system boundaries expansion. The economic allocation is the most applied method with nine papers, followed by physical allocation (e.g., mass and energy criteria) with seven papers and biophysical allocation with one paper. There are also four papers where multifunctionality does not need to be applied, 19 papers where it is not specified which application of allocation is performed, nor whether it is avoided and why, and two papers that consider the allocation approach, allocating a single product (Figure 2.7b).

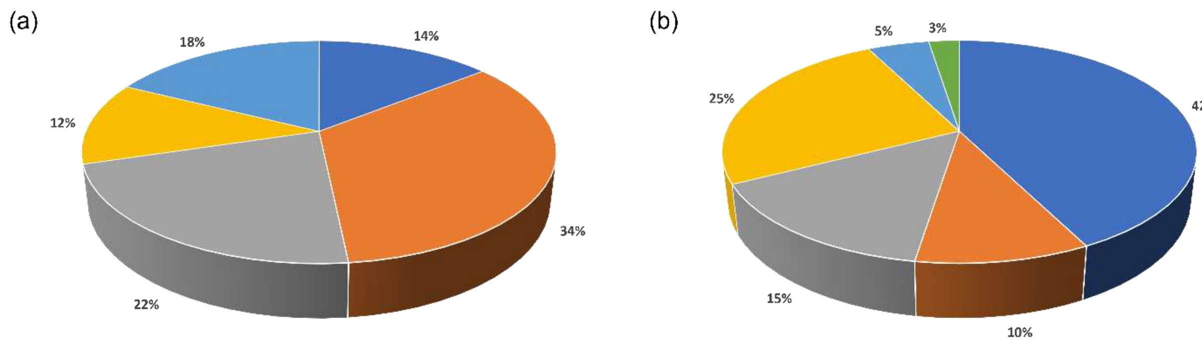

Figure 2.7 Graphs related to data type (a) and allocation performed (b). In graph (a): primary data (blue), secondary data from the literature (orange), secondary data from databases (grey), secondary data

from other literature sources (yellow) and tertiary data (light blue). For graph (b): allocation not specified (blue), no allocation performed (orange), physical allocation (grey), economical allocation (yellow), allocation to a single product (light blue) and biophysical allocation (green).

2.2.2.3 Impact categories selected and methodology of impact assessment

The most investigated category was found to be Global Warming Potential (GWP) which represents certainly the most important topic in LCA of agriculture systems because the impacts in these systems can lead to up to 24% of Greenhouse gases (GHG).

End-point categories are not widely assessed, with only three papers considering them. Lathuillière et al. (2017) considered a generic endpoint with an economic value to consider the effect of mid-point analysis. Golkowska et al. (2016) used the "classical" end-point categories present in the ReCiPe method: Natural Resources, Ecosystem Quality and Human Health. Núñez et al. (2013) analysed the end-point impact way for soil erosion, considering human health and ecosystem quality. Finally, it can be found that only one paper considers all ESs, and this supports the idea that a new area of protection related to ESs analysis should be developed in the future to include the potential damages. There is a diversity of impact assessment methods used to assess ESs in LCA. Two methods are more widely used than others, e.g., Recipe with six papers and CML with seven papers. Three papers apply the TRACI method and consider Recipe and CML as well, sixteen papers used a method developed in Europe or North America. As previously mentioned, this can be traced back to the origin of LCA, which started to develop in Europe and North America. There is also an application for a single ecosystem service, i.e., RUSLE for erosion or IPCC for climate mitigation. UNEP-SETAC was applied by five papers, but this method is innovative for the creation of seven specific categories for ESs and biodiversity assessment. Eight papers did not specify the method used while twelve papers used other methods.

The software to perform an LCA can be very different: nineteen papers used a software LCA (e.g., Simapro, GaBi, OpenLCA, etc.), fourteen papers did not specify the software, and eleven papers used other types of software. Furthermore, the most widely used LCA software is Simapro, with ten papers having used it.

2.2.3 *LCC details*

Life Cycle Costing (LCC) was applied only in two papers. Brandão et al. (2010) used a conventional LCC to explore complementary fields to the environmental aspect. The system boundary covered is cradle to gate and includes all costs supported by the land manager related to the specific cultures (wheat, oilseed rape, Scots Pine, willow and Miscanthus) for each land use (production of food, energy, or timber). Consequently, costs related to end-of-life and use costs are not considered because they are supported by non-land managers and are outside system boundaries. The assessment of LCC was performed in parallel to LCA starting from LCA's steps and considering the corruptive steps of LCC. The data type was secondary data, mainly from literature, considering the equivalent economic data from the environmental data of LCA.

Fan et al. (2020) did not specify the type of LCC, but they used this methodology to assess land use capitalisation resources. The costs considered in the study are related to materials, energy, labour and mechanical equipment during the whole life cycle. They considered primary data, obtained through field research and interviews, for the economic value of materials, energy, labour, mechanicals and fee and tertiary data to establish the economic value of ESs through the calculation of their economic values. Formulas to obtain the values of ESs were obtained from different papers and allow the calculation of different ESs values, e.g., the food production value or the value of biodiversity, but they also allow the calculation of negative services, e.g., economic loss due to cadmium pollution or pesticide pollution.

2.3 **Discussion**

2.3.1 *Main findings from articles reviewed*

The overall objective of all the articles examined was the integration of ESs into the LCA. As shown in the results section, this integration is very different article-to-article, because it changes depending on the purpose of the study and the method of evaluation of ESs.

The use of LCA and LCC has only been applied in two papers (Brandão et al. 2010; Fan et al. 2020) with different methods and, consequently, different results. The first articles compared different land uses, different crops, and different land management to assess ESs and biodiversity impacts and thus establish which have the greatest benefits and impacts. The second article analysed "land tickets" in China, i.e., a system to incentive farmers to recover abandoned lands by agriculture and their connected, by also considering ESs, as for example, climate regulation for regulating services and soil conservation for supporting services, and some dis-services, like pollution by pesticides and pollution by fertilisers and found that the positive benefits from ESs in this system are greater than the negative impacts. Different methodologies were applied in some papers, as described below. Baral et al. (2012) used a hybrid Eco-LCA model that combines the Input–Output life-cycle economic inventory with the process-based inventory to evaluate different energy raw materials while also considering thermodynamic indicators. Glendining et al. (2009) combined the LCA with the economic Total Factor Productivity (TFP) index using land-use assessment for the ESs analysis to estimate the optimal level of all inputs to reduce pollution with a minimum number of resources and maintain agricultural income at the highest possible level. Núñez et al. (2013) developed their specific methodology to study the regulatory service “soil erosion,” and the analysed end-point category has considered natural resources and ecosystem quality by emphasising the importance of a regionalised assessment for soil erosion due to its variability. Liu et al. (2020) observed rice cultivation in the United States, China and India through the application of the cascade model developed by Rugani et al. (2019), analysing four ESs, namely water supply, carbon sequestration and air/water quality regulation and identifying some critical issues in integrating ES-LCA framework, e.g., the generic nature of this methodology to analyse other products and sectors in which human activity have an important contribution for the delivering of ESs or the lacking of data or the complexity to run this method. Finally, Wang et al. (2022) used an energy-based LCA framework to assess the ecosystem services and disservices of six crops.

The methodology of the UNEP-SETAC guidelines (Koellner et al. 2013) has been applied by Milà i Canals et al. (2013), Muñoz et al. (2014), Helin et al. (2014), Piastrellini et al. (2015) and Lathuillière et al. (2017), in different fields and products: in agriculture for the evaluation of margarine and soya, in bioenergy sector for the comparison between bio-based and fossil-based ethanol production, and in forestry through the assessment of the energy produced from wood, agro-biomass and peat.

All outcomes have found that the main critical issue concerns land occupation, which results in being the main driver of impacts instead of land transformation, and the importance of the development of characterisation factors specifying also soil management (e.g., conventional vs. biological).

The assessment of ESs can also be carried out on a continental scale, as demonstrated by Jeswani et al. (2018) who applied the LANCA model (Bos et al. 2016).

A further possibility for the study of ESs and biodiversity is the coupling of ALCA and CLCA, as carried out by Styles et al. (2015, 2016) in the field of bioenergy, the assessment can be carried out in both quantitative and qualitative terms, considering, for example, an increase or decrease in ESs compared to a baseline.

The application of allocation methods for the comparison of different land use management practices does not seem to influence the results of impacts if the assessment is carried out on a local scale and under similar conditions, as highlighted by Salvador et al. (2016). However, when ESs are not considered in the analysis and they need to be allocated, GHG emissions decrease by increasing the intensification from a pasture-based system (lowest intensity), to a zero-grazing system (highest intensity) otherwise, GHG emissions decrease by decreasing the intensification, from a zero-grazing system to a pasture-based system, as reported by Ripoll-Bosch et al. (2013).

A similar outcome can be found in Bragaglio et al. (2020). They assessed four livestock systems (“traditional” Podolian system, Specialized extensive system, Cow-calf intensive system and

Fattening system) at the beginning without considering ESs, e.g., the co-production of milk as a provisioning service or the presence of a festival as a cultural service, and after considering them.

When the assessment field is bioenergy, the most studied ESs are those related to carbon sequestration and soil organic carbon, as reported, for example, by Tichenor et al. (2017), Baumert et al. (2018) and Nguyen et al. (2022). However, these are not the only cases where the study focuses on GHG emissions. Agriculture is another sector where the assessment of GHG emissions is on the rise because, as mentioned above, this sector produces up to 24% of GHGs, and for example, Fiore et al. (2018), Bais-Moleman et al. (2019), Martinelli et al. (2019), Bessou et al. (2020) and Rowntree et al. (2020).

Some works used comparisons between different land management practices to understand how outcomes are affected, such as Jarchow et al. (2015) and Berti et al. (2016, 2017). Similarly, other studies consider different crops or scenarios for agricultural and livestock systems to observe how the results will change, as in Marton et al. (2016), Golkowska et al. (2016), Hessle et al. (2017), Cecchin et al. (2021) and Dick et al. (2022).

To conclude, Souza et al. (2021) evaluated the production of electricity from sugarcane biomass, making it possible to identify the most promising scenario concerning, for example, bioenergy production or water recycling, with, moreover, spatial planning for energy production, not limiting the analysis of impacts to LCA results alone, but expanding this assessment to ESs.

2.3.2 *Analysis of the use of ESs' classification*

As mentioned at the beginning of this document, several approaches are available for ESs classification, but the most important are Millennium Ecosystem Assessment (MA), The Economics of Ecosystems and Biodiversity (TEEB) and Common International Classification of Ecosystem Services (CICES). Despite their widespread use in the assessment of ESs, as already mentioned, they are not widely used in LCA. The MA classification was used as the basis for the creation of the UNEP-SETAC guidelines with seven new categories in LCA and it is used in some documents (Milà i Canals et al. (2013), Muñoz et al. (2014), Helin et al. (2014), Piastrellini et al. (2015), Lathuillière et al. (2017)). The TEEB classification is not used in any documents retrieved from this review, and the CICES classification is used in only one document (Liu et al. 2020). This article explicitly reported the type of classification instead of the articles that adopted the MA classification. Furthermore, it is easy to establish a single ES due to the hierarchy structure of the CICES classification.

The use of classification can be helpful to understand the areas where actions are needed to improve the ESs group. It can also be applied to understand which area of investigation should be taken into account for future research: cultural services, for example, are important services for human cultural heritage but are not much included in the LCA.

2.3.3 *Methodological aspects*

The methodological aspects used to analyse ESs are similar to those of other LCA studies: e.g., the FU is based on mass, land, energy or economical value. These types of FUs focus on a product or a group of products and not on ecosystem services. A single article (Martinelli et al. 2019) used a particular land-based FU, namely one hectare of an agroforestry system which also includes ecosystem functions.

Data availability is another important aspect to consider when analysing ESs. Europe and North America are two macro-areas with a substantial amount of data from databases, statistics, scientific literature, and other sources. In other areas, this can be very difficult. As an example, Lathuillière et al. (2017) and Dick et al. (2022) point out the lack of data in Brazil to differentiate the different factors influencing biodiversity and ecosystem services. In addition to the above, it is important to use spatial differentiation of information to ensure a more accurate result. A solution can be found in the use of ecoregions, which are “relatively large units of land containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land-use change” (Olson et al. 2001, 933).

Managing multifunctionality can be a real problem when it is necessary to allocate flows to each output, including ESs; because LCA can be applied to products and services, ESs are, thus, an outcome that must be allocated for its production. This can completely change the results as reported by Ripoll-Bosch et al. (2013) and Bragaglio et al. (2020) in which the impacts of the systems analysed can change from the choice of an industrial system to a local or extensive system. Another example of multifunctionality management is given using economic allocation to distribute environmental impacts to honey and pollination service (Arzoumanidis et al. 2021). Regarding environmental categories, some mid-point categories can assess some ESs, e.g., BPP, CRP, etc. Areas of protection (AoP) cover only human health, ecosystem quality and resource, but there is also the possibility to develop other AoP represented by the different typologies of ESs.

2.3.4 *Drivers or ESs?*

It is important to differentiate ESs and biodiversity from the drivers that affect them. Drivers produce negative consequences, e.g., reduction or loss in biodiversity and ESs. According to MA, these causes are the following: Habitat change (due to land use and land occupation); Pollution; Overexploitation; Exotic species; Climate change. Measuring one of them, it is evaluated the contribution of a driver to the loss of biodiversity and ESs. The measurement of ESs and biodiversity should be conducted using specific categories for what concerns the first topic and using different indicators, e.g., the functional diversity index (Maia de Souza et al. 2013) or types of relationships (e.g., SAR, matrix-calibrated SAR, countryside SAR) for the last one.

2.3.5 *Reference situation*

Reference situation is a “standard” condition for assessing environmental impacts related to biodiversity and ESs. According to Koellner et al. (2013), three options are available from the literature:

- Applying “Potential Natural Vegetation (PNV)”: “which describes the expected state of mature vegetation in the absence of human intervention” (Chiarucci et al. 2010, 1172).
- Considering natural or semi-natural land cover as a reference status for each region/biome.
- Using the actual mix of land use in Europe (Koellner and Scholz 2006).

It is very important to consider a reference situation to understand the extent of impact or how much a driver affects ESs or biodiversity. The UNEP-SETAC Guidelines (Koellner et al. 2013) and Milà i Canals et al. (2007) suggested using the second option, considering the quantity and quality of data available and the impracticality of land use.

2.4 **Conclusions**

This review represents, to the authors’ knowledge, the first overview of LC methodologies applied to ESs studies in the agricultural and forestry sectors.

Although this selection strategy may have led to the exclusion of studies using less standardised terminology, the core principles of LCA integration with ecosystem services remain well represented in the analysed dataset. Future research could explore additional terminological variations to further validate these findings.

The topic of ESs in LCA in these two fields was increased after the publication of UNEP-SETAC guidelines and is evaluated, more or less, constantly, even though there is no uniform framework to implement these two topics efficiently. The ESs most analysed were “provisioning” and “regulating” services, even though there is an interest in “cultural” and “supporting” services. Identifying from the studies analysed the relationships between indicators and ecosystem services was a difficult task because ESs are connected one to the other. More detailed classifications, such as the Common International Classification of Ecosystem Services (CICES) can help to overcome this problem by identifying more precisely the ecosystem services through coding; it would allow the double-counting problem, which could occur, for example, in the evaluation of soil quality that can be considered "regulating" and "supporting" service.

In general, LC approaches are applied following current advice about the functional unit, system boundaries, and life cycle inventory. Regarding impact categories, the common LCIA methods are also used for the assessment of ESs thanks to their easy estimation through LCA software. They can analyse different pressures on ESs caused mainly by land use and land-use change. Of particular interest is the UNEP-SETAC methodology (Koellner et al. 2013) because it allows for the assessment of different aspects of ESs, in particular, "regulating" services such as Biotic Primary Production (BPP), Climate Regulation Potential (CRP), Erosion Regulation Potential (ERP), Biodiversity Damage Potential (BDP).

The integration of ecosystem services into LCA can serve multiple stakeholders with distinct needs. Researchers benefit from detailed impact assessment models that allow for methodological refinements. Policymakers may prioritise frameworks that provide clear, actionable sustainability indicators. Businesses and industry stakeholders require streamlined LCA tools that can efficiently support decision-making processes.

Future research should focus on the assessment of cultural and supporting ecosystem services, as these aspects remain largely underrepresented in the current literature. Likewise, integrating economic and social dimensions into LCA would be a valuable addition, as only a limited number of studies currently incorporate these methodologies. Expanding LCA to include these new dimensions would provide a more holistic understanding of how alterations to ecosystem services impact not only environmental sustainability but also economic and social outcomes.

Abbreviations

ALCA	Attributional Life Cycle Assessment
BPP	Biotic Production Potential
CICES	Common International Classification of Ecosystem Services
CLCA	Consequential Life Cycle Assessment
CRP	Climate Regulation Potential
ERP	Erosion Regulation Potential
ESs	Ecosystem Services
HANPP	Human appropriation net primary production

LCA	Life Cycle Assessment
LCC	Life Cycle Costing
MA	Millennium Ecosystem Assessment
NPP	Net primary production
PDF	Potentially Disappearing Fraction
PNV	Potential Natural Vegetation
PRISMA	Preferred Reporting Items for Systematic Reviews and Meta-Analyses
SAR	Species-Area Relationship
SETAC	Society of Environmental Toxicology and Chemistry
TEEB	The Economics of Ecosystems and Biodiversity
WoS	Web of Science
WPP-PCF	Water Purification Potential related to physicochemical filtration
WPP-MF	Water Purification Potential related to mechanical filtration

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Chapter 3.
MODELS AND METHODS: STATE-OF-ART[2]

Using the systematic review from Chapter 2, this chapter moves the study forward by sorting and reviewing current frameworks for putting ecosystem services into a life cycle context. The goal is to find out what these approaches' methodological strengths and weaknesses are, so it is possible to see how well they work in different environmental and economic assessment models. The goal of this chapter is to find important gaps and chances to make Life Cycle (LC) methods for evaluating Ecosystem Services (ESs) more reliable. Because it's hard to include ESs in the LC perspective, many models and frameworks have been created to help make assessments more accurate. This study examines some of the most widely used frameworks, highlighting their main strengths and weaknesses. Additionally, it aims to determine whether a particular framework effectively integrates environmental and economic assessments within life-cycle approaches.

To achieve this objective, this chapter seeks to answer the following research question:

1. Which frameworks and methodological approaches provide the most robust integration of ESs within LC, and how do they differ in their applicability across environmental and economic assessment models?

3.1 **Materials and methods**

Based on the research by Rugani et al. (2019) and the results of a literature review by Soldati et al. (2022), this paper has made a distinction between empirical studies that use one or more specific assessment frameworks for ecosystem services. This paper's investigation has distinguished between empirical studies that use one or more specific assessment frameworks for ecosystem services, based on the analysis conducted by Rugani et al. (2019) and the findings of a literature review by Soldati et al. (2022). According to the Oxford, a framework is defined as “a set of beliefs, ideas, or rules that is used as the basis for making judgments, decisions, etc.”. Based on this definition, the studies can be

[2] This chapter is based on:

1. the following oral contribution: Soldati, C., Falcone, G., Iofrida, N., Spada, E., Gulisano, G., De Luca, A. I. Evaluating Frameworks for the Integration of Ecosystem Services (ESs) in Life Cycle Methodologies. In Conference Procedeeings “30 anni di Life Cycle Assessment: sviluppi metodologici e applicativi”. June 28-39 2023. Milan. Personal contribution to the article: setting up the methodology, conduction of literature research, data analysis and writing of the first draft. These activities were conducted in collaboration with the other co-author
2. the following oral contribution (it will be published in the future in “Lecture Notes in Networks and Systems” (Springer)): Soldati, C., Iofrida, N., Gulisano, G. Integrating Ecosystem Services into Life Cycle Methods: An Analytical Assessment of Frameworks and Challenges. May 22-24 2024. Reggio Calabria. Personal contribution to the article: setting up the methodology, conduction of literature research, data analysis and writing of the first draft. These activities were conducted in collaboration with the other co-author

categorised into two distinct types: “biophysical” frameworks, which focus exclusively on environmental valuations, and “monetary” frameworks, which include economic valuations (Harrison et al. 2018). Through a screening process, 26 works were identified and grouped into 21 primary frameworks (Table 3.1). Articles were excluded if they were reviews or guidelines without a methodological development that could be evaluated. In particular, two consolidated ESs valuation frameworks are noteworthy: the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) framework developed by the Natural Capital Project (Sharp et al. 2018) and the Multiscale Integrated Earth Systems Model (MIMES) framework (Boumans et al. 2015). Furthermore, these frameworks are very important because they can be used to do financial assessments of ESs and can be used on their own or with Life Cycle (LC) methodologies.

This investigation's approach aligns with the need to categorise and thoroughly understand the methodologies employed in ESs assessments. By distinguishing between biophysical and monetary frameworks, the study provides a clearer picture of how different types of valuations are approached and applied in empirical research. The InVEST and MIMES frameworks are highlighted due to their comprehensive capabilities and adaptability, making them valuable tools for standalone and integrated assessments.

Table 3.1. Evaluated papers and frameworks.

Frameworks	Reference	Type
Ecologically Based LCA (Eco-LCA)	Zhang et al. (2010)	B
Bare 2011’s framework	Bare (2011)	B
ES-LCA (Cascade model)	Rugani et al. (2019)	B+M
Nexus	Karabulut et al. (2018)	B
UNEP-SETAC	Koellner et al. (2013)	B
Techno-Ecological Synergies (TES-LCA)	Bakshi et al. (2015); Liu et al. (2018b); Liu and Bakshi (2019)	B
LUCI-LCA	Chaplin-Kramer et al. (2017)	B
Blanco (2018)’s framework	Blanco et al. (2018)	B+M
Müller-Wenk and Brandão (2010)’s framework	Müller-Wenk and Brandão (2010)	B
Brandão and Milà i Canals (2013)’s framework	Brandão and Milà i Canals (2013)	B
Alejandre et al. (2019)’s framework	Alejandre et al. (2019)	B
Saad et al. (2013)’s framework	Saad et al. (2013)	B
Cao et al. (2015)’s framework	Cao et al. (2015)	B+M
Núñez et al. (2013)’s framework	Núñez et al. (2013)	B
Liu et al. (2018)’s framework	Liu et al. (2018a)	B
Xue et al. (2014)’s framework	Xue et al. (2014)	B
Ecotope formation’s framework	Baitz (2002)	B
Alejandre et al. (2022, 2023)’s framework	Alejandre et al. (2022, 2023)	B
Environmental Priority Strategies in Product Design 2015 (EPS2015) (Steen 2015)	Steen (2015)	B+M

Land Use Indicator Value Calculation in Life Cycle Assessment (LANCA) (Beck et al. 2010; Bos et al. 2016; Horn and Maier 2018)	Beck et al. (2010); Bos et al. (2016); Horn and Maier (2018)	B
InVEST	Sharp et al. (2018)	B+M
MIMES	Boumans et al. (2015)	B+M

In the last column: B= "biophysical evaluation," M= "monetary evaluation," B+M= "both biophysical and monetary evaluation"

The next step was to put the frameworks into groups based on nine specific criteria that were set after a thorough full-text analysis and thought about the main points of all the chosen studies (Table 3.2). Subsequently, these frameworks were categorised based on these nine particular criteria, derived from a meticulous review of the full text and taking into account the key characteristics shared by all the chosen studies. Based on these criteria, a tiered approach and scoring system for a comprehensive assessment have been developed.

Table 3.2. Criteria of analysis in a tiered approach to frameworks for ESs evaluation.

Criteria	Description	Scoring system
Expertise Level	Identify the ease of application of the proposed framework, even with previous gaps derived from other fields.	S = The framework is easy to understand and apply B = The framework is fairly simple but requires specific expertise to interpret some methodological steps D = The framework requires specific expertise to understand and apply.
Provisioning Services	Identify coverage of provisioning services	S= The framework covers all provisioning services B= The framework covers several provisioning services D= The framework does not cover the provisioning services or does not provide any information
Regulating Services	Identify coverage of regulating services	S= The framework covers all regulating services B= The framework covers several regulating services D= The framework does not cover the regulating services or does not provide any information
Supporting Services	Identify coverage of supporting services	S= The framework covers all supporting services B= The framework covers several supporting services D= The framework does not cover the supporting services or does not provide any information
Cultural Services	Identify the coverage of cultural services	S=The framework covers all cultural services B= The framework covers several cultural services D= The framework does not cover the cultural services or does not provide any information
Biodiversity	Identify the coverage of biodiversity (which is considered a special category of ES)	S= The framework covers biodiversity B= The framework covers partially biodiversity D= The framework does not cover biodiversity
Monetary valuation	Identify the possibility of incorporating economic evaluation into the framework	S= The framework integrates economic assessment into environmental assessment B The framework partially integrates economic assessment into environmental assessment D= The framework does not integrate economic assessment into environmental

		assessment
Input data	Identify the necessary characteristics of the input data, such as the amount of data required and the ease with which it can be obtained	S= The framework uses a set of data that is very easy to obtain. B= The framework uses a lot of data, and/or some calculations need to be integrated. D= The framework does not provide any information about the data needed and how to obtain it
Output data	Identify the type of output returned by the framework and its interpretation	S= The framework provides some outputs, but they are straightforward to interpret B= The framework provides some outputs, but these need to be interpreted by an expert D= The framework does not provide any information about the outputs it provides

Once the frameworks had been scrutinised according to the established criteria, they were ranked based on how well they met the requirements, similar to Vidal Legaz et al. (2017) work. The rankings were divided into three tiers: S (best performance), B (average performance), and D (worst performance). An overall assessment was also made by converting these levels into numerical scores: 1 for S ratings, 3 for B ratings, and 5 for D ratings. The average was then calculated, rounding up or down as necessary to give a single score. Finally, a new reconversion was carried out, assigning two new ratings, A and C, based on the average calculation, which yielded scores of 2 and 4, respectively. This thorough evaluation process ensured a comprehensive and nuanced understanding of the performance of each framework.

3.2 **Results and discussion**

Table 3.3 displays the classification of the selected frameworks (Table 3.1) based on the aforementioned criteria. As can be seen, all methods require a certain level of expertise due to the use of supplementary tools, technical skills, or matrix forms of Life Cycle Inventory (LCI). For example, Zhang et al. (2010), Bakshi et al. (2015), Liu et al. (2018b) and Liu and Bakshi (2019) evaluate ecosystem services (ESs) by employing engineering concepts such as emergy, industrial cumulative exergy consumption (ICEC), or ecological cumulative exergy consumption (ECEC). However, the implementation of the TES-LCA framework offers a compelling resolution. It enables the combination of the impacts of ESs that exhibit variability, which can arise from the use of diverse units of measurement on a standardised scale, namely emergy. Additionally, the utilisation of a hierarchical set of metrics simplifies the analysis of the system being examined. The only study that uses qualitative analysis is the one by Karabulut et al. (2018), which looks at how the investigated ESs improved or got worse. It also considers how these services interact, including their synergistic or antagonistic effects. This approach highlights the importance of considering both the quantitative and qualitative aspects of ESs in comprehensive assessments.

Table 3.3. Valuation of frameworks considering the selected criteria.

Framework / **Criteria**	**Expertise Level**	**Provisioning Services**	**Regulating Services**	**Supporting Services**	**Cultural Services**	**Biodiversity**	**Monetary valuation**	**Input data**	**Output data**	**Overall tiered**
Zhang et al. (2010)	D	B	B	B	D	D	D	D	D	C
Bare (2011)	B	B	B	B	B	B	B	D	B	B
Karabulut et al. (2018)	B	B	D	D	D	D	D	D	D	D
Blanco et al. (2018)	B	B	B	D	D	D	B	B	B	C
TES-LCA (Bakshi et al. 2015; Liu et al. 2018b; Liu and Bakshi 2019)	D	B	B	B	D	D	D	B	D	C
Chaplin-Kramer et al. (2017)	B	D	B	D	D	B	S	D	S	B
Müller-Wenk and Brandão (2010)	B	D	B	D	D	D	D	B	S	C
Brandão and Milà i Canals (2013)	B	B	D	D	D	D	D	B	S	C
Alejandre et al. (2019)	S	B	B	D	D	D	S	B	S	B
Saad et al. (2013)	B	D	B	D	D	D	D	S	S	C
Cao et al. (2015)	B	B	B	D	D	D	B	B	S	B
Núñez et al. (2013)	B	D	B	B	D	D	D	S	B	C
Xue et al. (2014)	S	D	B	D	D	D	B	S	S	B
Liu et al. (2018a)	B	D	B	D	D	D	D	S	B	C
Liu et al. (2020)	B	B	B	D	D	D	B	S	S	B
LANCA (Beck et al. 2010; Bos et al. 2016; Horn and Maier 2018)	B	D	B	B	D	D	D	B	B	C
Baitz (2002)	B	D	D	B	D	B	D	B	D	C
Alejandre et al. (2022, 2023)	S	D	B	D	D	B	D	S	S	B
EPS2015 (Steen 2015)	S	S	D	D	D	D	S	S	S	B
InVEST (Sharp et al. 2018)	B	B	D	B	D	S	B	B	B	B
MIMES (Boumans et al. 2015)	B	B	D	B	B	S	B	B	B	B

The assessment of ecosystem services (ESs) is a controversial topic in framework analysis. Some frameworks explicitly mention which services are analysed (e.g., Zhang et al. (2010); Chaplin-Kramer et al. (2017); Karabulut et al. (2018)), while others do not Theoretically, it is thought that these types of services are either not valued (like cultural services, which are rarely thought of) or are too broad to be included in the theoretical framework for methods. For example, Liu et al. (2020) paper shows how the framework from Rugani et al. (2019) can be used in real life, though with some changes, and how the associated ESs are valued. Provisioning services are generally well-described within these frameworks, although some do not include all of them. For example, Zhang et al. (2010) do not consider certain provisioning services, such as genetic resources or wildlife products, because there is currently no method to account for them in Life Cycle Assessment (LCA). Regulatory services, such as soil erosion control (Koellner et al. 2013) and carbon sequestration (Bare 2011), are the most commonly considered. Among the reviewed frameworks, only two consider pollination services. Zhang et al. (2010) involve the calculation of pollination through the Eco-LCA methodology using method-specific metrics. Similarly, Alejandre et al. (2022, 2023) allow the analysis of pollination through the assumption of Lonsdorf's model (Lonsdorf et al. 2009), which presents a weakness in not assessing pollinators' species richness. Cultural and support services are the least studied. Cultural services are related to human subjective perceptions, while support services are considered “already provided” as they form the basis for provisioning, regulating, and cultural services.

It's interesting that Zhang et al. (2010) still rate support services by looking at soil formation, photosynthesis, primary production, nutrient cycling, and water cycling, even though water cycling is only partially talked about. Some studies (Zhang et al. (2010); Bakshi et al. (2015); Liu et al. (2018b); Liu and Bakshi (2019); Rugani et al. (2019)) don't look at biodiversity at all, while others do a good job of including it (Bare (2011); Koellner et al. (2013); Chaplin-Kramer et al. (2017)). Biodiversity valuation mainly considers species diversity, as genetic biodiversity has not yet been implemented, while ecosystem biodiversity can be evaluated through life cycle categories such as acidification or eutrophication. Monetary valuation of ESs is rarely applied. Rugani et al. (2019) give a lot of information about how to turn the environmental value of ESs into economic values. They do this by multiplying impact values from the impact assessment phase by prices from other sources. Economic evaluations conducted by frameworks such as InVEST and MIMES are also noteworthy. InVEST uses estimation techniques for its modules, applying Net Present Value (NPV) for sequestered carbon and the cost of avoided treatments for nutrient retention. It also conducts more straightforward economic assessments, such as a pollination module calculating a simplified bee index for agricultural production. Similarly, MIMES uses production functions and shadow prices for economic services, incorporating willingness to pay for these services. Expertise is needed to apply these models due to underlying assumptions derived from models outside the LCA framework. This includes evaluating outputs, as results from external models may not be easily interpreted (e.g., trends in soil-stored organic carbon over time), as well as applying the framework itself. Some frameworks, like the TES-LCA method (Zhang et al. 2010; Bakshi et al. 2015; Liu et al. 2018b; Liu and Bakshi 2019), involve calculating multiple metrics such as Resource Intensity, Efficiency, Renewability Index, and Physical Return on Investment. These metrics often require further indicators, like ICEC or ECEC, involving the transformation of inventory flows into a standardised energy unit, such as exergy and emergy. Rugani et al. (2019) computed a matrix model of life-cycle inventory (LCI) within the ES cascade model framework. This model uses various matrices in next stages of the cascade model. Liu et al. (2020) did a study that used the Environmental Policy Integrated Climate (EPIC) software for the inventory and impact assessment phases. This software made some parts of the calculation easier, but it demands a considerable amount of input data, including factors such as land use, weather conditions, economic data, and crop information. Given the high variability, even at the local scale, a single framework may not be adequate for assessing ESs. Combining multiple frameworks with georeferenced information could lead to more reliable results. For example, Liu et al. (2020) used the online tool i-tree canopy to evaluate air purification through georeferenced analysis of pollutants. This tool looks at the pros and cons of an existing area, the cost of cleaning up pollution, and how to improve things by connecting survey points to land cover classes. The LANCA model, currently implemented in LCA software like GaBi or Simapro, aims to provide a georeferenced version using raster databases. Similarly, the model for calculating ecotope formation can be updated with georeferenced data referring to species richness. Although there is no georeferenced information for pollination calculation, according to Alejandre et al. (2022, 2023), future developments could include databases for pollinators, akin to the GLOBIO model through Mean Species Abundance (MSA). Finally, the EPS2015 method (Steen 2015) is recommended for provisioning services. Although it does not possess georeferencing, it is already implemented in LCA software and aligns with life cycle approach principles.

3.3 **Conclusions**

The predominance of EPS2015 in this evaluation does not imply absolute superiority over other methods but reflects its suitability for monetising environmental impacts, a key aspect of decision-making frameworks. Its focus on monetary values, on the other hand, might mean that it lacks some ecosystem services that are hard to measure in economic terms. This shows the need for different approaches to work together. To address this, LANCA was integrated to assess soil-related ecosystem functions, while i-Tree enhances the evaluation of regulating services through spatially explicit analysis. The combined use of these tools strengthens methodological robustness and mitigates the limitations of individual models.

Integrating georeferenced data makes it easier to show how impacts change over time and space, which improves the accuracy of assessments. This is because ecosystem services, especially those provided by soil, are naturally diverse. Creating separate databases for geospatial models is also important to make sure that environmental sustainability assessments can be done quickly and easily within the LCA framework.

Recent efforts to improve ecosystem service evaluation have led to the development of various models within the broader life cycle framework. While each model has strengths and weaknesses, they mark a significant step toward ES quantification, which has often been overlooked in favour of classification initiatives. Because ecosystem services, especially those related to soil, change over time, it is important for models to include georeferencing in order to show differences at the local level, like on a farm. Along with building models, creating geospatial databases is necessary for making life cycle analysis more efficient and easier to use for sustainability assessments.

Abbreviations

BDP	Biodiversity Damage Potential
CICES	Common International Classification of Ecosystem Services
ECEC	Ecological cumulative exergy consumption
Eco-LCA	Ecologically Based LCA
EPIC	Environmental Policy Integrated Climate
EPS	Environmental Priority Strategies in product design

ESs	Ecosystem Services
ES-LCA	Ecosystem Services LCA
ICEC	Industrial cumulative exergy consumption
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs
LANCA	Land Use Indicator Value Calculation in Life Cycle Assessment
LC	Life Cycle
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
MA	Millennium Ecosystem Assessment
MIMES	Multiscale Integrated Earth Systems Model
MSA	Mean Species Abundance
NPV	Net Present Value
TEEB	The Economics of Ecosystems and Biodiversity
TES	Techno-Ecological Synergies LCA

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Chapter 4.
MATERIALS AND METHODS

4.1 **The SustainOlive project**

The SustainOlive project, an acronym for "novel approaches to promote the SUSTAINability of OLIVE cultivation in the Mediterranean," is an international initiative comprising a consortium of 22 partners from Spain, Portugal, Italy, Greece, Tunisia, and Morocco. It is funded by the PRIMA (Partnership for Research and Innovation in the Mediterranean Area) program and co-funded by Horizon 2020. This project seeks to identify solutions and strategies that harmonise profitable and sustainable practices in olive cultivation, safeguarding the environment and preventing the overexploitation of natural resources by integrating ecological, territorial, and socioeconomic knowledge while designing and assessing sustainable strategies and methods rooted in agroecological principles.

The overarching vision of Sustainolive[3] is to formulate innovative strategies associated with management practices rooted in agroecological principles, aiming to create more efficient and context-specific solutions for Mediterranean olive environments through the active participation of various stakeholders in the implementation of Sustainable Technological Solutions (STSs) and measures.

The discovery and execution of a site-specific array of STSs is fundamental to the many responsibilities of the project. Informed by scientific principles of agroecology, the STSs to be evaluated on experimental olive farms have been collaboratively established with farmers. These techniques emphasise the use of natural resources and the closure of biological cycles at the farm or local level, which involves minimising external inputs and enhancing the quality and efficiency of internal inputs. These are either grown and reproduced on the farm or bought through socially regulated trade between farmers and other people in the food chain. This helps farmers become more independent, which is important for both economic and environmental resilience (van der Ploeg et al. 2019). Research increasingly suggests (e.g., Jeswani et al. (2018)) that the internal regulation of ecological functions in agroecosystems is significantly influenced by their biodiversity, which provides a range of agroecosystem services beyond food production, including nutrient recycling and retention, micro-climate regulation, local hydrological processes, and pest

[3] https://sustainolive.eu

and disease control or mitigation. These lead to more benefits, like protecting crops and making the soil fertile, even when outside inputs like water, nutrients, and so on are limited, which happens a lot in the Mediterranean. SustainOlive wants to find out how much better things could be by putting together management and strategic plans for increasing the variety of life in olive farming systems.

Farms were chosen as case studies for the project based on their compatibility with the Sustainolive experimental design. As a result, the selection included both traditional olive farms and others that were similar in terms of area, plant age and density, yields, and pedoclimatic conditions but used sustainable farming techniques (STSs). Consequently, the Sustainolive initiative selected pairs of STS and non-STS olive farms from Portugal, Spain, Italy, Greece, Morocco, and Tunisia. Each pair included a standard conventional olive grove operating as usual, while the other member was a similar olive farm that had implemented a specific combination of agroecological management strategies for a minimum of 8 years.

In the survey concerning olive farms, 59 key informants from various olive farms participating in the Sustainolive project were selected, comprising 6 Portuguese farms, 16 Spanish farms, 7 Moroccan farms, 11 Tunisian farms, 9 Italian farms, and 10 Greek farms. Direct interviews were conducted in 2022 by interviewers who were adequately trained and instructed to collect data uniformly across all participating countries.

The specific characteristics of the olive farms that use STS practices are reported in Table 4.1.

Table 4.1 Farms analysed and their characteristics.

Farm ID	Country	STS characteristics
N01_CHR_STS_GR	Greece	Shredding and use of chicken manure
N02_CHR_STS_GR	Greece	Use of pomace and chicken manure
N07_KAL_STS_GR	Greece	Organic, Shredding, Use of compost
N08_KAL_STS_GR	Greece	Low production, high incidence of fencing and irrigation system, Use of compost, Shredding, Burning of 50% of pruning residues, Very expensive irrigation system
N03_CHR_STS_GR	Greece	Use of pomace and manure Very high energy cost for irrigation
N01_POR_STS_PT	Portugal	Organic, Cover crops, Shredding of pruning residues
N03_ALC_STS_PT	Portugal	Conventional, Tillage, Pruning rotation system, No chemical inputs
N06_SER_STS_PT	Portugal	Organic, manure application, pruning residue shredding, no herbicides
N01_DE_STS_SP	Spain	Organic, Application of compost, olive pomace and mill residues, Shredding of turf, Shredding of pruning residues

N03_ES_STS_SP	Spain	Organic, Use of pomace, leaves from the mill, manure, shredding and grazing
N05_PV1_STS_SP	Spain	Organic, Organic fertiliser, Shredding
N07_PV2_STS_SP	Spain	Integrated, Shredding, Spontaneous grassing
N08_JT_STS_SP	Spain	Organic, Use of pomace, manure, 60 sheep, Continuous grazing
N09_IS_STS_SP	Spain	Integrated, Chemical weeding, Shredding, Spontaneous grassing
N10_CH_STS_SP	Spain	Integrated, Chemical weed control
N12_MR_STS_SP	Spain	Integrated, use of olive leaves, shredding
N13_PA_STS_SP	Spain	Organic, Manure application, Shredding of pruning residues, Grazing with sheep
N15_GA_STS_SP	Spain	Weeding, Shredding, Adding Manure
N18_CC_STS_SP	Spain	Organic, Application of compost and pomace, Shredding of residues
NO7_CA_STS_IT	Italy	Organic, Use of compost
NO1_PU_STS_IT	Italy	Integrated agriculture, fertilisation with manure, shredding of pruning residues and grass cover, no use of herbicides
NO5_TO_STS_IT	Italy	Organic, Shredding, Manure Use
NO3_LA_STS_IT	Italy	Integrated, shredding, processing, organic mineral fertilisation, green manure, shredding of pruning residues
NO9_LI_STS_IT	Italy	Organic, Shredding, Manure Use
N02_OUA_STS_MO	Morocco	Burnt pruning residues
N03_AME_STS_MO	Morocco	Green manure, manure
N05_TAN_STS_MO	Morocco	Burning of pruning, addition of livestock manure
N06_OUA_STS_MO	Morocco	Burning of pruning residues, very low production
N02_SID_STS_TU	Tunisia	Organic, Spreading of vegetation water, Use of pruning residues for animal feed
N04_MEN_STS_TU	Tunisia	Organic, Grazing by herbivores, who also eat pruning residues
N09_CHI_STS_TU	Tunisia	Pruning residues are shredded and "composted"
N11_SID_STS_TU	Tunisia	Pruning residues are used for animal feed
N01_TOU_STS_TU	Tunisia	
N05_ELF_STS_TU	Tunisia	Organic, cover crops, shredding of pruning residues

4.2 **Methods chosen**

Because there are many frameworks for evaluating ecosystem services (ES), picking the best methods for this study required a lot of thought based on the methods that were looked at in the previous chapter. A multi-method approach was used that combined EPS2015, LANCA, and i-Tree to make sure that the evaluation was complete in the context of Life Cycle Assessment (LCA). These tools were chosen for their complementary strengths in capturing economic, biophysical, and spatial aspects of ESs.

The EPS2015 method was chosen for its well-established capability in monetising environmental impacts, making it particularly suitable for decision-making contexts. To address its limitations in assessing non-market ESs, LANCA was integrated to enhance the evaluation of soil-related ecosystem functions. Additionally, i-Tree was chosen to improve the assessment of regulating services through spatially explicit modelling. It makes the results more reliable and helps show how impacts change over space more accurately, which is important because ESs are naturally different. Lastly, the Ecotope Formation method was added to take into account how landscapes can build and keep ecological structures by interacting with living and non-living things. This metric provides a broader perspective on ecosystem functionality by integrating species richness, structural diversity, and human influence. Landscape-scale ecological processes are becoming more important in sustainability assessments. Using Ecotope Formation makes the analysis more complete by including things that traditional LCA-based methods don't directly look at.

The selected approaches were chosen to ensure a comprehensive evaluation of ecosystem services. The following sections provide a detailed explanation of each method and its role within this assessment framework.

4.2.1 *EPS 2015*

The EPS development started in 1989 with the requirement made by Volvo and with the collaboration of several institutes such as the IVL Swedish Environmental Research Institute, Volvo itself, and the Swedish Federation of Industries (inside the Swedish Life Cycle Center).

This method can be applied in LCA and includes an evaluation method composed of a characterisation and ponderation model. The results are expressed as damage costs for emissions and natural resource use, and they are expressed in Environmental Load Units (ELUs). One ELU represents an externality that corresponds to one Euro under specific conditions (Steen 2015).

This impact assessment method is available in two versions: EPS2015dx, which excludes climate impacts caused by secondary particles (e.g., SO_2 and NO_x) due to uncertainties in their modelling, and EPS2015d, which includes these impacts but requires careful interpretation of the results (Steen 2015).

The EPS follows five hierarchical principles (Steen 2015):

1. Top-down principle → In typical product development contexts, the potential to improve the product's environmental performance determines the use of any model, data, or process. This suggests weighing the product's enhanced environmental performance against the time and cost of the evaluation.
2. Index principle → A product life cycle can be described in terms of materials and processes using the EPS system, and ready-made weighted impact assessments can be used as indices for these descriptions. The indices will show the individual weighted and aggregated effects of different types of materials on the environment with regard to their production, processing, and waste management.
3. Default principle → For a rapid study of any product design aspect under consideration, default indices shall be made available. Later, more precise information might be employed.
4. Uncertainty principle → Reality includes uncertainty; thus, it will also be included in the analysis. An uncertainty measure and the best estimate must be used to represent data. Sensitivity studies will be available to show how rigidly a priority is set.
5. Choice of default index → Actual default indices are chosen while taking into account the user's needs and state of understanding. Both inventory data and effect assessment data are subject to decision-making. In order for inventory data to be utilised in a modular fashion for the manufacturing, processing, and waste management of materials or components, it must be consistently arranged with adherence to allocation criteria. Data from impact assessments are arranged to reflect the monetary values of the effects that emissions and resource consumption have on the environment. Thus, default indices represent in monetary terms the environmental effects of the manufacture, processing, and disposal of materials and components.

Monetary values in EPS are considered as follows (Steen 2015):

1. With the product concepts under consideration, the monetary measure aims to inform the product creator of the relevance of the sustainability consequences they make. The monetary measure accomplishes this by applying values that individuals similar to the product creator—specifically, the typical OECD resident—would assign to the state metrics.

2. The amounts are expressed in environmental load units (ELU), where, under certain circumstances, 1 ELU is equivalent to 1 EUR.
3. There is a 0% discounting of future effects.

With this method, 5 ESs are analysed (4 provisioning + 1 cultural), and all of them are based on measured value as market values (Steen 2015):

- Crop growth capacity [kg]
- Production capacity for fruit & vegetables [kg]
- Wood growth capacity [kg]
- Fish&meat production capacity [kg]
- Quality time [person·year].

However, “quality time” is not implemented in LCA software as well as provisioning services, and the model to calculate separately is not available in the documents that illustrate the methods. For this reason, this service is excluded from the analysis of this work.

4.2.2 *LANCA model*

The Fraunhofer Institute for Building Physics developed the first version of the Land Use Indicator Value Calculation for Life Cycle Assessment (LANCA) (Beck et al. 2010). This method was later revised by Bos et al. (2016), introducing significant updates, including modifications to the calculation of Erosion Resistance. Horn and Maier (2018) further refined the Characterisation Factors (CFs), leading to the latest version of LANCA, developed within the Orienting project, a European initiative aimed at enhancing sustainability assessments.

The current version of LANCA evaluates five key soil quality indicators, which are also considered ecosystem services (ESs):

- Erosion Resistance (ER)
- Mechanical Filtration (MF)
- Physicochemical Filtration (PCF)
- Groundwater Regeneration (GWR)
- Soil Organic Carbon (SOC)
- Human Appropriation for Net Primary Production (HANPP)
- Biodiversity (BioMAPS)

Initially, GWR, HANPP and BioMAP were also included. However, these indicators were excluded due to data availability constraints and the complexity of integrating their results into the selected assessment framework. Given these limitations, alternative indicators were chosen to maintain methodological consistency and ensure applicability to the case study.

The sections that follow go into more detail about each chosen indicator. They do this by using sources like the LANCA 2.0 Handbook (Bos et al. 2016), Ulrike Bos' dissertation (Bos 2019), and Orienting project documents.

4.2.2.1 Erosion Resistance

Soil erosion is a process of soil removal and movement driven by two atmospheric agents: water and wind, according to the most recent edition of the LANCA Handbook (Bos et al. 2016). In addition to the hydrological cycle, the nutrient cycle, and productivity/fertility, all are threatened by soil erosion. Erosion Resistance is, therefore, a crucial ES and ought to be used in the evaluation of land use.

As one of the most used models, the RUSLE model is used to evaluate Erosion Resistance. The Revised Universal Soil Loss Equation (RUSLE) model is an update to the Universal Soil Loss Equation (USLE) model. It is based on information about how land is managed and a number of factors, such as rainfall, soil type, and terrain. It is crucial to stress that USLE/RUSLE models only take into account erosion brought on by water and ignore erosion brought on by the wind.

Soil loss is calculated through Equation 4.1 as follows:

Equation 4.1 RUSLE equation (Renard et al. 1997).

$$A = R \cdot K \cdot LS \cdot C \cdot P$$

Where:

A is the predicted average annual soil erosion rate caused by water [t ha^{-1} y^{-1}]

R is the rainfall erosivity factor [MJ mm ha^{-1} h^{-1} y^{-1}]

K is the erodibility factor [t ha h MJ^{-1} ha^{-1} mm^{-1}]

LS is the slope length factor [dimensionless]

C is the land cover factor [dimensionless]

P is the support practice factor [dimensionless]

Therefore, the Erosion Resistance can be calculated through the opposite of this equation, namely (Equation 4.2):

Equation 4.2 Erosion Resistance formula (Bos et al. 2020).

$$ER = -(R \cdot K \cdot LS \cdot C \cdot P)$$

Where:

ER is the erosion resistance [$t\ ha^{-1}\ y^{-1}$].

Figure 4.1 defines, respectively, the calculation steps necessary for the calculation.

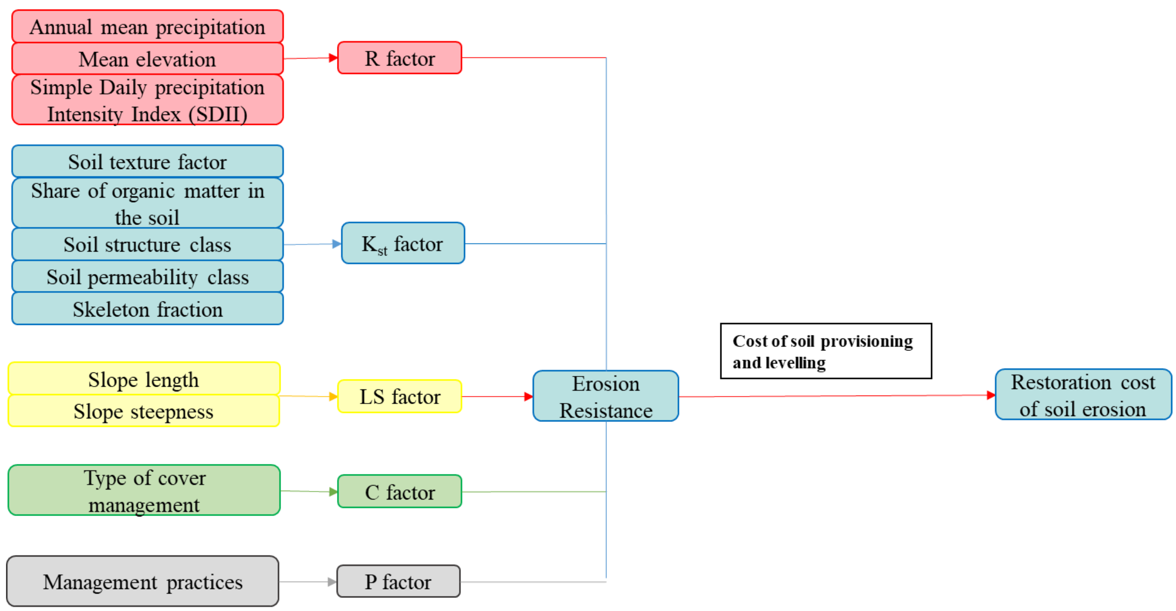


Figure 4.1 Steps to calculate Erosion Resistance in LANCA model (elaboration based on Bos et al. (2016)).

4.2.2.1.1 Rainfall erosivity factor

The Rainfall Erosivity Factor (R-factor) describes the transfer of kinetic energy from a raindrop to the soil. It depends on the quantity of kinetic energy per area and the intensity of precipitation. The R-factor can be obtained in two ways: firstly, by multiplying the total kinetic energy of the rainfall for the maximum rain intensity in 30 minutes, and secondly, by using a regression equation.

In the older version of LANCA, this part was found using a set of regression equations that were specific to the area defined by the Köppen & Geiger climate zone classification (Rubel and Kottek 2010).

However, the new version of LANCA is GIS-based, the “Global Rainfall Erosivity,” a rainfall erosivity map that covers the entire world with a resolution of 30 arc-sec (~1 km at the Equator), was downloaded.

4.2.2.1.2 Erodibility factor

The erodibility factor (K-factor), as defined by Wischmeier and Smith (1978), describes the susceptibility of the soil to erosion. The first version of the formula for the calculation of the K-factor was proposed by Renard (1997) and subsequently modified by Panagos et al. (2014) to include the stoniness sub-factor St (Equation 4.3):

Equation 4.3 K-factor equation (Panagos et al. 2014).

$$K = \frac{2.1 \cdot 10^{-4} \cdot M^{1.14}(12 - OM) + 3.25(s - 2) + 2.5(p - 3)}{100} \cdot 0.1317 \cdot St$$

Where:

M is the textural sub-factor [dimensionless]

OM is the organic matter content sub-factor [dimensionless]

s is the soil structure class sub-factor [dimensionless]

p is the permeability class sub-factor [dimensionless]

St is the stoniness sub-factor [dimensionless]

The textural factor (M-sub-factor) describes the composition of the soil texture, taking into account the amount of clay, sand, and silt, and it can be calculated using the following equation (Renard et al. 1997; Panagos et al. 2014) (Equation 4.4):

Equation 4.4 M-sub-factor (Renard et al. 1997; Panagos et al. 2014).

$$M = \left(\%_{silt} + \%_{fine\ sand}\right) \cdot \left(100 - \%_{clay}\right)$$

The organic matter share is described by OM-sub-factor, and it can be derived from several sources, such as the soil analysis, the application of formulas, or the use of databases like Harmonized Soil World Database (HSWD).

The soil structure class (s-sub-factor) is defined by Panagos et al. (2014), which takes into account the following values (Table 4.2):

Table 4.2 s-sub-factor (Panagos et al. 2014).

	Soil structure class	Particles sized d_{max} (mm)
1	very fine granular	2
2	fine granular	5
3	medium or coarse granular	10
4	blocky, platy, or massive	100

The permeability class (p-sub-factor) is similar to the S-sub-factor and, according to Panagos et al. (2014), assumes the following values (Table 4.3):

Table 4.3 p-sub-factor (Panagos et al. 2014).

	Permeability class	Texture	Saturated hydraulic conductivity [mm h^{-1}]
1	Fast and very fast	Sand	>61.0
2	Moderate fast	loamy sand, sandy loam	20.3-61.0
3	Moderate	loam, silty loam	5.1-20.3
4	Moderate low	sandy clay loam, clay loam	2.0-5.1
5	Slow	silty clay loam, sand clay	1.0-2.0
6	Very slow	silty clay, clay	<1.0

The stoniness factor (St-sub-factor) is related to the gravel content of the soil. The values that it can assume vary between 0 and 1, with 1 being soil without any stones and values close to 0 being for a high percentage of stones in the soil. The value of the St parameter is based on Panagos et al. 2014 (Table 4.4).

Table 4.4 St-sub-factor (Panagos et al. 2014).

	GRAVEL codes and their meaning	Gravel content (by volume)	Stoniness factor
0	no stones or gravel	0%	1
1	very few	stones≤10%	1
2	few	10%<stone<25%	0.74
3	frequent or many	25%≤stone<50%	0.332
4	very frequent, very many	stones≥50%	0.074

Panagos et al. (2014) underline three important aspects of the correct application of Equation 4.3:

1. The content of silt should not exceed the percentage of 70%. In case the percentage is over 70%, it is necessary to apply the upper limit.
2. As very fine sand, which is part of sand, is not an object of soil analysis, as usual, it is determined as 20% of the normal sand.
3. If the content of organic matter exceeds 4%, the upper limit is applied.

Panagos et al. (2014) have created a raster dataset with a resolution of 500 m x 500 m for Europe. For the rest of the world, such as for North Africa, it is possible to use the Gupta et al. (2024) raster dataset, an update of the Panagos et al. (2014) based on the soil erodibility formulated by Wischmeier and Smith (1978), which includes the saturated hydraulic conductivity. The resolution of this raster is 1 km x 1 km, and for the purpose of this work, the suggestion of the authors is followed, considering the Panagos et al. (2014) for European olive farms and the Gupta et al. (2024) dataset for Northern African olive farms.

4.2.2.1.3 Length-Slope factor

This factor considers topographic elements, i.e., the slope length (L) and slope steepness (S) following Wischmeier and Smith (1978). The calculation of the L parameter in the RUSLE model is based on the following equation:

Equation 4.5 L-factor equation (Renard et al. 1997).

$$L = \left(\frac{\lambda}{22.13}\right)^{m}$$

Where:

λ is the slope length [m]

22.13 is the slope length of a unit plot [m]

m is a parameter slope based on the following factors (Table 4.5):

Table 4.5 m-parameter based on the slope (Renard et al. 1997).

Slope [%]	**m**
>5	0.5
3-4	0.4
1-3	0.3
<1	0.2

For the steepness, two formulas (Equation 4.6) were developed based on the slope:

Equation 4.6 S-factor equations (Renard et al. 1997).

$$S = 10.8 \cdot \sin\theta + 0.03 \qquad if\ \theta < 0.09$$

$$S = 16.8 \cdot \sin\theta - 0.50 \qquad if\ \theta \geq 0.09$$

Where:

θ is the slope angle [°].

Panagos et al. (2015a) have produced a raster dataset for Europe in two versions, one continental with a resolution of 100 m x 100 m and one for each European country with a resolution of 25 m x 25 m. Considering the functional units chosen for the analysis, the dataset at a continental scale was chosen because it is in line with the FU of 1 ha. For the Northern Africa olive farms, the calculations were performed in a different way. As a first step, the Digital Elevation Model (DEM) was downloaded from the Shuttle Radar Topography Mission (SRTM), an international mission in which it is possible to obtain a DEM of the entire world (Farr and Kobrick 2000). The dataset obtained has a high resolution of 1 arcsec (~ 30 meter at the Equator). The software SAGA (Conrad et al. 2015) supported the calculation of the LS factor using the downloaded DEM for the specific area.

4.2.2.1.4 Cover factor

Cover factor refers to land management and cropping practices in an area (Wischmeier and Smith 1978). The only values that this factor can assume vary between 0 and 1. The C-factor can be derived from several methods from literature sources to GIS methods. Panagos et al. (2015b) produced a dataset specific for Europe with a resolution of 100 m x 100 m. For northern African olive farms, the C factor was calculated considering the Normalized Difference Vegetation Index (NDVI). Several equations can be found in literature, many of which are based on NDVI according to Xiong et al. (2023), which provide a complete overview concerning the methods and the equations for the calculation of the C factor) For the purpose of this work, the formula proposed by Van der Knijff et al. (1999, 2000) was chosen since it was specially made for the calculation of the C factor in a Mediterranean context and also applied by Manaouch et al. (2021) for an elaboration in Morocco. Therefore, the formula used is the following (Equation 4.7):

Equation 4.7 C factor formula for North African farms (Van der Knijff et al. 1999, 2000).

$$C\ factor = e^{\left[-\alpha \cdot \frac{NDVI}{(\beta - NDVI)}\right]}$$

Where:

α and β are parameters that describe the shape of the NDVI curve, and they are set respectively at 2 and 1.

NDVI is the Normalized Difference Vegetation Index proposed the first time by KRIEGLER (1969) and used the first time by Rouse et al. (1974). The most widely used index for assessing vegetation in a given area range from -1 to 1. There are several equations for its calculation based on the Space mission used to recover the data. In this work, data from Sentinel-2 Level 1C was used, in particular using bands 4 and 8 and the following formula (Addabbo et al. 2016) (Equation 4.8):

Equation 4.8 NDVI formula (Addabbo et al. 2016).

$$NDVI = \frac{(Band\ 8 - Band\ 4)}{(Band\ 8 + Band\ 4)}$$

4.2.2.1.5 Practice factor

Some practices that stop soil erosion are contour tillage, strip cropping on the contour, and terrace systems (Wischmeier and Smith 1978; Kuok et al. 2013). The practice factor looks at the results of these practices. The values that can be assumed vary between 0 and 1, in a similar way to the C factor. These practices affect only on a local scale: in this case, a value close to 1 is assumed for global erosion data while a value of 0 is used for considering forest and water environments. Similar to the C-factor, one can derive the P-factor from literature sources or GIS methods. Panagos et al. (2015c) produced a raster dataset for Europe with a resolution of 1 km x 1km. For northern African olive farms, the P factor was calculated considering the empirical method of Wenner and Kenya (1981) and consolidated by the formulation used by Lufafa et al. (2003) (Equation 4.9):

Equation 4.9 P factor formula for Northern African farms (Wenner and Kenya. Soil Conservation Extension Unit 1981; Lufafa et al. 2003).

$$P\ factor = 0.2 + 0.03 * slope$$

The slope required for the calculation is quantified through the Digital Elevation Model (DEM), which is the representation of the elevations or height in a geo-rectified point-based or area-based grid. In particular, the slope represents the first derivative of the DEM (Mukherjee et al. 2013), and this latter was downloaded from SRTM (Farr and Kobrick 2000). According to Clark et al. (2015), Fu et al. (2011), Khosrokhani and Pradhan (2013), and Li et al. (2017), the above-mentioned formulation is also applied to calculate water soil erosion by RUSLE.

The economic value of ER was calculated considering mainly two components:

- the provisioning of soil to restore the previous status.
- the levelling of soil that is mainly related to labour.

The provisioning of land for Europe was calculated considering the average of three countries (Spain, Italy, and Portugal) because a specific price list, for example, for each region of Italy, was not available. The prices found for these countries were similar, and the average is correctly in line with these prices.

The levelling, and in particular the labour to perform this operation, was calculated considering the cost of labour for each cultivation operation. Specifically, the representative cost for levelling for Italy was set considering the regional price lists in the case study areas because it allows obtaining the effective price of levelling, not considering the furniture of soil. In particular, the average of agronomic operations was calculated for each country, and the following proportion was applied (Equation 4.10):

Equation 4.10 Formula for establish the leveling costs of countries partner of SustainOlive (except Italy).

$$Operational\ cost\ (Italy){:}\ levelling\ cost\ (Italy) = Operational\ cost\ (country){:}\ x$$

Where:

- Operational (Italy) is the average cost of agronomic operations for Italy, considering the entire sample of olive farms.
- Levelling cost (Italy) is the representative cost of levelling in Italy.
- Operational cost (country) is the average of agronomic operations for the country, considering the entire sample of olive farms.
- x is the levelling cost of the country.

These calculations can also be applied to North African olive farms, considering a reduction of 30% for the furniture of soil. This percentage was obtained considering the average of inputs (e.g., manure) and the average of labour for levelling operations.

The price of ER is the sum of the furniture of soil and the levelling.

Table 4.6 summarises the databases used for the ER assessment.

Table 4.6. Databases used for the ER assessment.

Parameter	Database	Spatial coverage	Temporal coverage	Pixel size	Reference
R factor	Global Rainfall Erosivity Database	World	Data collection since 2013 for elaboration	30-arc-sec (~1kmx1km at the Equator)	Panagos et al. (2023)
K_{st} factor	High-Resolution Dataset for Europe	Europe	2014	500mx500m	Panagos et al. (2014)
K_{ksat} factor	Global soil erodibility	World	N.A.	1kmx1km	Gupta et al. (2024)
LS factor	LS factor (Slope Length and Steepness factor) for the EU	European Union (28 countries)	2015	100mx100 and 25mx25m	Panagos et al. (2015)
	N.A.	World	N.A.	1-arc-sec (~30mx30m at the Equator)	Software SAGA
C factor	Cover Management Factor for the EU	European Union (28 countries)	2010	100mx100	Borrelli and Panagos (2020)
	N.A.	World	N.A.	1-arc-sec (~30mx30m at the Equator)	Sentinel 2
P factor	Support Practices factor (P factor) for the EU	European Union	2010, 2016	30-arc-sec (~1kmx1km at the Equator)	Panagos et al. (2020)
	N.A.	World	N.A.	1-arc-sec (~30mx30m at the Equator)	Software SAGA and Lufafa et al. (2003)

Source: **authors' elaborations.**

4.2.2.2 Mechanical Filtration

Soil also has the function of filtering water. This can be done in two ways: through mechanical filtration or physicochemical filtration. Marks et al. (1989 defined the first method as the soil's ability to mechanically infiltrate a suspension. More specifically, the water permeability (k_f) represents the amount of water that can infiltrate into the soil in a given period of time. The first way, as defined by), is the capacity of the soil to be mechanically infiltrated by a suspension. More specifically, this is the amount of water that can be infiltrated into the soil and can be represented by the water permeability (k_f). Water permeability represents the amount of water that passes through the soil in a given period of time. This parameter depends on several factors, i.e., soil texture, pore distribution, soil type, sediment sequence, groundwater-surface, distance to groundwater, and land use type. The characterisation factor to express this ES is Infiltration Reduction Potential and the parameters for its calculation are soil texture, distance from the surface to groundwater, and surface sealing (Figure 4.2).

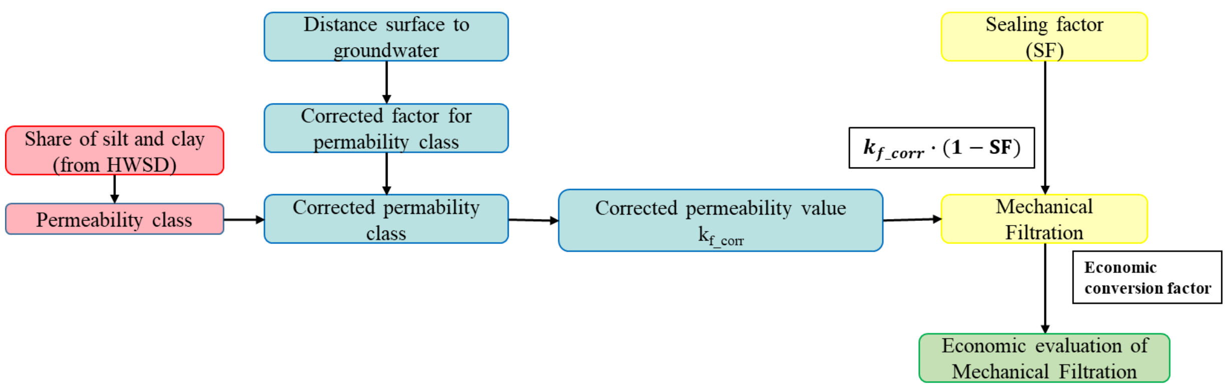


Figure 4.2 Steps to calculate Mechanical Filtration in LANCA model (elaboration based on Bos et al. (2016)).

As the first step for the determination of Mechanical Filtration is necessary to determine the permeability by classifying soil texture into permeability classes. This classification is carried out through Table 4.7:

Table 4.7 Correspondence between soil type, permeability class effective CEC of clay based on several literature sources (elaboration based on (Bos 2019)).

Clay [%]	Silt [%]	Soil type[4]	Permeability	CEC_{eff} clay

[4] Soil type is based on German soil classification

min	max	min	max		class	[cmol kg^{-1}]
65	100	0	35	T	1	38
45	65	0	15	Ts2	1	28
45	65	15	30	Tl	1	29
45	65	30	55	Tu2	1	29
35	45	0	15	Ts3	1	20
25	35	0	15	Ts4	1	15
25	45	15	30	Lts	2	19
35	45	30	50	Lt3	2	22
25	35	30	50	Lt2	2	17
30	45	50	65	Tu3	2	21
25	35	65	75	Tu4	2	18
17	25	0	15	St3	3	11
17	25	15	30	Ls4	3	12
17	25	30	40	Ls3	3	12
17	25	40	50	Ls2	3	13
17	30	50	65	Lu	3	15
17	25	65	82	Ut4	3	14
5	17	0	10	St2	3	6
12	17	10	40	Sl4	3	9
7	12	10	40	Sl3	3	6
7	17	40	50	Slu	3	9
7	17	50	65	Uls	3	9
12	17	65	82	Ut3	3	11
7	12	65	92	Ut2	3	9
5	7	10	20	Sl2	4	4
0	5	0	10	S	5	2
0	5	10	25	Su2	4	2
0	7	25	40	Su3	4	4
0	7	40	50	Su4	4	4

0	7	50	80	Us	4	5
0	7	80	100	U	4	6

Next, an average value of permeability was assigned considering the following table (Table 4.8):

Table 4.8 Correspondence between permeability class and permeability average value based on several literature sources (elaboration based on (Bos 2019)).

Permeability class	Permeability average [cm d^{-1}]
1	0.5
2	5.5
3	25
4	70
5	350

Permeability class was corrected through the distance of groundwater to the surface (Table 4.9):

Table 4.9 Correspondence between groundwater floor distance and permeability correction factor based on several literature sources (elaboration based on (Bos 2019)).

Groundwater floor distance [m]	Upper limit of groundwater floor distance class [m]	Correction factor permeability class
>30	100.000	+2
10-30	30	+1
0.8-10	10	0
<0.8	0.8	-1

The corrected permeability value was multiplied by a sealing factor subtracted by 1. The sealing factor was derived from the correspondence described in the Global Land Cover dataset. A formula derived from the steps described above is the following (Equation 4.11):

Equation 4.11 Mechanical Filtration formula based on the steps as described in Bos (2019)

$$MF = Corrected\ permeability \cdot (1 - sealing\ factor)$$

Table *4.10* shows the databases and the details used for the computation of this ES. According to Cao et al. (2015), the economic value of MF is found by taking into account the capital cost of the primary treatment in a wastewater treatment plant, which is based on Qasim (1998). This equation also considers the Water Purification Potential (WPP) that is quantified through the withdrawal used for agricultural purposes, and it can be determined using the Aquastat Database (FAO, 2024). In this paper, the equation was corrected by considering a scale factor based on the plot of the farm and the Utilised Agriculture Area (UAA) of the country because the method of Cao et al. (2015) was created for the calculation of the national level. The result of the economic conversion factor is calculated in \$ m^{-3} and it was used for the determination of the economic value of MF. The result represents the cost per cubic meter of water that has to be filtered through mechanical soil processes (considered as primary wastewater treatment).

Table 4.10. Databases used for the MF assessment.

<table>
<tr><th>Parameter</th><th>Database</th><th>Spatial coverage</th><th>Temporal coverage</th><th>Pixel size</th><th>Reference</th></tr>
<tr><td>Soil type</td><td>Harmonized World Soil Database v1.2 (HWSD)</td><td rowspan="3">World</td><td rowspan="2">N.A.</td><td rowspan="3">30-arc-sec (~1kmx1km at the Equator)</td><td>Nachtergaele et al. (2012)</td></tr>
<tr><td>Distance surface to groundwater</td><td>Global water table depth</td><td>Fan et al. (2022)</td></tr>
<tr><td>Sealing factor</td><td>Harmonisation, mosaicing and production of the Global Land Cover 2000 database</td><td>2000</td><td>Fritz et al. (2003)</td></tr>
</table>

Source: authors' elaborations.

4.2.2.3 Physicochemical Filtration

The other way that the soil has to filter water is through Physicochemical Filtration (PCF), which represents the amount of adsorbable cationic contaminants in the water. The first step in figuring out physicochemical filtration is to find the effective cation exchange capacity (CEC). This is affected by the type of soil, the amount of clay and silt, and other things.

Figure 4.3 defines the calculation steps for the PCF category.

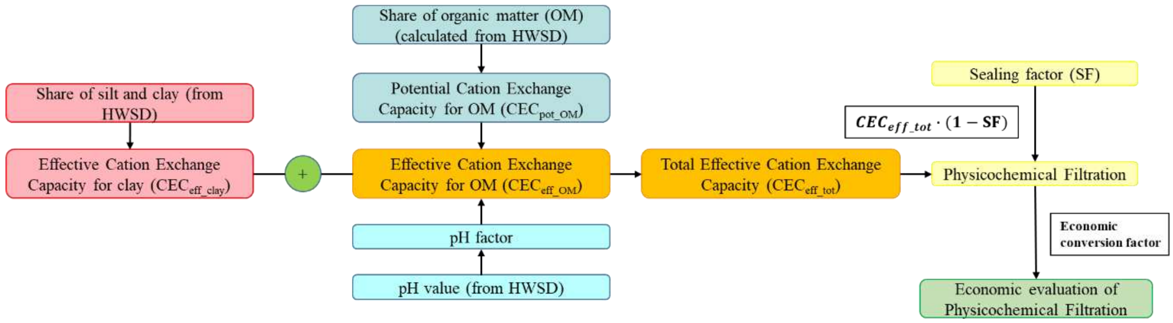


Figure 4.3 Steps to calculate Physicochemical Filtration in LANCA model (elaboration based on Bos et al. (2016)).

As reported in Table 4.11 and Table 4.12, the first step involves assigning the effective cation exchange capacity based on clay share while taking soil types into consideration.

Table 4.11 Correspondence between soil organic matter and potential cation exchange capacity of clay share based on several literature sources (elaboration based on (Bos 2019)).

Organic matter in soil [%]	CEC pot OM [cmol kg^{-1}]
0-1	0
1-2	3
2-4	7
4-8	15
8-15	25
15-30	50
30-100	110

Table 4.12 Correspondence between pH value and pH correction factor based on several literature sources (elaboration based on (Bos 2019)).

pH value	pH factor
<3.5	0.15
3.5-4.5	0.25
4.5-5.5	0.4
5.5-6.5	0.6

6.5-7.5	0.8
≥7.5	1

A formula derived from the steps described above is as follows (Equation 4.12):

Equation 4.12 Physicochemical Filtration formula based on the steps as described in Bos (2019).

$$PFC = \left(CEC_{eff_{clay}} + \left(CEC_{eff_{OM}} \cdot pH\ factor\right)\right) \cdot sealing\ factor$$

Table 4.13 provides all databases and details used for the computation of PCF. The economic value of the same ES is computed, as previously, considering Cao et al. (2015). In particular, to calculate the capital cost of the secondary and tertiary treatment in a wastewater treatment plant, based on Qasim (1998). This equation, as for MF, also considers WPP, determined in the same way, i.e., through the withdrawal used for agricultural purposes using the Aquastat Database. The equation was corrected by considering a scale factor based on the plot of the farm and the UAA of the country because the method of Cao et al. (2015) was created for the calculation at the national level. The result of the economic conversion factor is calculated in $ m^{-3} and it was used for the determination of the economic value of PCF. The result indicates the cost per cubic meter of water required for filtering through mechanical soil processes, which are considered primary wastewater treatment.

Table 4.13. Databases used for the PCF assessment.

Parameter	Database	Spatial coverage	Temporal coverage	Pixel size	Reference
Clay and silt share	Harmonized World Soil Database v1.2 (HWSD)	World	N.A.	30-arc-sec (~1kmx1km at the Equator)	Nachtergaele et al. (2012)
CEC_{eff_clay}					AD-HOC-ARBEITSGRUPPE BODENKUNDE (1996, 2005)
OM share					Nachtergaele et al. (2012)
pH					Nachtergaele et al. (2012)
pH factor					AD-HOC-ARBEITSGRUPPE BODENKUNDE (1996)
Bulk density					Nachtergaele et al. (2012)

Sealing factor	Harmonisation, mosaicing and production of the Global Land Cover 2000 database	2000	Fritz et al. (2003)

Source: authors' elaborations.

4.2.2.4 Soil Organic Carbon

A simple definition proposed by the Food and Agriculture Organization (FAO) identifies Soil organic carbon (SOC) as the carbon that remains in the soil following the partial decomposition of any substance created by living organisms (Triantakonstantis and Detsikas 2021; Fantin et al. 2022). The soil is composed of Soil Organic Matter (SOM), which is the set of organic compounds, and the main component of the SOM is the carbon, which takes the name of Soil Organic Carbon (SOC). The SOC plays an important role in the carbon cycle through the decomposition and storage processes in the soil. The carbon stored in the soil represents an important pool of this component: it is possible to find carbon up to 1 meter in depth, and this represents twice the amount found in the atmosphere and three times the amount found in vegetation (Smith 2012). The role of SOC in the global response to climate change is both vital and evident: the depletion of soil organic carbon (SOC) has a detrimental impact on soil health and agricultural production, while also exacerbating climate change. Decomposing SOM releases greenhouse gases (GHGs) into the atmosphere, which contributes to global warming (Paustian et al. 2019).

It's not a new idea to use outside methods and models to count SOC and include them in LCA. Several authors have used models outside of LCA to try to include them in order to get a bigger picture of changes in farming methods and climate change. Some models that have been used in this way are C-TOOL (Mcconkey et al. 2007; Hillier et al. 2012; Hamelin et al. 2012; Petersen et al. 2013; Knudsen et al. 2014), ICBM (Kimming et al. 2011b; Kimming et al. 2011a; Korsaeth et al. 2012), DAYCENT (Kim et al. 2009), APEX (Costello and Gautam 2018), EPIC (Sinistore et al. 2015), and RothC (Hillier et al. 2009; Cherubini and Ulgiati 2010; Nguyen et al. 2013; Yao et al. 2017. That being said, the RothC model was supposed to be used to figure out the amount of organic carbon in the soil when olives are grown in the 3rd draft of the Product Environmental Footprint Category Rule (PEFCR) for olive oil, but it isn't available yet because it hasn't been approved. In this paper, however, and like the other ESs that were looked at, a GIS method is used to figure out SOC that is part of LCA. This method is incorporated in the LANCA model as a new impact category, and it is based on IPCC guidelines. It takes into account three parameters:

- Land use and climatic variables
- Management practices
- Organic input

These parameters are calculated jointly with the SOC map that represents the Soil Organic Carbon in reference conditions. To be consistent with the method, the map is not elaborated in R and R Studio as made by the authors of the methods, but it is taken from FAO as a raster to be implemented directly in QGIS software. The equation for the calculation of this indicator is the following (Equation 4.13):

Equation 4.13 Soil Organic Carbon formula based on the steps as described in De Laurentiis et al. (2024).

$$SOC = SOC_{ref} \cdot F_{LU} \cdot F_{MG} \cdot F_{I}$$

Where:

SOC is the final soil organic carbon.

SOC_{ref} is the reference soil organic carbon.

F_{LU} is the parameter related to land use and climatic variables.

F_{MG} is the parameter related to the management practices.

F_{I} is the parameter related to organic input.

The framework for the computation of SOC is reported in the following figure:

4.2.3 *I-tree canopy tool*

I-tree canopy is part of the i-tree suite of software tools. These tools were developed by the United States Department for Agriculture (USDA) Forest Service and provide various forestry analyses both for urban and rural landscapes, as well as an assessment of environmental benefits, i.e., ESs. In particular, the i-Tree canopy tool was chosen in this study for the assessment of air purification potential, considering the removal rate of certain pollutants in the air.

Establishing a boundary for the area under examination is the first step in calculating air purification. The default loaded boundaries in the tool are for the USA and the UK. However, it is possible to load other areas through simple processing with GIS software, which allows a specific area to be selected even in another country (Figure 4.4).

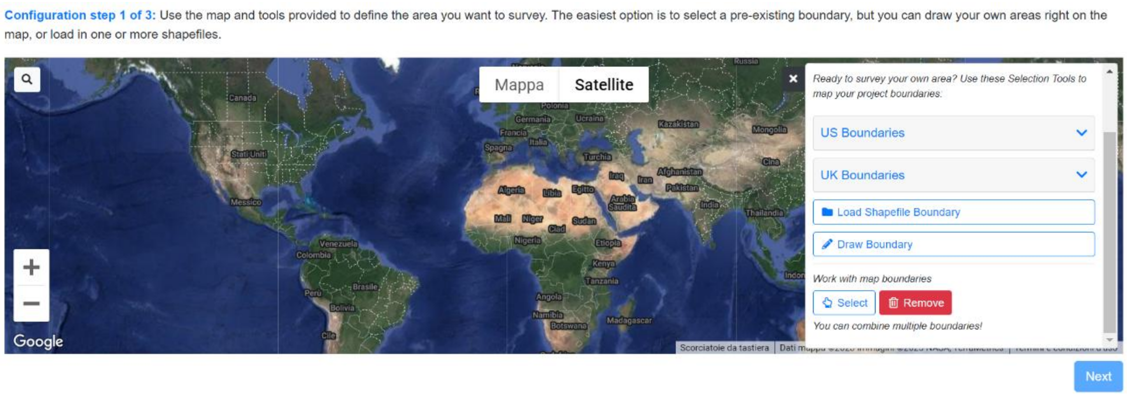
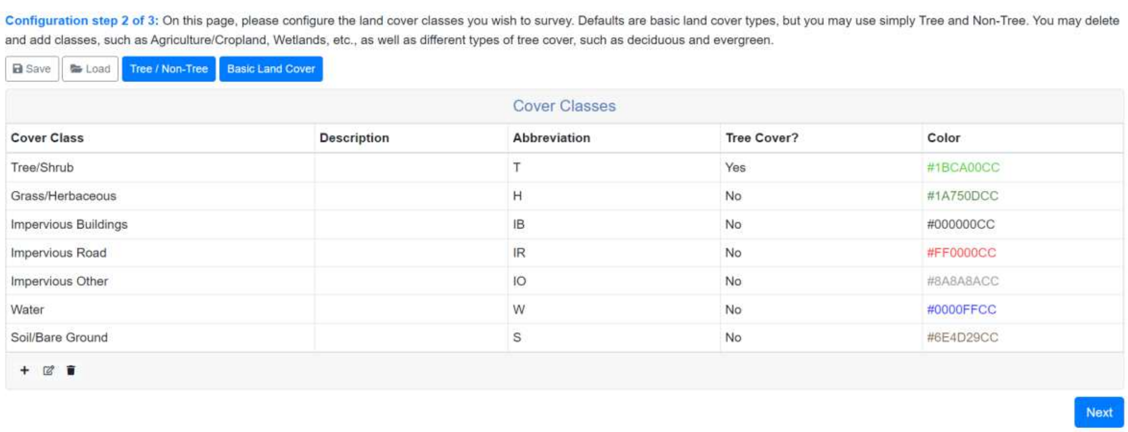


Figure 4.4 First step of evaluation with i-tree canopy tool.

Next, labels are selected with which to classify the land use in the area chosen in the previous step. There are two predefined cover classes (trees and non-trees), but there is the possibility of including other classes that discriminate against, for example, roads or buildings (Figure 4.5).



Cover Class	Description	Abbreviation	Tree Cover?	Color
Tree/Shrub		T	Yes	#1BCA00CC
Grass/Herbaceous		H	No	#1A750DCC
Impervious Buildings		IB	No	#000000CC
Impervious Road		IR	No	#FF0000CC
Impervious Other		IO	No	#8A8A8ACC
Water		W	No	#0000FFCC
Soil/Bare Ground		S	No	#6E4D29CC

Figure 4.5 Second step of evaluation with i-tree canopy tool.

Finally, there is the quantification of air purification by considering the removal rates of certain pollutants, in particular:

- CO
- NO_2
- O_3
- PM_{10}
- $PM_{2.5}$
- SO_2.

The removal rate can be expressed in metric units (g m^{-2} y^{-1}) or in English units (lbs ac^{-1} y^{-1}). The tool returns not only an environmental assessment of removal rates but also a monetary assessment of them. This can be expressed in the unit suitable for the study, from the US dollar to the euro. (Figure 4.6).

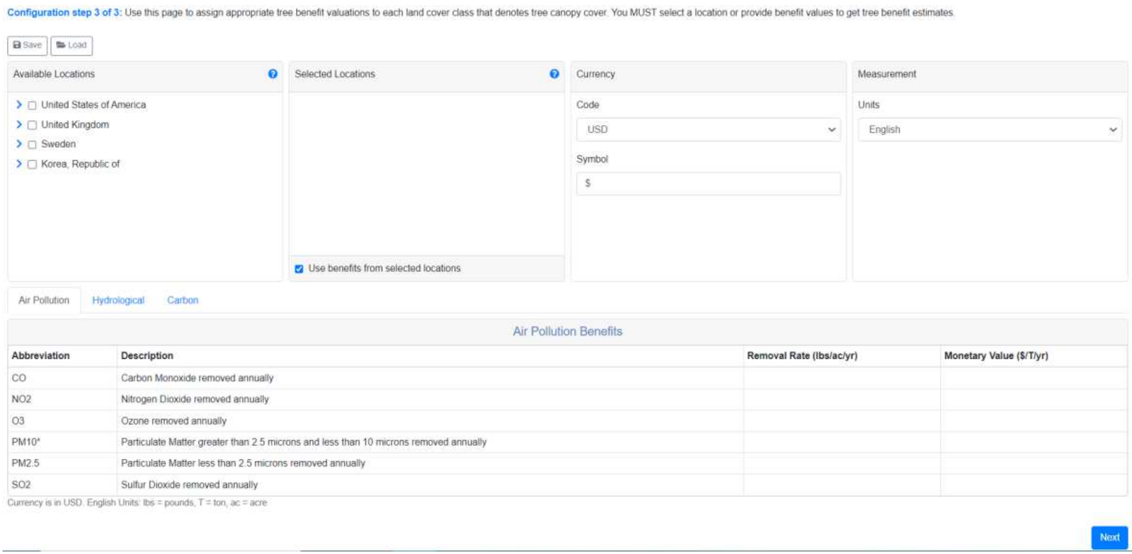


Figure 4.6 Third step of evaluation with i-tree canopy tool.

It is also possible to obtain the standard error considering the user's classification of the chosen points. In fact, through this classification, it is possible to examine if the points are correctly classified with the labels chosen in step 2. To make this statistical analysis, it is important to remember that a high number of chosen points reduce the standard error (SE). To give an example, consider an area with 1000 chosen points. These points are classified with the labels of step 2, and to simplify the example, they are considered as "Tree" and "No Tree". The former label considered 330 points, so it is possible to calculate SE with the following formula (Equation 4.14):

Equation 4.14 SE formula.

$$SE = \sqrt{\frac{pq}{N}}$$

Where:

n is the total number of points classified in a label, e.g. as "Tree"

N is the total number of the sampled points

p is the frequency of Tree-point, i.e., n/N=330/1000=0.33

q is the frequency of No-Tree-points, calculated as complementary to 1 of the probability, i.e., 1-0.33=0.67

In this example SE is:

$$SE = \sqrt{\frac{0.33 \cdot 0.67}{1000}} = 0.0149$$

The SE assumes the lowest value when p is low, and assumes the highest value when p is high (Figure 4.7).

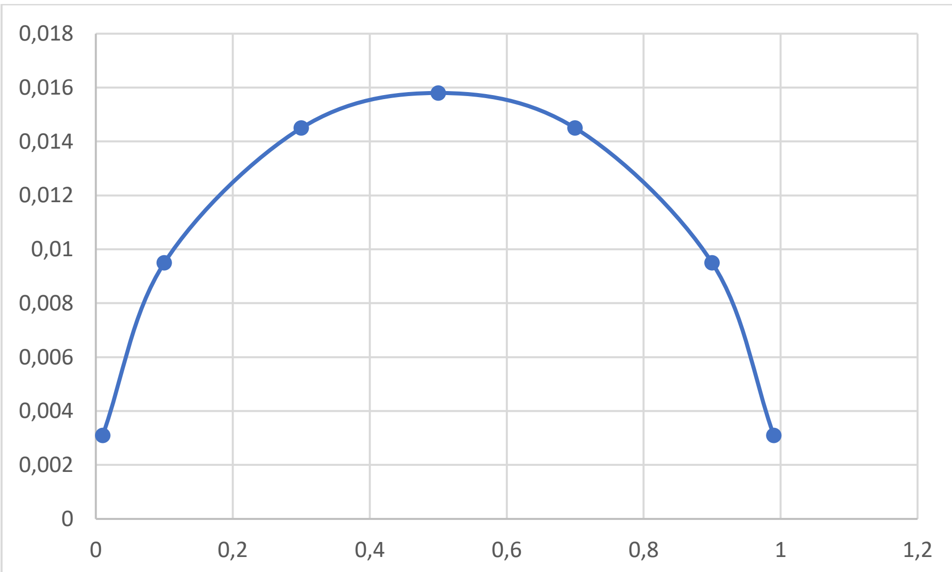


Figure 4.7 Relationship between sampled and classified points and SE.

With the value of SE, it is also possible to calculate a confidence interval, e.g., 95%.

If the number of sampled points is less than 10, it is necessary to apply a different formula because it is not relied upon for small sample dimensions. The formula for the calculation of SE for a number of sampled points less than 10 is (Equation 4.15):

Equation 4.15 SE if n is less than 10.

$$SE = \sqrt{\frac{n}{N}}$$

The strength of this method is the possibility to obtain, at the same time, environmental and economic results. The economic values obtained are based on the Willingness to Pay (WTP) and the cost of illness methodology and the BENMap CE tool.

4.2.4 *Ecotope formation*

Ecotope is an element of the landscaping area. A well – consolidated definition of an ecotope is defined as the smallest ecologically distinct feature of a landscape. Ecotope Formation (EF is an ES that shows how the landscape can build ecological structures by interacting with the biotic and abiotic components of the landscape. These structures are kept up and partly rebuilt. The ability of ecosystems to create stable, self-regulating units through the interaction of different biotic and abiotic factors is a measure of how well they work. The function of EF also encompasses species richness and structural diversity as parameters.

The calculation is performed, considering only the raster related to species richness and including the other parameters as constant values. Equation 4.16 provides the formula for EF.

Equation 4.16 Ecotope Formation formula based on Baitz (2002).

$$EF = MC + NC + D + AI$$

Where:

EF [dimensionless] is the ecotope formation.

MC [dimensionless] is the maturity class.

NC [dimensionless] is the naturalness class.

AI [dimensionless] is the anthropic influence.

D [dimensionless] is the diversity:

$$D = \frac{SR + SD}{2}$$

SR [dimensionless] is the species richness.

SD [dimensionless] is the structural diversity.

However, some olive groves have more than one variety to compensate for eventual loss related to pathogens, adverse climatic conditions, or adverse soil conditions. As a result, the variety has the potential to either increase or decrease the SR, as the survival of multiple varieties can support a greater diversity of species. The concept of variety requires the introduction of the standard deviation. Since the number of varieties is very low (at a maximum of four), it is not suitable for the application of the Gauss distribution to integrate the variety concept. Other distributions, such as the Poisson distribution or t-student distribution, can be applied, but they have some problems: the first one has the unit of time at the denominator (e.g., hospitalised patients day^{-1}) while the second one has a problem for the case of two varieties, which produce an infinite standard deviation.

To introduce variety and overcome the mentioned distributions, the beta distribution was chosen because it does not have the problems of the other distributions and can also be applied for a limited number of varieties.

The formula to quantify standard deviation in accordance with the beta distribution is the following:

$$\sigma = \sqrt{\frac{\alpha \cdot \beta}{(\alpha + \beta)^2 \cdot (\alpha + \beta + 1)}}$$

α and β are parameters that determine the shape of the beta distribution. In particular the formulas for their computation are:

$$\alpha = \mu \cdot \left(\frac{\mu \cdot (1 - \mu)}{s^2} - 1\right)$$

$$\beta = (1 - \mu) \cdot \left(\frac{\mu \cdot (1 - \mu)}{s^2} - 1\right)$$

Where:

μ is the sampling mean

s^2 is the population variance

The formula of diversity (D) becomes as follows:

$$D_{VAR} = \frac{\left[SR + \left(1 - \frac{1}{\rho}\right)\right] + SD}{2}$$

Where:

$$\rho = varieties \cdot (1 - \sigma)$$

The formula of D_{VAR} is applicable if there is more than one variety and if the population variance is different from 0 (the percentage of varieties should be different, not equal). If the population variance is 0, the ρ parameter coincides with the number of varieties:

$$\rho = varieties$$

In general:

$$\rho = \begin{cases} varieties \cdot (1 - \sigma) & if\ \sigma > 0\ and\ varieties > 1 \\ varieties & if\ \sigma = 0\ and\ varieties > 1 \end{cases}$$

$$D/D_{VAR} = \begin{cases} \dfrac{\left[SR + \left(1 - \frac{1}{\rho}\right)\right] + SD}{2} & if\ \sigma \geq 0\ and\ varieties > 1 \\ \dfrac{SR + SD}{2} & if\ \ varieties = 1 \end{cases}$$

The determination of the other parameters is exposed as follows.

The maturity class is a parameter regarding the succession of a vegetal community. The development of a succession starts from the initial stage until the climax stage, which is an equilibrium status with the environment. Two types of successions can be identified: the succession is natural if the stage is in line with the natural potentiality; else the succession is secondary when it is formed by human impacts, e.g., after a deforestation. Table 4.14 establishes the maturity class degrees.

Table 4.14 MC parameter.

MC degree	Description	Examples
1	Initial communities in the initial stage	(old ecosystems) arctic tundra, tropical rainforests, prairies, coral reefs, differentiated mixed forest, beech forest
2	Initial communities and short life additional communities	Alluvial forests, grazed forests, raised bogs
3	Long life addition communities	Forests, coppice forests, hay meadows, dry grassland
4	Stable communities (stable with external conditions)	Gardens, parks, vineyards
5	Climax communities	Field, turf

The naturalness class represents how far an ecosystem is far from natural conditions. It is related to plant association: this is natural when it is inherent to the ecological conditions, allowing for easy regeneration in response to perturbing influences and elements. Table 4.15 defines the degrees of naturalness class.

Table 4.15 Naturalness class parameter.

Value	Description	Examples
5	Close to natural	Rock, moor, water, tundra, high mountains, lightly thinned or grazed forests, dunes, moors, salt marshes
3	Semi-natural	Forestry, meadows, pastures, former quarries, heaths, dry grasslands, landscape parks
1	Far from natural	Forest monoculture, orchards, parks, farm gardens, intensively used meadows and pastures, unvegetated bodies of water, arable land, garden areas, sports turf, viticulture, loose rural development, intensively used fishponds
0	Artificial	Sports areas, landfills, turf, soil excavation areas, peri-urban development, ind. commercial areas with low sealing, campsites, railway facilities, sealed areas, technogenic areas, buildings, landfills with basic sealing, closed inner-city development, ind. commercial areas with high sealing

Diversity refers to the abundance of species and the complex structure of the connection. Ecosystems that exhibit high levels of diversity tend to be stable, indicating that the flow of energy and materials within the system is well-structured. Simultaneously, following a profound disturbance, it undergoes a more challenging process of regeneration, similar to that observed in less complex organised systems. Typically, climax ecosystems have greater biodiversity compared to pioneer or subsequent relationships. Diversity is related to two parameters: species richness (Table 4.16) and structural diversity (Table 4.17).

Table 4.16 SR parameter.

Number of species	Value
> 40	Initial communities in the initial stage
31 – 40	Initial communities and short life additional communities
21 – 30	Long life addition communities
11 – 20	Stable communities (stable with external conditions)
1 – 10	Climax communities

Table 4.17 SD parameter.

Height of vegetation unit	50 – 100%	25 – 50%	5 – 25%

High wood (10-20 m)	1	0.6	0.3
Low wood (< 10 m)	1	0.6	0.3
Bush (higher than 2 m)	1	0.6	0.3
Low bush (lower than 2 m)	0.5	0.3	0.2
Grass higher than 30 cm	1	0.6	0.3
Grass lower than 30 cm	0.5	0.3	0.2

The anthropic influence is related to the activities performed by humans. Examples of these activities are cultivation, drainage, road construction, irrigation, irregular depositions, etc. (Table 4.18).

Table 4.18 AI parameter.

Numeric values	Description	Proportion of vegetation damaged
5	No affected	0 %
4	Slightly affected	1 – 2 %
3	Compromised	2 – 5 %
2	Damaged	5 – 20 %
1	Seriously damaged	20 – 50 %
0	Extremely damaged	> 50 %

The EF can assume values ranging from 1.5 to 20. Not many studies of this ES are present in the scientific literature: the only one found in the scientific literature regards the evaluation of sodic soils in Hungary (Kürti and Ilona 2003). However, this ES represents an important function provided by the ecosystem, which also reflects the influence of human, and it also represents an ES provided by biodiversity.

Abbreviations

AI	Anthropic Influence
CEC	Cation Exchange Capacity
CF	Characterisation Factor
CICES	Common International Classification of Ecosystem Services
D	Diversity

DEM	Digital Elevation Model
EF	Ecotope Formation
ELU	Environmental Load Unit
ER	Erosion Regulation
ESs	Ecosystem Services
FAO	Food and Agriculture Organization
GHGs	Greenhouse Gases
GIS	Geographic Information System
GWR	Groundwater Regeneration
HANPP	Human appropriation net primary production
LANCA	Land Use Indicator Value Calculation for Life Cycle Assessment
LC	Life Cycle
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
MA	Millennium Ecosystem Assessment
MC	Maturity Class
MF	Mechanical Filtration
NC	Naturalness Class
NDVI	Normalized Difference Vegetation Index
PCF	Physicochemical Filtration
PEFCR	Product Environmental Footprint Category Rule
PRIMA	Partnership for Research and Innovation in the Mediterranean Area
RUSLE	Revised Universal Soil Loss Equation
SD	Structural Diversity
SE	Standard Error

SOC	Soil Organic Carbon
SOM	Soil Organic Matter
SR	Species Richness
SRTM	Shuttle Radar Topography Mission
STSs	Sustainable Technological Solutions
TEEB	The Economics of Ecosystems and Biodiversity
UAA	Utilised Agriculture Area
USDA	United States Department for Agriculture
USLE	Universal Soil Loss Equation
WPP	Water Purification Potential
WTP	Willingness to Pay

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Chapter 5.
RESULTS AND DISCUSSION: 1ST PART[5]

This chapter reports the first part of the results of a quantitative assessment of ecosystem services (ESs) in the SustainOlive pilot olive farms, building upon the methodologies described in Chapter 4. The evaluation of ESs within a Life Cycle perspective incorporates both environmental and economic outcomes. The research questions guiding this analysis and informing actionable recommendations for stakeholders are as follows:

1. Do sustainable agroecological practices in olive orchards significantly enhance the provision of ecosystem services compared to conventional practices?
2. Are Life Cycle Assessment methodologies—such as EPS2015 and LANCA—effective tools for quantifying both provisioning and regulating ecosystem services in Mediterranean olive orchards?
3. What is the added value of integrating ecosystem service assessments into LCA frameworks for improving the economic and environmental sustainability of olive farming in the Mediterranean region?

5.1 **Material and methods**

5.1.1 *The case studies*

As mentioned before, ESs were evaluated in the olive-growing context. In particular, twelve companies from each country in the SustainOlive project (Figure 5.1) were chosen to test ESs using methods that will be explained in the next paragraph. These companies used both STS and non-STS systems.

[5] This chapter is based on the following scientific article: Soldati, C., Falcone, G., Spada, E., Gulisano, G., De Luca A. I. Modelling and measuring variations in benefits and values of agro-ecosystem services using life-cycle based methods and tools: an application to Mediterranean olive growing contexts. Submitted to International Journal of Life Cycle Assessment. Status: under review. Personal contribution to the article: conceptualization, data curation, formal analysis, investigation, methodology and writing of the first draft. These activities were conducted in collaboration with the other co-author

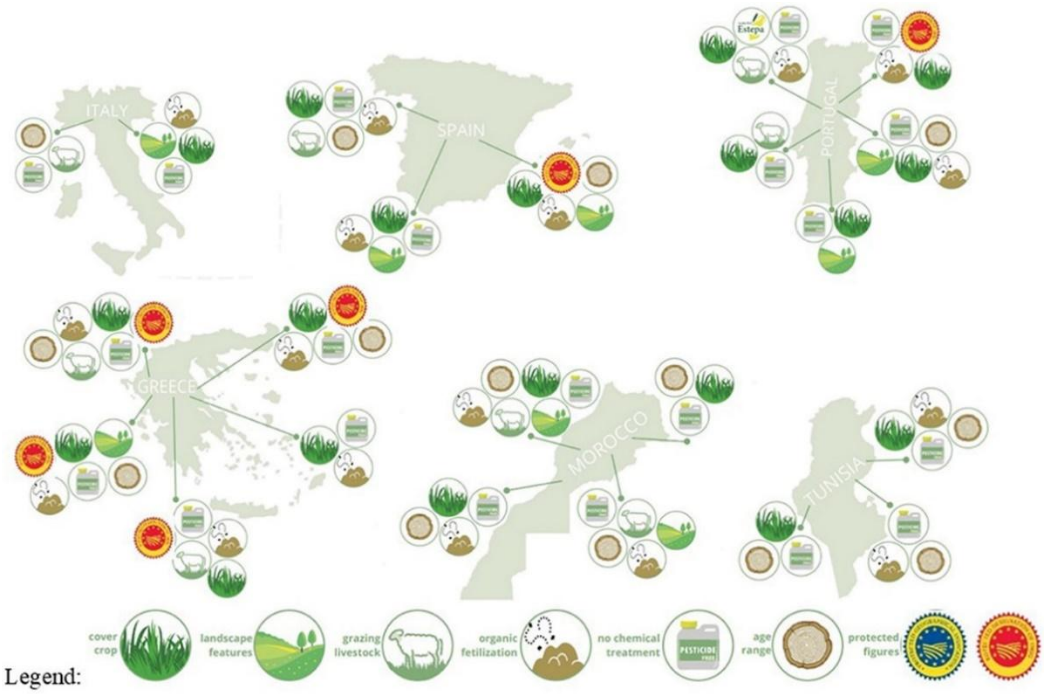

Figure 5.1. SustainOlive partner countries and STS applied (source: De Luca et al. (2023)).

The six STS farms are characterised by applying cropping methods based on agroecological principles (De Luca et al. 2023) and featuring several farming operations made singularly or combined. Among others, for example, the use of cover crops that could provide several benefits ranging from the decrease of soil loss to the improvement of infiltration; the livestock integration resulting in weed control, fertilisation, and maintenance of biodiversity; the management of tree pruning and other residues to promote and increase the level of nitrogen and carbon and create habitats for olive orchard biodiversity; and the organic fertilisation with effects on the nutrient cycle and retention and biodiversity protection. In particular, the olive farms considered in this work use the following agroecological practices:

N01_PU_STS_IT has an integrated conduct system, with the use of manure as fertiliser, the shredding of pruning and turf, and the non-use of pesticides.

N01_DE_STS_SP has an organic conduct system, the application of compost, pomace and olive mill residues, the shredding of turf, and pruning residues.

N02_CHR_STS_GR uses pomace and poultry manure.

N01_POR_STS_PT has an organic conduct system, with cover crops and the shredding of pruning.

N02_OUA_STS_MO burns pruning residues.

N02_SID_STS_TU has an organic conduct system that spreads vegetation water and uses pruning residues for animal feed.

Table 5.1 displays the characteristics of the farms analysed.

Table 5.1. Characteristics of STS and non-STS case studies analysed.

Olive farm	**N01_PU_STS_IT**	**N02_PU_nSTS_IT**	**N01_DE_STS_SP**	**N02_DE_nSTS_SP**	**N02_CHR_STS_GR**	**N05_CHR_nSTS_GR**	**N01_POR_STS_PT**	**N02_EVO_nSTS_PT**	**N02_OUA_STS_MO**	**N01_SPI_nSTS_MO**	**N02_SID_STS_TU**	**N05_SID_nSTS_TU**
Country	Italy		Spain		Greece		Portugal		Morocco		Tunisia	
Location	Apulia		Andalucía		Peloponnese		Évora		Tangeri-Tetouan-Al Hoceima		Sousse Governorate	
Production mode	Org./ Intg.	Intg.	Org.	Intg.	Org.	Conv.	Org.	Conv.	Org.	Org.	Conv./ Org.	Org.
Size of farm (ha)	18.45	17.60	255	30	2.2	1.3	470	220	5	2	24	136
Plant density (trees ha^{-1})	240	200	156	143	160	230	190	60	100		157	1000
Water management	Irr.		N.I.	Irr.	Irr.		N.I.		N.I.		Irr.	
Conduct system	Intv.	Trad.	Trad.		Trad.		Trad.		Trad.		Intv.	Trad.
Orography	Plain		Max 16°		Slope		Max 6%	Plain	Slope	Plain	Plain	
Mechanisation level	Medium-high		High		Medium-high		Medium	Low	Low		High	Medium
Yield (t ha^{-1})	10.5	9	3.4	5.7	3.3	2.2	2.5	3	3.1	0.7	4.7	4.3

Org. = Organic, Intg. = Integrated, Conv. = Conventional, Intv. = Intensive, Trad. = Traditional, Irr. = Irrigated, N.I. = Non-Irrigated.

Source: authors' elaborations.

5.1.2 *ESs assessment in Life Cycle perspective*

Life Cycle Methodologies are applied to evaluate ESs in olive orchards. Four phases make up the framework of LCA, which is potentially adaptable for Life Cycle Costing (LCC) and Social Life Cycle Assessment (S-LCA) (Figure 5.2).

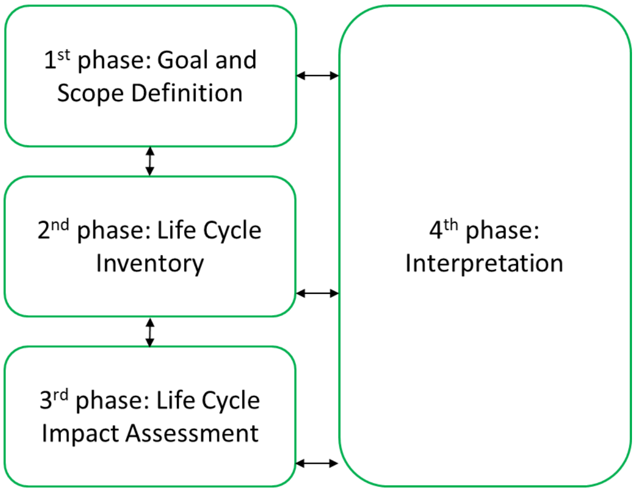

Figure 5.2. LCA framework (UNI EN ISO 14040:2021).

The first step is to define the goal and scope. During this step, specifics like the functional unit (FU) or the system boundaries must also be defined. The FU is a unit that measures the functions that have been identified in the system being studied. For this study, the FU chosen is one hectare (ha) of plot in 1 year since the aim is to analyse the ecosystemic functions of an agricultural process. Plot is a term that refers to characteristics of homogeneity in a part of a farm, so the choice of an FU that refers to a plot allows for the minimisation of the differences intra-farm. Also, because farming is a productive process, a mass-based FU (1 tonne of olives) was used to figure out how the product affected the ecosystem and to see how using different FUs affected the ecosystem.

The system boundaries are set from cradle to gate, considering the particular agriculture stage, because ESs are provided mainly in this phase. No allocation rules or cut-offs were defined. The time horizon of the analyses refers to the entire life cycle of the olive grove.

The Life Cycle Inventory (LCI), the second part of LCA, was made as a specific task in the SustainOlive project. It was made by gathering first-hand information from all operations in the foreground. A questionnaire was designed to collect data on, among other things: type of tillage, time of tillage and machinery used, fuel consumption, type of phytosanitary treatment, number of treatments, time of treatment and method, type of fertilisation, number of applications, time of application and method, type of irrigation, number of irrigations, amount of water per irrigation and irrigation method, method of harvesting, type of machinery or equipment used, type and quantity of fuel used. Data for background processes, such as the electric grid mix, are taken from the Ecoinvent v3.10 database. For this work, data collected were used for the computation of specific ESs, in particular for provisioning ESs. However, other data are needed for the quantification of other ESs and, therefore, some specific datasets whose characteristics are reported in the next paragraph.

The main focus of this paper is on the Life Cycle Impact Assessment (LCIA), which aims to use specific methods for measuring ESs. More information on this will be given in the next section. The last phase is the interpretation, in which some recommendations and suggestions are provided considering the results obtained.

5.1.3 *Characterisation models in the LCIA phase*

Based on the MA classification from 2005, Table 5.2 displays the ESs that are being evaluated in this chapter along with the LC methods that can be used to do so.

Table 5.2. Correspondence between ESs and Life Cycle-based methods.

ES Group	ES	Method
Provisioning	Crop growth capacity	EPS2015
	Production capacity for fruit and vegetables	
	Wood growth capacity	
	Fish and meat production capacity	
Regulating	Soil Erosion Resistance	LANCA
	Mechanical Filtration	
	Physicochemical Filtration	
	Ecotope Formation	Baitz 2002

For details on the specific procedures, databases used, and equations, please refer to Chapter 4.

5.2 **Results and discussion**

Table 5.3 shows the results expressed in 1 hectare · y^{-1} of STS and non-STS farms analysed and in 1 tonne of harvested olives · y^{-1} produced by the same farms, while Figure 5.3 shows the percentage results for provisioning services. The ESs evaluated with EPS2015 assume negative values when there is a positive impact (avoided impact) and positive values when there is a negative impact. Their interpretation requires the knowledge of the safeguard subjects and the relationship between the indicator “decreased production capacity of.” Overall, they perform better for European STS than non-STS olive farms. In Northern Africa, the results vary: Tunisian farms achieve similar outcomes, and Moroccan farms experience worse outcomes. The interpretation pertains to the amount of nitrogen used, specifically nitrogen oxide (NO). Indeed, EPS2015 considers a positive direct linear relationship between the quantity of nitrogen and the quantity of fish for the category “Fish and meat production capacity," considering NO, while for the category “wood growth capacity,” the positive direct linear relationship also involved NH3 emissions. Regulating services exhibit a wide range of values, making their interpretation more complex than provisioning services. Mechanical Filtration (MF) and Physicochemical Filtration (PCF) assume positive values because the water permeability and cation exchange capacity cannot be negative. Both have a high value, corresponding to better performance in water filtration through the soil. Most STS farms show better or the same results for MF and PCF; this can be interpreted considering the database used (HWSD). The database has a resolution of 1 km x 1 km, which does not allow for the discrimination of soil properties (e.g., soil texture, soil pH, etc.). This means that it is possible to insert most STS and non-STS farms into the same soil map unit, leading to an equal outcome. Economic evaluation can help to understand that, in general, STS olive farms have a higher economic value than non-STS farms.

Soil erosion resistance (ER) can be subject to different interpretations and trends. This service has negative values because it represents the soil’s resistance to the water erosion process. The maximum value that ER can assume is zero, representing a situation with no soil erosion, the lower the erosion resistance, the greater the soil loss through water erosion. In general, it is not possible to identify a trend for STS and non-STS olive farms (e.g., it is not possible to assert that STSs have a lower soil loss than non-STSs) because several parameters influence the final result, considering climatic, pedological, and management factors. Similar considerations can be made for the economic values related to the restoration costs. The pixels, which represent a portion of soil measuring 100m x 100m, or 1 ha, have a high ER (lower absolute value), indicating a low required restoration cost. The utilisation of this representation can prove beneficial in elucidating which section of the olive farm is more vulnerable to erosion, thus enabling the identification of cultivation techniques that could prove advantageous in maintaining this ecosystem service.

Table 5.3. Annual results of ESs assessed for each olive farm (FU = 1 ha · y^{-1}).

ESs	Unit of measure	N01_PU_STS_IT	N02_PU_nSTS_IT	N01_DE_STS_SP	N02_DE_nSTS_SP	N02_CHR_STS_GR	N05_CHR_nSTS_GR	N01_POR_STS_PT	N02_EVO_nSTS_PT	N02_OUA_STS_MO	N01_SPI_nSTS_MO	N02_SID_STS_TU	N06_SID_nSTS_TU
Crop	kg ha^{-1}	31.48	29.94	-6.18	12.25	-75.31	2.63	37.01	8.84	8.21	8.28	19.06	17.43
	€ ha^{-1}	6.92	6.59	-1.36	2.69	-16.57	0.58	8.14	1.95	1.81	1.82	4.19	3.83
Fruit and vegetables	kg ha^{-1}	1.68	1.94	-2.36	0.27	-12.42	-2.45	2.44	0.21	0.40	0.41	1.61	1.44
	€ ha^{-1}	0.66	0.76	-0.92	0.10	-4.84	-0.96	0.95	0.08	0.16	0.16	0.63	0.56
Wood	kg ha^{-1}	-55.90	-37.69	-122.44	-41.21	-564.63	-195.75	-42.93	-25.55	-12.93	-12.62	-14.02	-13.40
	€ ha^{-1}	-2.24	-1.51	-4.90	-1.65	-22.59	-7.83	-1.72	-1.02	-0.52	-0.50	-0.56	-0.54
Fish and meat	kg ha^{-1}	0.79	0.87	-0.84	0.18	-4.55	-0.79	1.04	0.12	0.18	0.18	0.67	0.60
	€ ha^{-1}	1.66	1.82	-1.76	0.37	-9.56	-1.66	2.19	0.25	0.38	0.38	1.40	1.26
ER	t ha^{-1} y^{-1}	-0.39 to -3.95	-0.65 to -0.85	-1.66 to -20.53	-2.47 to -10.83	-0.08 to -8.18	-3.16 to -5.38	-0.81 to -3.69	-0.76 to -1.56	-0.27 to -27.07	-0.08 to -4.80	0.27 to – 5.05	-0.38 to -2.15

	€ ha^{-1} y^{-1}	9.47 to 95.44	15.67 to 20.53	37.44 to 473.00	56.94 to 249.55	0.57 to 60.16	63.24 to 107.75	17.28 to 79.03	14.38 to 29.36	1.67 to 165.39	0.47 to 29.33	1.60 to 30.44	2.29 to 13.04
MF	m^{3} ha^{-1} y^{-1}	$2.43\cdot10^{6}$	$8.67\cdot10^{5}$	$2.43\cdot10^{6}$-$1.21\cdot10^{7}$	$2.43\cdot10^{6}$-$1.21\cdot10^{7}$	$9.13\cdot10^{5}$-$2.56\cdot10^{6}$	$1.28\cdot10^{7}$-$2.56\cdot10^{6}$	$2.43\cdot10^{6}$-$1.21\cdot10^{7}$	$2.43\cdot10^{6}$	$1.91\cdot10^{5}$	$1.91\cdot10^{5}$	$1.28\cdot10^{7}$	$1.28\cdot10^{7}$
	€ ha^{-1} y^{-1}	7.98	2.43	6.83-65.70	4.60-33.04	6.65-19.67	20.06-122.84	29.25 – 204.97	23.80	0.82	0.77	243.3	120.92
PCF	mol ha^{-1}	$1.92\cdot10^{6}$	$1.92\cdot10^{6}$	$2.49\cdot10^{6}$-$2.68\cdot10^{6}$	$2.49\cdot10^{6}$	$2.12\cdot10^{6}$	$2.12\cdot10^{6}$	$1.84\cdot10^{6}$-$1.85\cdot10^{6}$	$9.06\cdot10^{5}$	$1.82\cdot10^{6}$	$1.82\cdot10^{6}$	$1.60\cdot10^{6}$	$1.60\cdot10^{6}$
	€ ha^{-1}	33.27	22.35	53.80-57.86-126.99	27.53-51.19	24.90-28.67	81.89-125.31	125.77 – 229.53	38.13	35.51	30.62	304.71	132.69

Source: authors' elaborations.

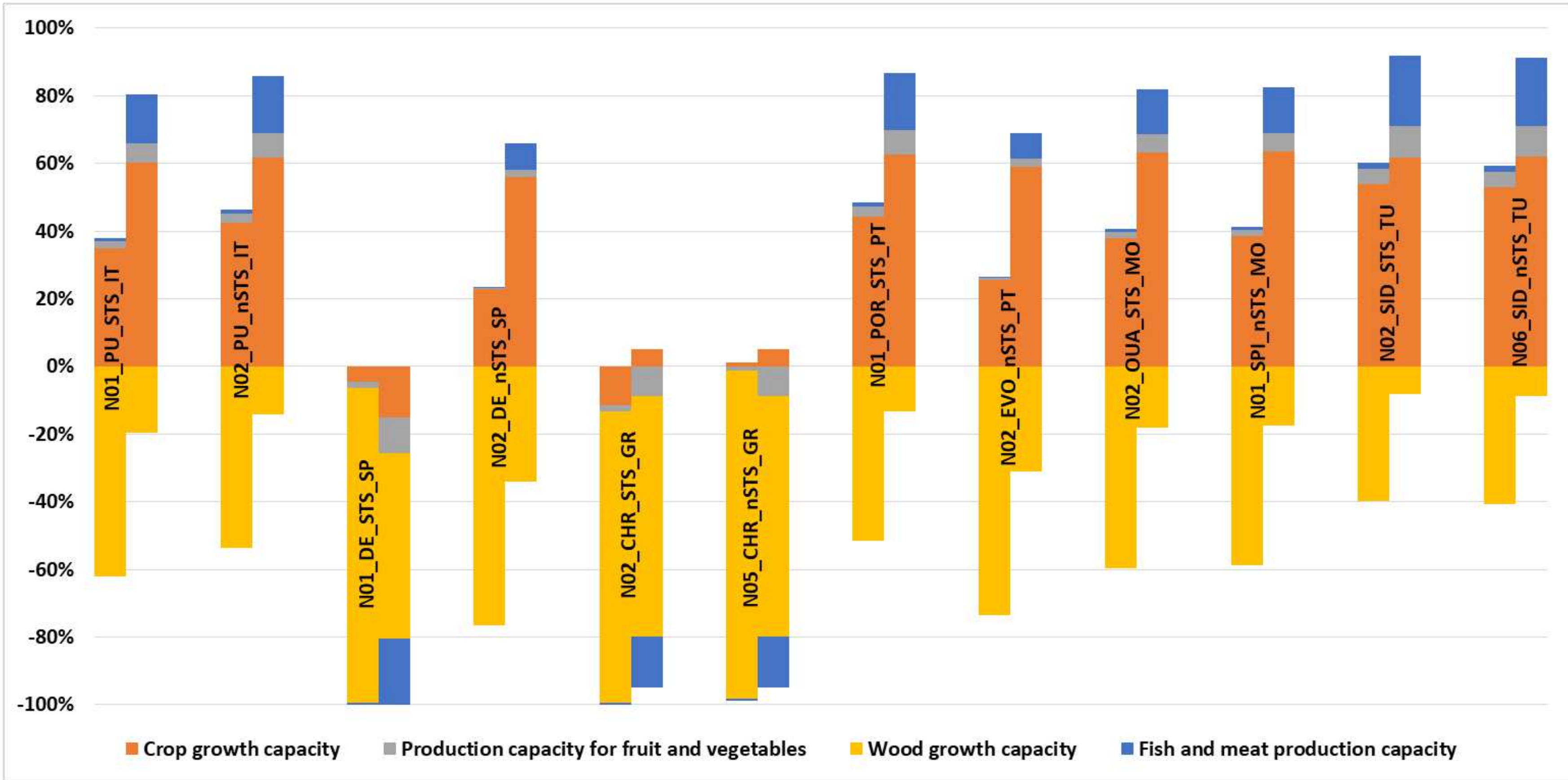

Figure 5.3 Percentage results related to provisioning services in the FU = 1 ha · y^{-1}. (Source: authors' elaborations).

Table 5.4. Annual results of ESs assessed for each olive farm (FU = 1 tonne of harvested olives · y^{-1}).

ESs	**Unit of measure**	**N01_PU_STS_IT**	**N02_PU_nSTS_IT**	**N01_DE_STS_SP**	**N02_DE_nSTS_SP**	**N02_CHR_STS_GR**	**N05_CHR_nSTS_GR**	**N01_POR_STS_PT**	**N02_EVO_nSTS_PT**	**N02_OUA_STS_MO**	**N01_SPI_nSTS_MO**	**N02_SID_STS_TU**	**N06_SID_nSTS_TU**
Crop	kg ha^{-1}	3.59	3.99	-2.32	4.40	-27.46	1.43	20.04	3.68	3.78	15.52	4.70	4.77
	€ ha^{-1}	0.79	0.88	-0.51	0.97	-6.04	0.31	4.41	0.81	0.83	3.41	1.03	1.05
Fruit and vegetables	kg ha^{-1}	0.19	0.26	-0.89	0.10	-4.53	-1.33	1.32	0.09	0.18	0.77	0.40	0.39
	€ ha^{-1}	0.07	0.10	-0.35	0.04	-1.77	-0.52	0.51	0.03	0.07	0.30	0.15	0.15
Wood	kg ha^{-1}	-6.38	-5.02	-45.96	-14.80	-205.90	-106.45	-23.24	-10.63	-5.95	-23.66	-3.46	-3.67
	€ ha^{-1}	-0.26	-0.20	-1.84	-0.59	-8.24	-4.26	-0.93	-0.43	-0.24	-0.95	-0.14	-0.15
Fish and meat	kg ha^{-1}	0.09	0.12	-0.31	0.06	-1.66	-0.43	0.56	0.05	0.08	0.34	0.17	0.16
	€ ha^{-1}	0.19	0.24	-0.66	0.13	-3.49	-0.90	1.19	0.10	0.17	0.72	0.35	0.35
ER	t ha^{-1} y^{-1}	-0.45 to -0.04	-0.11 to -0.09	-0.56 to -6.98	-0.57 to -2.49	-0.03 to -3.00	-1.72 to -2.92	-0.44 to -2.02	-0.32 to -0.65	-0.13 to -12.46	-0.14 to -9.00	0.07 to -1.26	-0.10 to -0.59

	€ ha^{-1} y^{-1}	1.08 to 10.90	2.09 to 2.73	12.73 to 160.81	13.07 to 57.29	0.57 to 60.16	34.39 to 58.59	9.46 to 43.27	6.02 to 12.28	0.77 to 76.12	0.88 to 55.00	0.40 to 7.60	0.63 to 3.60
MF	m^3 ha^{-1} y^{-1}	$2.77 \cdot 10^5$	$1.16 \cdot 10^5$	$8.26 \cdot 10^5$-$4.13 \cdot 10^6$	$5.58 \cdot 10^5$-$2.79 \cdot 10^6$	$9.39 \cdot 10^5$-$3.35 \cdot 10^5$	$1.39 \cdot 10^6$-$6.95 \cdot 10^6$	$1.33 \cdot 10^6$-$6.65 \cdot 10^6$	$1.02 \cdot 10^6$	$8.79 \cdot 10^4$	$3.58 \cdot 10^5$	$3.19 \cdot 10^6$	$3.52 \cdot 10^6$
	€ ha^{-1} y^{-1}	0.61	0.32	2.32-22.34	1.05-7.59	2.44-7.22	10.91-66.80	16.01-112.22	9.95	0.38	1.45	60.75	33.32
PCF	mol m^{-2}	$2.19 \cdot 10^5$	$2.55 \cdot 10^5$	$8.47 \cdot 10^5$-$9.11 \cdot 10^5$	$5.72 \cdot 10^5$	$7.77 \cdot 10^5$	$1.15 \cdot 10^6$	$1.00 \cdot 10^6$-$1.01 \cdot 10^6$	$3.79 \cdot 10^5$	$8.39 \cdot 10^5$	$3.42 \cdot 10^6$	$3.98 \cdot 10^5$	$4.40 \cdot 10^5$
	€ ha^{-1}	30.82	3.69	18.29-19.67-43.17	6.32-11.75	28.67-24.90	44.53-68.14	68.86-125.68	15.95	16.35	57.42	76.08	36.56

Source: authors' elaborations.

Table 5.4 shows the results expressed in tonnes of harvested olives per year. Even though the average annual production is an important parameter in this case, the results are similar to those expressed in hectares. Indeed, when the annual production is lower than 1 t, the values are higher than those expressed in hectares. As mentioned in the presentation of the method, the provisioning services computed with EPS (2015d) translate the indicator "Decreasing production capacity," and the results are similar and comparable between countries. Basically, STS farms show better performances for all provisioning services; this can be explained in relation to the pathway of each provisioning (Steen and Palander 2016). For example, a farm that uses a higher quantity of fertiliser than a farm that uses a lower quantity will show a significant impact for all associated categories, such as for fish and meat production, which is related to the use of fertiliser as it is linked to acidification. A similar trend can also be seen in the results expressed in FU in tonnes.

Erosion Resistance (ER) shows "ambiguous" results considering the minimum and maximum values. Note that even with the best management, soil erosion is still less than if only the R, K_{st} (or K_{ksat}), and LS factors were used to estimate soil erosion. This fits with what Estrada-Carmona et al. (2016) found: the topographic and coverage factors (LS and C, respectively) are the most important factors in figuring out soil erosion. Changing these factors can help lower soil erosion. In addition, Rava (2020) highlighted that at the European level, some RUSLE parameters interact with them. This is crucial because changing one can change the results, including decrease soil erosion.

The results of mechanical filtration (MF) and physicochemical filtration (PCF) depend on the texture of the soil and the depth of the water table. In general, the results are better for STS farms, and these trends are confirmed for both FUs. It is important to highlight that MF and PCF show a unitary correlation even if they describe different processes (De Laurentiis et al. 2019).

For all *regulating services*, the results show a correlation with the area of the plots: this is important because taking a similar area between two farms, not only the conditions but also the results of, for example, erosion resistance can increase or decrease. This is the case, for example, of the two Italian farms in which the size plot differs significantly from each other: taking in N02_PU_nSTS_IT, an area similar to N01_PU_STS_IT, the variability increases.

This explanation works for some MF and PCF results because they are based on a bigger HWSD soil map unit, and the results can be very similar, especially when they are given in hectares. However, the plot size partially influences the economic results, as the correction factor typically hinges on it. The other parameter that influences the result is the soil properties that can produce a very high outcome due to, for example, the depth of the groundwater.

Of all regulating ESs, the most important is ER because it relates to climate change, one of the most important environmental issues of this millennium. According to JRC (2008), the extreme events that characterised Italy and the Mediterranean region, i.e., cloudbursts, can produce an erosion of 20-40 t ha^{-1} after only a single event (Tóth et al. 2008). In addition, the authors estimated that soil erosion results in a cost estimated between €0.7 and 14.0 billion for the EU-27. Furthermore, soil erosion is responsible for the reduction of soil organic matter (SOM), which is the most important element for soil fertility, due to the removal of the topsoil where it is concentrated. Several practices to reduce soil erosion can be applied for different reasons. For example, using cover crops decreases soil erosion by decreasing the C factor because bare soil is more susceptible to soil erosion than covered soil (Panagos et al. 2015b). It has been shown that temporary ditches reduce soil erosion by 67% compared to areas that don't have them (Francaviglia and Neri 2020). This is because they lower the topographic factor (LS factor) of the RUSLE equation. Another measure to reduce soil erosion is the use of external organic matter inputs that act on the K_{st} factor. For example, Gholamahmadi et al. (2023) have found that the use of biochar has reduced soil erosion by 16%, while Gholami et al. (2016) argue that the use of manure results in a decrease between 15% and 65% compared to non-manured.

Mechanical filtration and physicochemical filtration can also be linked with the soil erosion resistance and the measures for its reduction: the increasing of SOM (that affects the K_{st} factor in the RUSLE equation), like biochar can improve the soil permeability to water (Wong et al. 2022; Chen et al. 2023). This is particularly important for avoiding or reducing the risk of flooding caused by extreme weather events (e.g., the flood of the Emilia-Romagna region in 2023). SOM can also improve Cation Exchange Capacity (CEC) for the PCF (e.g., Zhang et al. 2015; Frank et al. 2020; Yao et al. 2022).

Summarising, some important considerations are related to the need for an in-depth understanding of the area and the crops grown because not all solutions mentioned before could apply to all soils. Indeed, basic soil is not suitable, for example, for the liming practice because it increases the pH of the soil significantly. Furthermore, the use of highly accurate databases is crucial, even though soil heterogeneity can lead to regionalised outcomes from GIS. This can lead to the creation of more accurate datasets for GIS elaboration, thereby reducing the need for raster resampling.

5.3 **Conclusion**

This work allowed an evaluation of ESs by combining LCA and the GIS methodologies. The use of EPS2015d has allowed the calculation of provisioning services, one of the groups of ESs, following MA's classification. The most important limitation of this LCIA method is related to the difficulty of obtaining site-specific results because it has generic characterisation factors. The introduction's limitations make the development of regionalised CFs and subsequent outcomes crucial. However, the strength of this method is the possibility of obtaining economic results simultaneously with environmental ones.

Instead, GIS techniques have made it possible to measure regulating service, which is the other group of MA's classification that this work looks at. Conversely, for EPS, the use of GIS has enabled the computation of regionalised results. However, the most important limitation of this method is the use of rasters with a low resolution, e.g., 1 km x 1 km. This requires an oversampling to reduce the resolution in the most common and practical resolution, i.e., 100 m x 100 m, which represents a result expressed per hectare. The problem is that the original raster is not very defined, and the oversampling produces more pixels but with the same information. The creation of rasters with high resolution could definitely reduce the problem.

The exclusion of some factors is another weakness of GIS methods, particularly the LANCA method. To give some examples, MF and PCF do not take into account some management parameters, like soil compaction through the use of heavy machines or fertilisers. These things are important because they can change the properties of the soil and the ESs that are connected to it, such as ER, MF, and PCF (Horn et al. 1995; Czarnecki and Duering 2014; Songül Gürsoy 2021).

From a methodological point of view, the methods presented can be improved by filling the gap (e.g., regionalised results in EPS) or minimising some operations that can compromise outcomes (e.g., resampling rasters). Furthermore, the creation of rasters with future projections, for example, the future value of rainfall erosivity (R factor) related to an increase in the strength of precipitation or the increase of cover crops in response to political measures that contribute to reducing soil erosion, can help to understand the future impacts in the soil.

Because the ecosystem services shown are closely connected to the farming methods used in olive groves, it is very important to measure them on both an environmental and economic level in order to make policy decisions. This is because agriculture serves many purposes indirectly. The STS management system was specifically designed to adhere to the agroecological measures mandated, such as those outlined, for example, in the Common Agricultural Policy (CAP) at the European level for the period of 2023-2027. Subsequently, each country transposed the general objectives of the CAP into specific ones at the national level. For example, Italy has transformed agroecological payments into ecosystems by implementing a series of procedures. The agroecological practices in this study occur under the Eco-scheme 2, which falls under the jurisdiction of the Ministry of Agriculture, Food Sovereignty and Forests (MASAF-Ministry of

Agriculture, Food Sovereignty and Forests 2023) for the year 2023. Presently, the expense of upholding the assets according to this ecological program amounts to €120 per hectare.

Nevertheless, the economic advantage of adopting an eco-scheme frequently fails to justify the associated expenses, as evidenced by the grievances voiced by farmers during the winter of 2023. Therefore, the suggested assessment is highly relevant for both the scientific community and government policymakers. By employing economic quantification, specific agroecological activities, such as chopping, can be measured in ways that facilitate the design of compliance – based incentives or premiums.

Further results concerning the other olive farms of SustainOlive can be found in the appendix at the end of this document.

Abbreviations

CAP	Common Agricultural Policy
CICES	Common International Classification of Ecosystem Services
EPS	Environmental Priority Strategy in product design
ER	Erosion Resistance
ESs	Ecosystem services
FU	Functional Unit
GIS	Geographic Information System
HWSD	Harmonised World Soil Database
LC	Life cycle
LCA	Life cycle assessment
LCC	Life cycle costing
LCI	Life cycle inventory
MA	Millennium ecosystem assessment
MF	Mechanical filtration
NDVI	Normalized Difference Vegetation Index
PCF	Physicochemical filtration
RUSLE	Revised Universal Soil Loss Equation

S-LCA	Social life cycle assessment
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
STS	Sustainable Technological System
TEEB	The Economics of Ecosystems and Biodiversity
UAA	Utilized Agriculture Area
WPP	Water Purification Potential

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Chapter 6.
RESULTS AND DISCUSSION: 2ND PART

The second part of the results of a quantitative assessment of ecosystem services (ESs) in the SustainOlive pilot olive farms are presented in this chapter. The methods used were similar to those in Chapter 4. The evaluation of ESs within a Life Cycle perspective incorporates both environmental and economic outcomes. The research questions guiding this analysis and informing actionable recommendations for stakeholders are as follows:

The research questions of the study are about:

RQ1: Does agroecology contribute to preservation and fostering of ecosystems services?

RQ2: Does i-Tree Canopy a suitable tool to integrate the assessment of ES in Life Cycle methodologies?

RQ3: Which is the asset of the integration of ES assessment in LCA for agricultural economics and politics?

6.1 **Material and methods**

6.1.1 *The case studies*

The olive farms assessed are the same of those evaluated in the previous chapter. For further details concerning the specific characteristics of the single farm, please refer to Chapter 5.

6.1.2 *ESs assessment in Life Cycle perspective*

The framework for the evaluation of ESs in Life Cycle perspective, i.e., which FUs are chosen, is the same as in the previous chapter. For further details concerning the specific characteristics of the single farm, please refer to Chapter 5.

6.1.3 *Characterisation models in the LCIA phase*

Based on the MA classification from 2005, Table 6.1 displays the ESs that are being evaluated in this chapter along with the LC methods that can be used to do so.

Table 6.1. Correspondence between ESs and Life Cycle-based methods.

ES Group	ES	Method
Regulating	Air purification	i-tree canopy
	Soil Organic Carbon	LANCA
Supporting	Ecotope Formation	Baitz 2002

For details on the specific procedures, databases used, and equations, please refer to Chapter 4.

This study is framed within the SustainOlive project, an international research project funded by the Prima Foundation, aimed at promoting sustainability in olive growing in the Mediterranean basin. The project involved six countries (Italy, Spain, Portugal, Greece, Morocco, and Tunisia) and evaluated two types of farms: sustainable technological solutions (STS) olive farms, which adopt agro-ecological practices such as cover cropping and livestock integration, and non-sustainable technological systems (non-STS) olive farms, which rely on conventional practices such as synthetic pesticides, chemical fertilisers, and deep tillage. Twelve olive farms were analysed, six of which belonged to the STSs group, applying agroecological practices such as the management of pruning residues to recycle organic matter, the use of cover crops, the integration of livestock in the orchards, and the use of manure and organic fertilisers (De Luca et al., 2023).

In this chapter, three ESs are assessed: air purification, ecotope formation, and soil organic carbon. Air purification is the process by which natural elements such as trees, shrubs, and other vegetation remove pollutants from the atmosphere (Nowak et al., 2006). In this study, air purification is evaluated using i-Tree Canopy, a tool from the i-Tree suite developed by the USDA Forest Service. It estimates the removal rates of six major air pollutants: carbon monoxide (CO), sulphur dioxide (SO_2), nitrogen dioxide (NO_2), ozone (O_3), particulate matter 2.5 ($PM_{2.5}$), and particulate matter 10 (PM_{10}). Four steps are necessary to be implemented:

1. The definition of a system boundary (e.g., the farm boundary)
2. The selection of cover classes (i.e., covered or non-covered by trees)
3. The selection of benefits to use for the assessment
4. The survey points to improving the results.

One of the strengths of using i-Tree Canopy is the ability to obtain both environmental and economic outcomes. Hirabayashi (2014) bases the economic valuation on the BenMap database and scientific literature. The BenMap is a tool that is based on two economic methods: Cost of Illness and Wiliness to Pay. The Cost of Illness means the financial burdens incurred by an individual due to hospital admissions, emergency department visits, and other consequences related to air pollution; it encompasses medical expenses and lost wages, excluding the subjective valuation of pain and suffering associated with the incident. On the other hand, Willingness to Pay (WTP) includes not only the direct expenses previously mentioned but also the value individuals attach to pain and suffering, loss of enjoyment, and free time. The scientific works used depend on the study area: urban values were calculated utilising national median externality values (Murray et al., 1994), which were adjusted to 2010 figures via the producer price index (U.S. Department of Labor Bureau of Labor Statistics, 2024). On the other hand, rural values were obtained by modifying urban values based on the ratio of rural to urban values for all BenMAP pollutants. To get the multipliers for removal quantity and monetary value, the total amount of trees that were cut down and their monetary value were divided by the total number of trees in each rural and urban area. The multipliers for the entire region were calculated by combining rural and urban regions within the region. In i-Tree Canopy, the annual quantity of air pollutants eliminated by trees and the corresponding monetary value can be determined by multiplying the tree cover in the specified area by multipliers derived from county-level data in the United States. For nations beyond the United States, county multipliers obtained from the total removal quantity, monetary valuation, and tree cover in the United States may be utilised. In this study, the amount of air pollution was calculated by looking at the annual value and the whole life cycle for the constant phase. The non-productive and increasing production phases were left out because olive trees can't remove pollutants or can only do so.

Table 6.2 and Table 6.3 report, respectively, the benefits used and the inputs necessary for the calculation of the air purification:

Table 6.2 Benefits used for the air purification (source: i-tree canopy).

	$g\ m^{-2}\ y^{-1}$	**$€\ kg^{-1}\ y^{-1}$**
CO	0.122	24.80
NO_2	0.675	2.79
O_3	6.019	18.60
PM10	2.388	116.56
PM2.5	0.237	678.02
SO_2	0.211	0.83

Table 6.3 Inputs necessary for air purification (source: i-tree canopy).

	AREA	GRASS COVER [%]	TREE COVER [%]	GRASS COVER AREA	TREE COVER AREA	LIFE CYCLE [Y]
N01_PU_STS_IT	7.52 ha	21.10	54.60	1.58672 ha	4.10592 ha	135
N02_PU_NSTS_IT	12259.38 m^2	27	49.20	3310.033 m^2	6031.615 m^2	135
N01_DE_STS_SP	16.15 ha	0.10	29.40	0.01615 ha	4.7481 ha	135
N02_DE_NSTS_SP	37.07 ha	0.10	18.30	0.03707 ha	6.78381 ha	65
N01_POR_STS_PT	6.43 ha	0.00	18.20	0 ha	1.17026 ha	26
N02_EVO_NSTS_PT	4635.16 m^2	51.45	33.27	2384.79 m^2	1542.118 m^2	61
N02_CHR_STS_GR	10022.94 m^2	17.30	41.10	1733.969 m^2	4119.428 m^2	62
N05_CHR_NSTS_GR	12947.11 m^2	29.70	55.50	3845.292 m^2	7185.646 m^2	62
N02_OUA_STS_MO	16648.78 m^2	0.00	31.77	0 m^2	5289.317 m^2	50
N01_SPI_NSTS_MO	7148.87 m^2	16.50	32.50	1179.564 m^2	2323.383 m^2	50
N02_SID_STS_TU	23.8 ha	0	25.00	0 ha	5.95 ha	49
N06_SID_NSTS_TU	5.91 ha	0	21.10	0 ha	1.24701 ha	49

In this study, the benefits of grass are also taken into account, considering 40% of the environmental benefits; this value is motivated by the results obtained by Nowak et al. (2014) and Gopalakrishnan et al. (2018). However, the economic value of the removal rate of the grass is considered the same as the removal rate of trees.

Ecotope Formation (EF) refers to the landscape's capacity to create ecological structures through the interaction of biotic and abiotic components. Maturity class, naturalness class, structural diversity, species richness, and anthropogenic influence are some of the most important factors used to measure EF (Baitz, 2002; Marks et al., 1989). Olive groves, however, often feature multiple varieties due to varying soil, climatic, and biological conditions, which can affect the entire farm's ecological dynamics. To take into account this complexity, a modification to incorporate the concept of variety into the EF quantification was made. This adjustment considers the distribution of olive varieties within each grove, using a beta distribution model to best represent the diversity across different farms. It was found that the beta distribution is best for measuring EF in agricultural landscapes like olive groves because it can accurately show how the presence and abundance of each variety change over time. The formula for the EF value is (Marks et al., 1989):

$$EF = MC + NC + D_{VAR} + AI$$

Where:

EF [Pt] is the ecotope formation

MC [dimensionless] is the maturity class

NC [dimensionless] is the naturalness class

D_{VAR} is the diversity value calculate as the median between 2 parameters, species richness SR and structural diversity SD, with the first one corrected with the parameter of cultivars ρ:

$$D_{VAR} = \frac{\left[SR + \left(1 - \frac{1}{\rho}\right)\right] + SD}{2}$$

The parameter of cultivars is the standard deviation of the cultivar in the olive farm considering a beta distribution.

AI [dimensionless] is the anthropic influence

The answers to the questionnaires collected in WP5 of the SustainOlive project (De Luca et al., 2023) were assessed to establish the correct value for each factor.

The economic value of EF is calculated considering the entire life cycle costs, retrieved from the results of SustainOlive project, with the removal of production activities like harvesting or phytoiatric treatment.

The value of each parameter is quantified considering the maximum value of the parameter achieved. Table 6.4 describes the inputs for each parameter that are necessary for the calculation of EF.

Table 6.4 Inputs necessary for ecotope formation.

VALUE	MATURITY CLASS	NATURALNESS CLASS	SPECIES RICHNESS (SR)	ANTHROPIC INFLUENCE
5	Closing communities (Climax)	Intact and natural or nearly natural ecosystem	SR >40	Unaffected
4	Permanent communities	N.A.	31<SR<40	Slightly affected
3	Natural and long-lived succession communities	Semi-natural ecosystem	21-30	Impaired
2	Natural pioneer and short-lived succession communities	N.A.	11-20	Damaged
1	Initial stages of pioneer communities	Moderately altered ecosystem, altered ecosystem	1-10	Seriously damaged
0	N.A.	Highly altered or artificial ecosystem	N.A.	Extremely damaged

	DEGREE OF COVERAGE (VERTICAL PROJECTION OF ALL PARTS OF THE PLANT ABOVE GROUND)			
LAYER OF VEGETATION	50-100%	25-50%	5-25%	50-100%
SHRUB LAYER > 2 M	1	0.6	0.3	1
SHRUB LAYER < 2 M	0.5	0.3	0.2	0.5
HERB LAYER > 30 CM	1	0.6	0.3	1
HERB LAYER < 30 CM	0.5	0.3	0.2	0.5

These methods were used on twelve olive farms, two per country partner with similar structural features: one STS and one non-STS. The goal was to compare how sustainable each farm was and see if i-Tree Canopy and EF could be used to include ES assessment in life cycle assessment. The economic value of EF is calculated considering the cost of maintaining the landscape in the constant phase for a single year. This is related to several aspects: the constant phase is the stage in which there is a constant production of olives and the choice of a single year excludes the problem of life span of olive groves that can be different.

6.2 **Results and discussion**

The results of the air purification are shown in Table 6.5, expressed per year and per life cycle:

Table 6.5 Air purification results.

		ANNUAL		PER LIFE CYCLE	
	Years of life cycle from constant phase	**kg y^{-1}**	**€ y^{-1}**	**kg**	**€**
N01_PU_STS_IT	135	457.60	26.26	61,775.55	3,545.12
N02_PU_NSTS_IT	135	71.00	4.07	9,584.53	550.07
N01_DE_STS_SP	135	458.94	26.34	61,957.36	3,555.56
N02_DE_NSTS_SP	65	656.25	37.66	42,656.39	2,447.93
N02_CHR_STS_GR	62	46.46	2.67	2,880.22	165.30
N05_CHR_NSTS_GR	62	84.20	4.83	5,220.51	299.61
N01_POR_STS_PT	26	112.96	6.48	2,937.00	168.55
N02_EVO_NSTS_PT	61	24.09	1.38	1,469.59	84.34
N02_OUA_STS_MO	50	51.05	2.93	2,552.62	146.50
N01_SPI_NSTS_MO	50	22.43	1.29	1,121.28	64.36
N02_SID_STS_TU	49	574.34	32.96	28,142.45	1,615.02
N06_SID_NSTS_TU	49	120.37	6.91	5,898.14	338.48

The column "years of life cycle from constant phase" refers to the life cycle stage of the olive orchard (Table 6.5), excluding the planting phase, the unproductive phase, and the increasing production phase because the trees are still growing, and air purification is not fully operative. The environmental values per year and per life cycle, given respectively in kg y^{-1} and kg, represent the sum of the pollutants mentioned previously. STS systems show better performances in almost all cases. The most important parameter influencing these results is the tree density: more trees imply a larger canopy area, which subsequently enhances air purification. Consequently, adopting an appropriate tree density that ensures a medium – long life cycle of an olive grove, combined with the plantation of cover crops, can result in a significant amount of pollutants being removed. This is evident in the cases of Italy and Spain (N01_PU_STS_IT and N01_DE_STS_SP, respectively). While tree density plays a crucial role in enhancing air purification, the size of the evaluated area can also significantly influence the results, as seen in the Spanish farms, in which the non-STS olive farm has an area more than double that of the STS olive farm.

Another critical factor is the parameter related to the cover crops. Although their contribution to pollutant removal is less pronounced compared to tree density due to lower removal rates, their role remains significant. For example, the N01_CHR_STS_GR system had the same ability to clean the air as other STS systems, even though it had fewer trees per square meters than those other systems. This was because it had vegetation cover. Cover crops not only contribute to pollutant removal but also play a particularly important role during the early stages of the olive grove life cycle, where they enhance soil stability and biodiversity before tree canopies reach maturity.

As mentioned, the life cycle of olive groves is a critical factor. The best benefits for cleaning the air happen during the constant production phase, when the tree canopies are fully grown and can remove the most pollutants. This trend can be seen in the STS systems in Italy and Spain, which have better values than the Greek and Tunisian systems, which have production phases that aren't constant. These findings highlight the importance of adopting agroecological practices and long-term management strategies to maximise the ecological and economic benefits of olive groves across different contexts.

Table 6.6 illustrates the results of the Ecotope Formation:

Table 6.6 Ecotope Formation results.

	ENVIRONMENTAL OUTCOMES	**ECONOMIC OUTCOMES**
	Pt	**€ $ha^{-1}y^{-1}$**
N01_PU_STS_IT	10.08	619.98
N02_PU_NSTS_IT	10.14	861.25
N01_DE_STS_SP	10.00	400.56
N02_DE_NSTS_SP	10.00	411.93
N02_CHR_STS_GR	9.69	368.92
N05_CHR_NSTS_GR	9.69	361.44
N01_POR_STS_PT	10.28	555.69
N02_EVO_NSTS_PT	10.22	239.62
N02_OUA_STS_MO	10.00	203.45
N01_SPI_NSTS_MO	10.00	102.97
N02_SID_STS_TU	10.00	302.13
N06_SID_NSTS_TU	10.20	754.11

The results of the EF show an average value of about 10; the olive farms that have a value slightly higher than 10 tend to have more than one cultivar in the orchard. This characteristic enhances their resilience to adverse conditions such as pathogens or climatic conditions. For example, in Tunisia, the non-STS farm N06_SID_NSTS_TU, with three cultivars, achieved a higher EF value than the STS farm N02_SID_STS_TU, which has only a single cultivar. In Greece, EF values are slightly lower than 10 because of the parameter of "Species Richness," which is 4 compared to 5, as in the other farms evaluated. Other parameters do not show a significant effect, as their values are equal across farms. However, the economic value of EF varies due to differences in management practices. STS farms, for example, often require a more complex approach, including the use of different fertilisers or different pruning schedules. Additionally, managing multiple cultivars within the same orchard may demand tailored inputs and varied techniques. Despite these added complexities, higher EF values correspond to higher economic outcomes. The relationship is evident in Tunisia and Portugal, where farms with elevated EF values also achieved higher economic performance.

The above tables show the results of the Soil Organic Carbon (SOC) test. The numbers are given per hectare and are shown in terms of both carbon stock (tC ha^{-3}) and economic value (€ ha^{-3}).

	ENVIRONMENTAL OUTCOMES		**ECONOMIC OUTCOMES**	
Unit of measure	tC ha^{-1}		€ ha^{-1}	
	min	max	min	max
N01_PU_STS_IT	36.61	37.47	381.67	390.67
N02_PU_nSTS_IT	34.26	34.26	357.22	357.22
N01_DE_STS_SP	30.96	33.19	565.42	606.14
N02_DE_nSTS_SP	30.77	31.33	561.9	572.16
N02_CHR_STS_GR	38.41	38.41	151.5	151.5
N05_CHR_nSTS_GR	25.96	29.56	102.38	116. 59
N01_POR_STS_PT	27.7	30.03	79.12	85.79

N02_EVO_nSTS_PT	35.11	35.14	100.29	100.37
N02_OUA_STS_MO	23.87	23.87	152.28	152.28
N01_SPI_nSTS_MO	44.24	44.24	282.25	282.25
N02_SID_STS_TU	18.17	18.87	85.42	88.71
N06_SID_nSTS_TU	18.87	18.96	88.71	89.12

The column "tC ha⁻¹" represents the total carbon stock per hectare, whereas "€ ha⁻¹" represents its economic valuation. The SOC levels vary significantly depending on farm management strategies. When it comes to accumulating SOC, STS systems usually do better than nSTS systems. This is because they use certain agricultural ecological practices, such as applying manure or reducing the amount of tilling that is done.

In general, the highest SOC values were recorded in Europe with Greek and Italian farms, while the Spanish STS system exhibited the highest economic valuation, reaching up to 606.14 € ha⁻¹.

The lowest SOC values were recorded in Tunisian farms, where N02_SID_STS_TU presented a range of 18.17 - 18.87 tC ha⁻¹, with economic returns significantly lower than other STS farms.

	ENVIRONMENTAL OUTCOMES		**ECONOMIC OUTCOMES**	
Unit of measure	tC ha^{-1}		€ ha^{-1}	
	min	max	min	max
N01_PU_STS_IT	4.18	4.28	43.58	44.61
N02_PU_nSTS_IT	4.56	4.56	47.59	47.59
N01_DE_STS_SP	10.53	11.28	192.23	206.07
N02_DE_nSTS_SP	7.06	7.19	128.99	131.35
N02_CHR_STS_GR	14.1	14.1	55.63	55.63
N05_CHR_nSTS_GR	14.12	16.07	55.67	63.4
N01_POR_STS_PT	14.76	17.49	43.33	46.97
N02_EVO_nSTS_PT	11.37	16.27	41.95	41.99
N02_OUA_STS_MO	10.99	10.99	70.09	70.09
N01_SPI_nSTS_MO	82.96	82.96	529.31	529.31
N02_SID_STS_TU	4.54	4.71	21.33	22.15
N06_SID_nSTS_TU	5.2	5.23	24.22	24.56

This table represents the Soil Organic Carbon per tonne of olive produced: N01_SPI_nSTS_MO (Morocco) exhibits the highest SOC accumulation per tonne of olives (82.96 tC t^{-1}), corresponding to the highest economic return (529.31 € t^{-1}). This suggests that specific management strategies in Moroccan olive groves maximise the carbon sequestration potential relative to production levels.

Conversely, Tunisian farms (N02_SID_STS_TU & N06_SID_nSTS_TU) exhibit the lowest SOC accumulation rates per tonne of olives, with values as low as 4.54 tC t^{-1}, indicating lower sequestration efficiency in relation to yield.

For European farms, the SOC accumulation per tonne of olives produced varies significantly, with Greek (N02_CHR_STS_GR), Portuguese (N01_POR_STS_PT), and Spanish (N01_DE_STS_SP) farms showing higher sequestration rates compared to other European counterparts. The Spanish STS system (N01_DE_STS_SP) exhibits one of the highest carbon sequestration efficiencies per ton of olives, reaching 10.53 - 11.28 tC t^{-1}, which also translates into a higher economic valuation (192.23 - 206.07 € t^{-1}). This means that European farms that use sustainable tree systems (STS) and different ways of managing their trees can greatly increase their ability to store carbon and their economic benefits.

6.3 **Conclusion**

European agricultural policies are mostly focused on the regulation of primary production. The Common Agricultural Policy's environmental rules encourage low-impact farming through a mix of required and voluntary mechanisms. These include the "greening" payment for farmers who adopt practices that are good for the climate and the environment, as well as agri-environmental measures and cross-compliance within the direct payment framework. The goals of the planned revision of the Common Agricultural Policy after 2020 call for farmers to play a bigger role in tackling climate change and providing healthy, long-lasting food, while also trying to make production more efficient (Gava et al., 2020). There exist various methodologies for effect mitigation, each possessing multifaceted objectives that seem to lack complete endorsement by the current post-2020 concept (Pe’er et al., 2019). To mitigate the effects of the food system and advance the attainment of the Sustainable Development Goals (SDGs), it is essential to adopt a novel and comprehensive approach to policy formulation through an updated policy mix (Flanagan et al., 2011) or mission-orientated (Mazzucato, 2018) policies that address both supply and demand aspects (Kanter et al., 2020). For evidence-based policy to work, it needs up-to-date data from thorough impact assessment studies and scenario assessments. This is to reduce uncertainty in policymaking and keep goals from being sacrificed. Information failure frequently influences the formulation of agricultural policy, making it particularly crucial. Using systems thinking to plan interventions could help deal with more than one aspect of sustainability at the same time by encouraging supply chains and stakeholders to work together. This would enable greater focus on intermediate stages of the supply chain, supply chain management alternatives, and the social acceptability of initiatives. The presence of numerous interacting components renders the assessment of intervention consequences on the environment, economy, and society a complicated endeavour (Gava et al., 2020).

Given this premise, the use of geo-spatial methods can allow the incorporation of ecosystem services into life cycle methodologies, creating the conditions for decision-makers to establish correct policies (Burkhard et al., 2012).

The i-Tree Canopy tool has proven to be an effective online resource for understanding the environmental and economic value of olive trees, as well as evaluating different management scenarios to identify the most sustainable ones in terms of air purification. The joint application of this method with the results of an LCA study provides a comprehensive understanding of the areas of interest influenced by the impacts and benefits of using STS practices at the expense of business-as-usual practices. Air pollution adversely affects crop yield, human health, and, by extension, the economy (Guerreiro et al., 2014). Air pollution contributes to pulmonary illnesses, cancer, and respiratory infections, elevating mortality rates and public health expenditures (Cohen et al., 2005). Furthermore, O_3 exposure diminishes plant transpiration and heat fluxes, resulting in reduced rainfall and elevated air temperature. These things have a negative impact on agricultural output by 18% to 45%, which is about 35% of the loss in gross primary productivity (GPP) (Li et al., 2018). Various studies demonstrate the health advantages of residing near green spaces, enhancing human well-being by lowering blood pressure, heart rate, and muscular tension (Hamer et al., 2009, p. 2011; Mitchell et al., 2011). Unfortunately, many people live far from natural places in their daily lives, and this distance may shape their perception of natural environments and local biodiversity. Nonetheless, local communities need to examine how social, cultural, and psychological contexts influence individual perceptions.

The use of methods highly specific for the territory under examination can help the decision -makers to establish the correct remuneration for the adoption of certain agricultural practices. In Europe, for example, the Ecotope Formation (EF) can be related to the eco-scheme 3, i.e. one of the payment schemes in agriculture aiming at the protection of the environment and climate. Eco-scheme 3 emphasises the importance of maintaining agricultural systems that contribute to biodiversity, soil health, and cultural heritage. Olive groves with high ecological and landscape values, such as those demonstrating elevated EF, align closely with the objectives of this scheme.

By integrating EF metrics into CAP assessments, decision-makers can better target funding to farms that enhance ecosystem services, ensuring both environmental and economic sustainability. This method is especially useful because eco-schemes are one of the most important new ideas in the CAP reform. They are meant to reward actions that go above and beyond basic environmental compliance (European Commission, 2021a, 2021b, 2021c; Pe'er et al., 2019). This kind of alignment between EF results and eco-scheme goals not only helps protect traditional landscapes, but it also gets more farmers to use sustainable farming methods. In addition to the EF linked to eco-scheme 3, air purification can also be incentivised through eco-scheme 2, which focuses on the adoption of cover crops as a means to enhance ecosystem services and ecological stability. While cover crops are not directly aimed at pollutant removal, their role in maintaining vegetation cover and promoting ecological stability contributes to the overall functionality of agroecosystems by indirectly supporting air purification. Eco-scheme 2 provides financial incentives to farmers who adopt cover crops as part of their agricultural management. Cover crops make the vegetation layer in STS systems stronger, which helps with things like carbon sequestration, habitat provision, and the overall functioning of the ecosystem (Blanco-Canqui et al., 2015). For instance, farms such as N01_PU_STS_IT demonstrate enhanced air purification capacity, partly due to the integration of cover crops that strengthen the ecosystem's ability to deliver multiple services.

By linking the adoption of cover crops to air purification and other ecosystem services, eco-scheme 2 provides a mechanism to reward farmers for their contributions to ecological stability and resilience. This makes sure that public money goes to activities that not only improve the way ecosystems work but also help reach bigger CAP goals for sustainability (European Commission, 2021a, 2021b, 2021c).

Finally, the results of this study highlight the critical role of sustainable management practices in enhancing soil organic carbon sequestration and maximising economic benefits. Farms using STS systems always have more SOC storage per hectare, showing that conservation-based farming methods have a positive effect.

However, the analysis of SOC per tonne of olives produced reveals an important trade-off: farms with higher yields tend to have lower SOC accumulation per tonne of product, whereas farms with lower yields but enhanced soil management demonstrate superior sequestration per unit of production. This pattern is evident in the Moroccan system (N01_SPI_nSTS_MO), which shows the highest SOC values per tonne of olives.

The results make it clear that a balanced approach to agricultural sustainability is needed, taking into account both carbon sequestration per hectare and carbon efficiency relative to yield. Future research should focus on refining agricultural techniques that optimise SOC accumulation while maintaining high productivity, ensuring a resilient and climate-friendly olive farming system across diverse geographical contexts.

Further results concerning the other olive farms of SustainOlive can be found in the appendix at the end of this document.

Abbreviations

AI	Anthropic Influence
D	Diversity
EF	Ecotope Formation
ESs	Ecosystem Services
EU	Europe
GPP	Gross Primary Productivity
LANCA	Land Use Indicator Value Calculation for Life Cycle Assessment
LC	Life Cycle
LCA	Life Cycle Assessment
MA	Millennium Ecosystem Assessment
MC	Maturity Class
NC	Naturalness Class
SD	Structural Diversity
SOC	Soil Organic Carbon
SR	Species Richness
STSs	Sustainable Technological Solutions
USDA	United States Department for Agriculture
WTP	Willingness to Pay

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Chapter 7.
CONCLUSIONS AND FUTURE PERSPECTIVE

Ecosystem services are not only a theoretical concept, but they also play an important role in several practices and decisions as well as events. An example is the two floods in the Emilia – Romagna region, where the particular climatic conditions, jointly with soil characteristics, created the perfect conditions for these events (Cremonini et al. 2024). Sealing the soil seems to be an important driver because water cannot increase the ESs linked to groundwater regeneration in the soil or filtration (Pistocchi et al. 2015). Other events where ESs are important include the recent pandemic, where the spillover happened due to a biodiversity threat (Lawler et al. 2021; Brema et al. 2022), and the global conflicts, such as the Ukrainian war, which reduced the strength to curb CO_2 emissions and nullify soil carbon storage (Pereira et al. 2022; Hapich et al. 2024). If on one side ESs are determinant in real contexts, on the other side there is a need to produce tools and methodologies to measure and quantify ESs with the final goal of producing effective policy decisions to protect ecosystem quality and biodiversity. In this context, tools like the Global Guidance for Life-Cycle Impact Assessment Method (GLAM), the Common Agricultural Policy (CAP), or the Gross Ecosystem Product (GEP) can set the right conditions to include these important elements in the decision – making process.

The Global Guidance for Life-Cycle Impact Assessment Method (GLAM) is a project divided into three phases. The third phase concerns the implementation of methods to measure impacts in several environmental impact categories of LCA. This phase involves the implementation of methods related to human health, ecosystem quality, natural resources, and ecosystem services. The methods used for the computation of ES impacts are based on the LANCA model, the same methods used in this thesis to measure most of the ESs considered.

For the economic evaluations, the use of Gross Ecosystem Product (GEP), which is founded on previous efforts that established integrated environmental-economic accounting, such as the UN SEEA (United Nations. Bureau of the Committee of Experts on Environmental-Economic Accounting and United Nations. Committee of Experts on Environmental-Economic Accounting 2014) and the SEEA Experimental Ecosystem Accounting framework (EEA) (European Union 2014). The most recent iteration of the SEEA-EA incorporates the idea of GEP as a prospective metric for quantifying ecosystem service flows in monetary terms. GEP is currently being applied in many decision-making situations throughout China and several other nations, including Colombia, Sri Lanka, and Sweden, who want to adopt GEP accounting.

In essence, GEP represents a novel advancement in the understanding and application of ecosystem services and natural capital in two fundamental aspects. Initially, GEP functions as an innovative composite metric of the value derived from utilised ecosystem services, encapsulating the essential contributions that nature provides to the economy (Ouyang et al. 2013; Ma et al. 2015; Ouyang and Jin 2017). The ideas and stringent definitions associated with GEP establish clear connections to decision-making, showcasing extensive policy relevance in China and application to other nations as well. It is crucial to recognise that GEP serves as a lower-bound estimate of the benefits of ecosystems for society, given that humans are entirely intertwined with nature in ways that cannot be comprehensively quantified in economic terms (e.g., Goulder and Kennedy (1997); Kimmerer (2013)). Nevertheless, GEP offers a robust methodology for reforming the economic system to recognise essential values that decision-makers currently overlook.

The GEP quantifies the contribution of ecosystems to human welfare in economic and monetary terms, which is crucial for evaluating the utilisation and valuation of ecosystem services. Analogous to GDP, the GEP uses market pricing and market price proxies to determine the accounting value of ecosystem services. These are subsequently consolidated into a metric reflecting the contribution of ecosystems to the economy. Using methodologies similar to those of GDP enhances the effectiveness of the GEP. GEP serves as a valuable adjunct to GDP, emphasising the ecological contributions overlooked in GDP assessments. It is essential to recognise the intersection between GEP and GDP, as many ecosystem service outputs encompassed in GEP also serve as inputs for the production of commodities and services quantified in GDP (e.g., agricultural products, lumber, and ecotourism). Consequently, one cannot only aggregate GEP and GDP to produce a significant value (Zheng et al. 2023). Despite several efforts to quantify this outcome, there is currently no common framework to calculate GEP.

In the context of policy implementation, especially within the European Union, ecosystem services are closely associated with specific agronomic practices employed in olive cultivation, and their environmental and economic quantification is crucial for assessing the indirect multifunctionality of agriculture. The STS was explicitly formulated to comply with designated agro-ecological strategies, such as those specified in the Common Agricultural Policy (CAP) at the European level for the 2023-2027 timeframe. Subsequently, each nation translated the overarching goals of the CAP into particular objectives at the national level. A succession of executed operations in Italy has transformed agroecological payments into ecosystems. This study examines agroecological practices categorised under Eco-scheme 2, governed by the Ministry of Agriculture, Food Sovereignty and Forestry (MASAF) for the year 2023. The current investment for the maintenance of heritage under this ecological initiative is €120 per hectare (MASAF-Ministry of Agriculture, Food Sovereignty and Forests 2023).

This study looked at how to include ecosystem services (ES) in Life Cycle Assessment (LCA) methods used for growing olives in the Mediterranean. While looking at different approaches and using certain tools, this thesis helped figure out the main problems with combining ES and LCA methods. It suggests using more than one method, including EPS 2015, LANCA, and i-Tree Canopy. Using this method on olive farming, the study gave a numerical estimate of ecosystem services and looked at how human activities affect the provision and evaluation of ES in LCA. The study also shows that current LCA frameworks need to be improved so that they better take into account ES functions other than providing and regulating them, such as cultural and supporting services.

One key aspect emerging from this research is the need to assess the scalability of the proposed methods to other agricultural systems. The methodologies used can also be adapted to other crops, particularly perennial tree systems (e.g., vineyards and orchards), which share similar characteristics with olive growing, as well as annual crops (e.g., cereals and vegetables), although adaptability requires specific modifications in ES evaluation models. The main limitations lie in the need for detailed data in each context and the availability of modelling tools that cover a broader range of agricultural systems. Furthermore, the effectiveness of the proposed methodologies in diverse climatic and soil conditions must be further investigated.

The results obtained can provide significant support for agroecological policies and payments based on ecosystem services. In particular, the proposed methodology could be used to develop schemes of payment for ecosystem services (PES), incentivising sustainable practices such as cover cropping to reduce soil erosion. Finding out how much olive trees help the environment could help Common Agricultural Policy (CAP) programs that try to be more sustainable. The findings could also be used to come up with criteria for environmental certifications (like organic farming and regenerative practices) based on ecosystem service metrics. Including ES valuation in policy frameworks could also help make subsidies and incentives more targeted, which would encourage best practices in agriculture that are in line with sustainability goals.

A fundamental aspect of the large-scale application of the proposed methodology is the balance between the complexity of the required data and the added value in sustainability assessment. In this context, a preliminary estimate of time and resources required is provided (the results were calculated considering the times of this study):

- Collection of primary data (olive groves) requires 2-3 months with measurements on farm boundaries, general characteristics of the farms like the varieties and the size plot, and the specific data for the LCA software.
- Application of the LANCA model requires 1 month of applying geological and climate data and researching specific databases.
- Analysis with EPS 2015 requires 2 weeks with LCA software (e.g., Simapro).
- Processing with i-Tree Canopy requires 1 month for the survey point set to 1000 points.
- Elaborations in GIS software require 2-3 months to set the specific characteristics of the software and set the methods.

Data collection remains the most burdensome element, suggesting the need to automate some phases through satellite data and predictive models to reduce workload. Moreover, the potential of remote sensing and machine learning should be further explored to enhance the accuracy and efficiency of ES assessments.

Another relevant aspect that emerged is the role of anthropogenic activities in influencing the provision and evaluation of ecosystem services. Human influence was analysed in relation to:

- Soil management, where the use of cover crops positively affects water regulation and erosion reduction.
- Irrigation practices, which can negatively impact local water availability and nutrient balance.
- Biodiversity, where habitat management influences the quality of ES provided by olive groves.

To make the integration of ES into LCA work better, it is necessary a dynamic, site-specific approach that takes into account how farming practices change over time. Further research is necessary to quantify the trade-offs between various agricultural practices and their influence on ecosystem services.

Based on the results obtained, several future perspectives can be outlined, such as:

- Expansion to other agricultural systems involves testing the proposed approach on different crops such as vineyards and cereals to verify its adaptability.
- Development of digital tools by implementing predictive models and GIS to automate data collection on ES.
- Integration into European policies by collaborating with research institutions and policymakers to operationalise the developed indicators within the CAP and the EU Green Taxonomy.

- Improvement of result communication by making ES data accessible to stakeholders, farmers, and policymakers through interactive dashboards and summary reports.

Abbreviations

CAP	Common Agricultural Policy
EEA	Experimental Ecosystem Accounting
ESs	Ecosystem Services
GDP	Gross Domestic Product
GEP	Gross Ecosystem Product
GLAM	Global Guidance for Life-Cycle Impact Assessment Method
LANCA	Land Use Indicator Value Calculation for Life Cycle Assessment
LCA	Life Cycle Assessment
MA	Millennium Ecosystem Assessment
MC	Maturity Class
NC	Naturalness Class
PES	Payment for Ecosystem Services
SD	Structural Diversity
SOC	Soil Organic Carbon
SR	Species Richness
STSs	Sustainable Technological Solutions
USDA	United States Department for Agriculture
WTP	Willingness to Pay

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APPENDIX

Provisioning services

Crop growth capacity, production capacity for fruit & vegetable, wood growth capacity and fish & meat production capacity show different trends depending on the farming practice adopted and the geographical context.

In Italy, the differences between STS and non-STS vary depending on the indicator considered. In crop growth capacity, STS farms generally record higher values than their non-STS counterparts, indicating that sustainable practices can lead to improved performance in certain contexts. For wood growth capacity, STS farms show a higher avoided impact than non-STS farms, highlighting the environmental benefits of sustainable practices in this context.

In Greece, STS farms are superior for most indicators. STS farms achieve significantly higher environmental values than their non-STS counterparts. However, in some specific cases, the differences are small, suggesting that local factors, such as soil quality and management techniques, can reduce the gap between the two practices.

In Spain, STS farms are generally more advantageous. The STS farms record a significantly higher avoided impact than their non-STS counterparts, especially in wood growth capacity, where the difference is particularly marked. However, there are situations where non-STS achieve comparable results, suggesting that the effectiveness of STS may depend on the specific context.

In Portugal, the difference between STS and non-STS is significant, with STS farms showing a higher avoided impact in crop growth capacity compared to non-STS farms. STS olive farms show superior environmental performance compared to their non-STS counterparts, but the gap is not as clear as in Spain or Greece. However, the economic benefit of STS is clearer in cases where the reduction of environmental impact is greater, leading to a reduction in operating costs.

In Morocco, STS farms demonstrate a superior ability to improve crop growth capacity and wood growth capacity compared to non-STS. The STS farms perform comparably to non-STS farms in crop growth capacity, with only minor differences observed. However, in some cases the difference between the two practices is small, suggesting that the local environmental context plays a role.

In Tunisia, the distinction between STS and non-STS is less pronounced in crop growth capacity, with both systems showing similar performance levels. The economic benefit of STS is evident in sectors with a greater reduction in environmental impact, resulting in lower operating costs.

At the inter-country level, the results show that Greece and Spain benefit more from STS farms, with significantly higher avoided impacts than non-STS. In Italy and Morocco, the situation is more balanced, with some indicators in favor of non-STS. In Portugal and Tunisia, although STS are generally more advantageous, the gap with non-STS is not so marked.

These results suggest that soil management and local environmental conditions significantly influence the effectiveness of the practices adopted. While in some countries STS practices demonstrate a clear superiority, in others the differences are less evident or even reversed depending on the context and the indicator considered.

These results should also be related to the EPS2015 modelling, as the impacts of nitrogen, for example, are attributed to fish & meat production capacity, which are not the subject of this study. This may constitute a limitation in the interpretation of the results if one does not have a deep knowledge of the EPS2015 modelling.

Table A. 1, Table A. 2, Table A. 3 and Table A. 4 show the environmental and economic results of EPS2015 model for both FUs (1 ha and 1 ton)

Table A. 1 Environmental results of EPS2015 model (FU=1 ha).

	CROP GROWTH CAPACITY	PRODUCTION CAPACITY FOR FRUIT&VEGETABLES	WOOD GROWTH CAPACITY	FISH&MEAT PRODUCTION CAPACITY
OLIVE FARMS	kg	kg	kg	kg
N01_CHR_STS_GR	-62.26	-11.34	-549.08	-4.12
N02_CHR_STS_GR	-75.31	-12.42	-564.63	-4.55
N03_CHR_STS_GR	-75.36	-12.43	-564.57	-4.55
N04_CHR_NSTS_GR	2.63	-2.45	-195.75	-0.79
N05_CHR_NSTS_GR	2.63	-2.45	-195.75	-0.79
N06_CHR_NSTS_GR	4.81	-2.32	-201.05	-0.72
N07_KAL_STS_GR	-14.35	-3.05	-134.26	-1.12
N08_KAL_STS_GR	-2.99	-1.66	-88.80	-0.59
N09_KAL_NSTS_GR	-18.95	-7.08	-160.30	-2.60
N10_KAL_NSTS_GR	-55.54	-16.06	-289.32	-6.00
NO1_PU_STS_IT	31.48	1.68	-55.90	0.79
NO2_PU_NSTS_IT	29.94	1.94	-37.69	0.87
NO3_LA_STS_IT	42.66	2.04	-92.76	0.94
NO4_LA_NSTS_IT	11.04	-0.90	-167.39	-0.12
NO5_TO_STS_IT	0.60	-1.41	-65.60	-0.50
NO6_TO_NSTS_IT	9.62	-0.46	-68.78	-0.11
NO7_CA_STS_IT	6.08	-0.90	-99.05	-0.27
NO8_CA_NSTS_IT	27.54	1.83	-30.81	0.78
NO9_LI_STS_IT	12.97	0.37	-26.03	0.18
N01_SPI_NSTS_MO	8.28	0.41	-12.62	0.18
N02_OUA_STS_MO	8.21	0.40	-12.93	0.18
N03_AME_STS_MO	15.24	1.13	-32.40	0.49
N04_AMG_NSTS_MO	15.02	1.20	-13.29	0.50
N05_TAN_STS_MO	14.93	1.19	-13.23	0.50
N06_OUA_STS_MO	14.93	1.19	-13.24	0.50
N07_DAR_NSTS_MO	14.93	1.19	-13.24	0.50
N01_POR_STS_PT	37.01	2.44	-42.93	1.04
N02_EVO_NSTS_PT	8.84	0.21	-25.55	0.12
N03_ALC_STS_PT	-45.84	-12.16	-98.61	-4.62
N04_SER_NSTS_PT	40.51	1.32	-131.85	1.22
N05_SER_NSTS_PT	-10.44	-4.53	-271.91	-1.48

N06_SER_STS_PT	31.06	1.30	-62.96	0.64
N01_DE_STS_SP	-6.18	-2.36	-122.44	-0.84
N02_DE_NSTS_SP	12.25	0.27	-41.21	0.18
N03_ES_STS_SP	9.72	0.67	-21.87	0.29
N04_ES_NSTS_SP	3.29	-1.03	-114.31	-0.26
N05_PV1_STS_SP	14.35	1.00	-33.35	0.46
N07_PV2_STS_SP	-2.04	-1.27	-135.33	-0.38
N08_JT_STS_SP	11.00	0.33	-29.19	0.17
N09_IS_STS_SP	4.97	-0.41	-87.15	-0.06
N10_CH_STS_SP	3.64	-0.57	-86.94	-0.12
N11_CH_NSTS_SP	0.62	-0.72	-93.03	-0.18
N12_MR_STS_SP	0.58	-0.95	-100.23	-0.26
N13_PA_STS_SP	14.56	-0.63	-156.81	-0.01
N14_PA_NSTS_SP	10.29	-0.04	-62.85	0.07
N15_GA_STS_SP	-11.52	-2.53	-144.89	-0.83
N16_GA_NSTS_SP	15.13	0.54	-59.16	0.35
N18_CC_STS_SP	44.32	3.55	-39.89	1.51
N01_TOU_STS_TU	13.35	0.98	-21.84	0.42
N02_SID_STS_TU	19.06	1.61	-14.02	0.67
N03_AMA_NSTS_TU	15.36	1.23	-12.63	0.52
N04_MEN_STS_TU	17.43	1.44	-13.40	0.60
N05_ELF_STS_TU	17.48	1.45	-13.48	0.60
N06_SID_NSTS_TU	17.43	1.44	-13.40	0.60
N07_MEN_NSTS_TU	15.35	1.23	-12.60	0.51
N08_CHI_NSTS_TU	39.67	1.47	-72.68	0.69
N09_CHI_STS_TU	29.20	2.15	-31.94	0.91
N10_SID_NSTS_TU	12.41	0.73	-16.21	0.32
N11_SID_STS_TU	10.67	0.65	-13.07	0.28

Table A. 2 Economic results of EPS2015 model (FU=1 ha).

	CROP GROWTH CAPACITY	**PRODUCTION CAPACITY FOR FRUIT&VEGETABLES**	**WOOD GROWTH CAPACITY**	**FISH&MEAT PRODUCTION CAPACITY**
OLIVE FARMS	ELU	ELU	ELU	ELU
N01_CHR_STS_GR	-13.70	-4.42	-21.96	-8.66
N02_CHR_STS_GR	-16.57	-4.84	-22.59	-9.56
N03_CHR_STS_GR	-16.58	-4.85	-22.58	-9.57
N04_CHR_NSTS_GR	0.58	-0.96	-7.83	-1.66
N05_CHR_NSTS_GR	0.58	-0.96	-7.83	-1.66
N06_CHR_NSTS_GR	1.06	-0.90	-8.04	-1.52
N07_KAL_STS_GR	-3.16	-1.19	-5.37	-2.34
N08_KAL_STS_GR	-0.66	-0.65	-3.55	-1.23
N09_KAL_NSTS_GR	-4.17	-2.76	-6.41	-5.45
N10_KAL_NSTS_GR	-12.22	-6.26	-11.57	-12.60
NO1_PU_STS_IT	6.92	0.66	-2.24	1.66
NO2_PU_NSTS_IT	6.59	0.76	-1.51	1.82
NO3_LA_STS_IT	9.38	0.79	-3.71	1.98
NO4_LA_NSTS_IT	2.43	-0.35	-6.70	-0.25

NO5_TO_STS_IT	0.13	-0.55	-2.62	-1.05
NO6_TO_NSTS_IT	2.12	-0.18	-2.75	-0.23
NO7_CA_STS_IT	1.34	-0.35	-3.96	-0.57
NO8_CA_NSTS_IT	6.06	0.71	-1.23	1.64
NO9_LI_STS_IT	2.85	0.14	-1.04	0.39
N01_SPI_NSTS_MO	1.82	0.16	-0.50	0.38
N02_OUA_STS_MO	1.81	0.16	-0.52	0.38
N03_AME_STS_MO	3.35	0.44	-1.30	1.03
N04_AMG_NSTS_MO	3.31	0.47	-0.53	1.06
N05_TAN_STS_MO	3.29	0.47	-0.53	1.05
N06_OUA_STS_MO	3.28	0.47	-0.53	1.05
N07_DAR_NSTS_MO	3.28	0.47	-0.53	1.05
N01_POR_STS_PT	8.14	0.95	-1.72	2.19
N02_EVO_NSTS_PT	1.95	0.08	-1.02	0.25
N03_ALC_STS_PT	-10.09	-4.74	-3.94	-9.70
N04_SER_NSTS_PT	8.91	0.52	-5.27	2.57
N05_SER_NSTS_PT	-2.30	-1.77	-10.88	-3.12
N06_SER_STS_PT	6.83	0.51	-2.52	1.34
N01_DE_STS_SP	-1.36	-0.92	-4.90	-1.76
N02_DE_NSTS_SP	2.69	0.10	-1.65	0.37
N03_ES_STS_SP	2.14	0.26	-0.87	0.62
N04_ES_NSTS_SP	0.72	-0.40	-4.57	-0.54
N05_PV1_STS_SP	3.16	0.39	-1.33	0.96
N07_PV2_STS_SP	-0.45	-0.50	-5.41	-0.81
N08_JT_STS_SP	2.42	0.13	-1.17	0.36
N09_IS_STS_SP	1.09	-0.16	-3.49	-0.12
N10_CH_STS_SP	0.80	-0.22	-3.48	-0.25
N11_CH_NSTS_SP	0.14	-0.28	-3.72	-0.38
N12_MR_STS_SP	0.13	-0.37	-4.01	-0.56
N13_PA_STS_SP	3.20	-0.25	-6.27	-0.01
N14_PA_NSTS_SP	2.26	-0.02	-2.51	0.14
N15_GA_STS_SP	-2.53	-0.99	-5.80	-1.75
N16_GA_NSTS_SP	3.33	0.21	-2.37	0.73
N18_CC_STS_SP	9.75	1.38	-1.60	3.17
N01_TOU_STS_TU	2.94	0.38	-0.87	0.88
N02_SID_STS_TU	4.19	0.63	-0.56	1.40
N03_AMA_NSTS_TU	3.38	0.48	-0.51	1.08
N04_MEN_STS_TU	3.83	0.56	-0.54	1.26
N05_ELF_STS_TU	3.85	0.56	-0.54	1.27
N06_SID_NSTS_TU	3.83	0.56	-0.54	1.26
N07_MEN_NSTS_TU	3.38	0.48	-0.50	1.08
N08_CHI_NSTS_TU	8.73	0.57	-2.91	1.45
N09_CHI_STS_TU	6.42	0.84	-1.28	1.91
N10_SID_NSTS_TU	2.73	0.29	-0.65	0.67
N11_SID_STS_TU	2.35	0.25	-0.52	0.59

Table A. 3 Environmental results of EPS2015 model (FU=1 ton).

	CROP GROWTH CAPACITY	PRODUCTION CAPACITY FOR FRUIT&VEGETABLES	WOOD GROWTH CAPACITY	FISH&MEAT PRODUCTION CAPACITY
OLIVE FARMS	kg	kg	kg	kg
N01_CHR_STS_GR	-23.51	-4.28	-207.37	-1.56
N02_CHR_STS_GR	-27.46	-4.53	-205.90	-1.66
N03_CHR_STS_GR	-41.22	-6.80	-308.79	-2.49
N04_CHR_NSTS_GR	1.48	-1.38	-110.58	-0.45
N05_CHR_NSTS_GR	1.43	-1.33	-106.45	-0.43
N06_CHR_NSTS_GR	1.82	-0.88	-75.93	-0.27
N07_KAL_STS_GR	-4.37	-0.93	-40.86	-0.34
N08_KAL_STS_GR	-0.96	-0.53	-28.45	-0.19
N09_KAL_NSTS_GR	-5.99	-2.24	-50.69	-0.82
N10_KAL_NSTS_GR	-13.25	-3.83	-69.00	-1.43
NO1_PU_STS_IT	3.59	0.19	-6.38	0.09
NO2_PU_NSTS_IT	3.99	0.26	-5.02	0.12
NO3_LA_STS_IT	8.49	0.41	-18.45	0.19
NO4_LA_NSTS_IT	2.06	-0.17	-31.30	-0.02
NO5_TO_STS_IT	0.15	-0.36	-16.71	-0.13
NO6_TO_NSTS_IT	2.15	-0.10	-15.34	-0.02
NO7_CA_STS_IT	1.40	-0.21	-22.77	-0.06
NO8_CA_NSTS_IT	5.62	0.37	-6.29	0.16
NO9_LI_STS_IT	6.41	0.18	-12.86	0.09
N01_SPI_NSTS_MO	15.52	0.77	-23.66	0.34
N02_OUA_STS_MO	3.78	0.18	-5.95	0.08
N03_AME_STS_MO	3.59	0.27	-7.63	0.12
N04_AMG_NSTS_MO	20.30	1.62	-17.96	0.68
N05_TAN_STS_MO	5.97	0.48	-5.29	0.20
N06_OUA_STS_MO	31.04	2.48	-27.52	1.04
N07_DAR_NSTS_MO	4.12	0.33	-3.65	0.14
N01_POR_STS_PT	20.04	1.32	-23.24	0.56
N02_EVO_NSTS_PT	3.68	0.09	-10.63	0.05
N03_ALC_STS_PT	-28.55	-7.57	-61.40	-2.88
N04_SER_NSTS_PT	4.87	0.16	-15.84	0.15
N05_SER_NSTS_PT	-1.37	-0.59	-35.61	-0.19
N06_SER_STS_PT	3.71	0.15	-7.52	0.08
N01_DE_STS_SP	-2.32	-0.89	-45.96	-0.31
N02_DE_NSTS_SP	4.40	0.10	-14.80	0.06
N03_ES_STS_SP	2.08	0.14	-4.67	0.06
N04_ES_NSTS_SP	0.56	-0.18	-19.53	-0.04
N05_PV1_STS_SP	6.01	0.42	-13.96	0.19
N07_PV2_STS_SP	-0.52	-0.32	-34.55	-0.10
N08_JT_STS_SP	5.22	0.16	-13.85	0.08
N09_IS_STS_SP	1.31	-0.11	-22.93	-0.02
N10_CH_STS_SP	0.70	-0.11	-16.79	-0.02
N11_CH_NSTS_SP	0.16	-0.19	-24.59	-0.05
N12_MR_STS_SP	0.10	-0.16	-16.38	-0.04
N13_PA_STS_SP	2.03	-0.09	-21.90	0.00

N14_PA_NSTS_SP	2.95	-0.01	-18.00	0.02
N15_GA_STS_SP	-3.41	-0.75	-42.93	-0.25
N16_GA_NSTS_SP	2.88	0.10	-11.25	0.07
N18_CC_STS_SP	8.04	0.64	-7.23	0.27
N01_TOU_STS_TU	3.38	0.25	-5.54	0.11
N02_SID_STS_TU	4.70	0.40	-3.46	0.17
N03_AMA_NSTS_TU	2.95	0.24	-2.43	0.10
N04_MEN_STS_TU	2.07	0.17	-1.60	0.07
N05_ELF_STS_TU	2.24	0.19	-1.73	0.08
N06_SID_NSTS_TU	4.77	0.39	-3.67	0.16
N07_MEN_NSTS_TU	5.84	0.47	-4.79	0.20
N08_CHI_NSTS_TU	15.26	0.56	-27.95	0.27
N09_CHI_STS_TU	7.33	0.54	-8.01	0.23
N10_SID_NSTS_TU	3.18	0.19	-4.16	0.08
N11_SID_STS_TU	2.24	0.14	-2.74	0.06

Table A. 4 Economic results of EPS2015 model (FU=1 ton).

	CROP GROWTH CAPACITY	**PRODUCTION CAPACITY FOR FRUIT&VEGETABLES**	**WOOD GROWTH CAPACITY**	**FISH&MEAT PRODUCTION CAPACITY**
OLIVE FARMS	ELU	ELU	ELU	ELU
N01_CHR_STS_GR	-5.17	-1.67	-8.29	-3.27
N02_CHR_STS_GR	-6.04	-1.77	-8.24	-3.49
N03_CHR_STS_GR	-9.07	-2.65	-12.35	-5.23
N04_CHR_NSTS_GR	0.33	-0.54	-4.42	-0.94
N05_CHR_NSTS_GR	0.31	-0.52	-4.26	-0.90
N06_CHR_NSTS_GR	0.40	-0.34	-3.04	-0.57
N07_KAL_STS_GR	-0.96	-0.36	-1.63	-0.71
N08_KAL_STS_GR	-0.21	-0.21	-1.14	-0.39
N09_KAL_NSTS_GR	-1.32	-0.87	-2.03	-1.72
N10_KAL_NSTS_GR	-2.91	-1.49	-2.76	-3.00
NO1_PU_STS_IT	0.79	0.07	-0.26	0.19
NO2_PU_NSTS_IT	0.88	0.10	-0.20	0.24
NO3_LA_STS_IT	1.87	0.16	-0.74	0.39
NO4_LA_NSTS_IT	0.45	-0.07	-1.25	-0.05
NO5_TO_STS_IT	0.03	-0.14	-0.67	-0.27
NO6_TO_NSTS_IT	0.47	-0.04	-0.61	-0.05
NO7_CA_STS_IT	0.31	-0.08	-0.91	-0.13
NO8_CA_NSTS_IT	1.24	0.15	-0.25	0.33
NO9_LI_STS_IT	1.41	0.07	-0.51	0.19
N01_SPI_NSTS_MO	3.41	0.30	-0.95	0.72
N02_OUA_STS_MO	0.83	0.07	-0.24	0.17
N03_AME_STS_MO	0.79	0.10	-0.31	0.24
N04_AMG_NSTS_MO	4.47	0.63	-0.72	1.43
N05_TAN_STS_MO	1.31	0.19	-0.21	0.42
N06_OUA_STS_MO	6.83	0.97	-1.10	2.18
N07_DAR_NSTS_MO	0.91	0.13	-0.15	0.29
N01_POR_STS_PT	4.41	0.51	-0.93	1.19

N02_EVO_NSTS_PT	0.81	0.03	-0.43	0.10
N03_ALC_STS_PT	-6.28	-2.95	-2.46	-6.04
N04_SER_NSTS_PT	1.07	0.06	-0.63	0.31
N05_SER_NSTS_PT	-0.30	-0.23	-1.42	-0.41
N06_SER_STS_PT	0.82	0.06	-0.30	0.16
N01_DE_STS_SP	-0.51	-0.35	-1.84	-0.66
N02_DE_NSTS_SP	0.97	0.04	-0.59	0.13
N03_ES_STS_SP	0.46	0.06	-0.19	0.13
N04_ES_NSTS_SP	0.12	-0.07	-0.78	-0.09
N05_PV1_STS_SP	1.32	0.16	-0.56	0.40
N07_PV2_STS_SP	-0.11	-0.13	-1.38	-0.21
N08_JT_STS_SP	1.15	0.06	-0.55	0.17
N09_IS_STS_SP	0.29	-0.04	-0.92	-0.03
N10_CH_STS_SP	0.15	-0.04	-0.67	-0.05
N11_CH_NSTS_SP	0.04	-0.07	-0.98	-0.10
N12_MR_STS_SP	0.02	-0.06	-0.66	-0.09
N13_PA_STS_SP	0.45	-0.03	-0.88	0.00
N14_PA_NSTS_SP	0.65	0.00	-0.72	0.04
N15_GA_STS_SP	-0.75	-0.29	-1.72	-0.52
N16_GA_NSTS_SP	0.63	0.04	-0.45	0.14
N18_CC_STS_SP	1.77	0.25	-0.29	0.57
N01_TOU_STS_TU	0.74	0.10	-0.22	0.22
N02_SID_STS_TU	1.03	0.15	-0.14	0.35
N03_AMA_NSTS_TU	0.65	0.09	-0.10	0.21
N04_MEN_STS_TU	0.46	0.07	-0.06	0.15
N05_ELF_STS_TU	0.49	0.07	-0.07	0.16
N06_SID_NSTS_TU	1.05	0.15	-0.15	0.35
N07_MEN_NSTS_TU	1.28	0.18	-0.19	0.41
N08_CHI_NSTS_TU	3.36	0.22	-1.12	0.56
N09_CHI_STS_TU	1.61	0.21	-0.32	0.48
N10_SID_NSTS_TU	0.70	0.07	-0.17	0.17
N11_SID_STS_TU	0.49	0.05	-0.11	0.12

Erosion Resistance

In general, STS practices demonstrate a higher soil conservation capacity than non-STS, with lower erosion and lower economic costs of restoration. However, in some countries or specific contexts, the difference between the two practices is less marked or even reversed.

In Italy, data show a varied behavior. The STS practices generally show a lower level of soil loss compared to non-STS, but in some cases, non-STS practices have slightly better erosion resistance values, though with much higher restoration costs. However, in some cases, non-STS practices show slightly better erosion resistance values, but with much higher restoration costs. This suggests that, although non-STS may sometimes have superior erosion resistance, the economic cost associated with their impact is generally higher.

In Greece, STS practices are generally more effective in soil conservation than non-STS. While STS farms often show lower soil loss, in some cases they exhibit higher soil loss values than non-STS, though restoration costs remain lower for STS. This is reflected in the economic costs, which are significantly lower for STS than for non-STS. However, some non-STS sites show similar erosion resistance values to STS, suggesting that under some environmental and management conditions the differences may be reduced.

In Spain, the comparison between the two practices shows varying results in terms of erosion resistance, with some STS farms exhibiting higher soil loss than their non-STS counterparts. However, restoration costs are generally lower for STS farms, confirming the potential economic benefits of these practices. Nevertheless, in some specific cases, the differences between the two practices are less evident.

In Portugal, the differences between STS and non-STS are less pronounced than in other countries. STS practices demonstrate greater resistance to erosion than their non-STS counterparts, but variability in the data suggests that in some areas the two practices may have a similar impact. The economic benefit of STS is manifested in cases where the erosion reduction is more marked, leading to a reduction in operating costs.

In Morocco, the comparison between STS and non-STS is more mixed. While STS farms generally exhibit lower soil loss compared to non-STS farms, in some cases non-STS farms show comparable or even better erosion resistance. Restoration costs vary significantly between the two practices, with STS farms often showing lower costs, but exceptions exist. These results suggest that the local environmental context significantly influences the effectiveness of the strategies adopted.

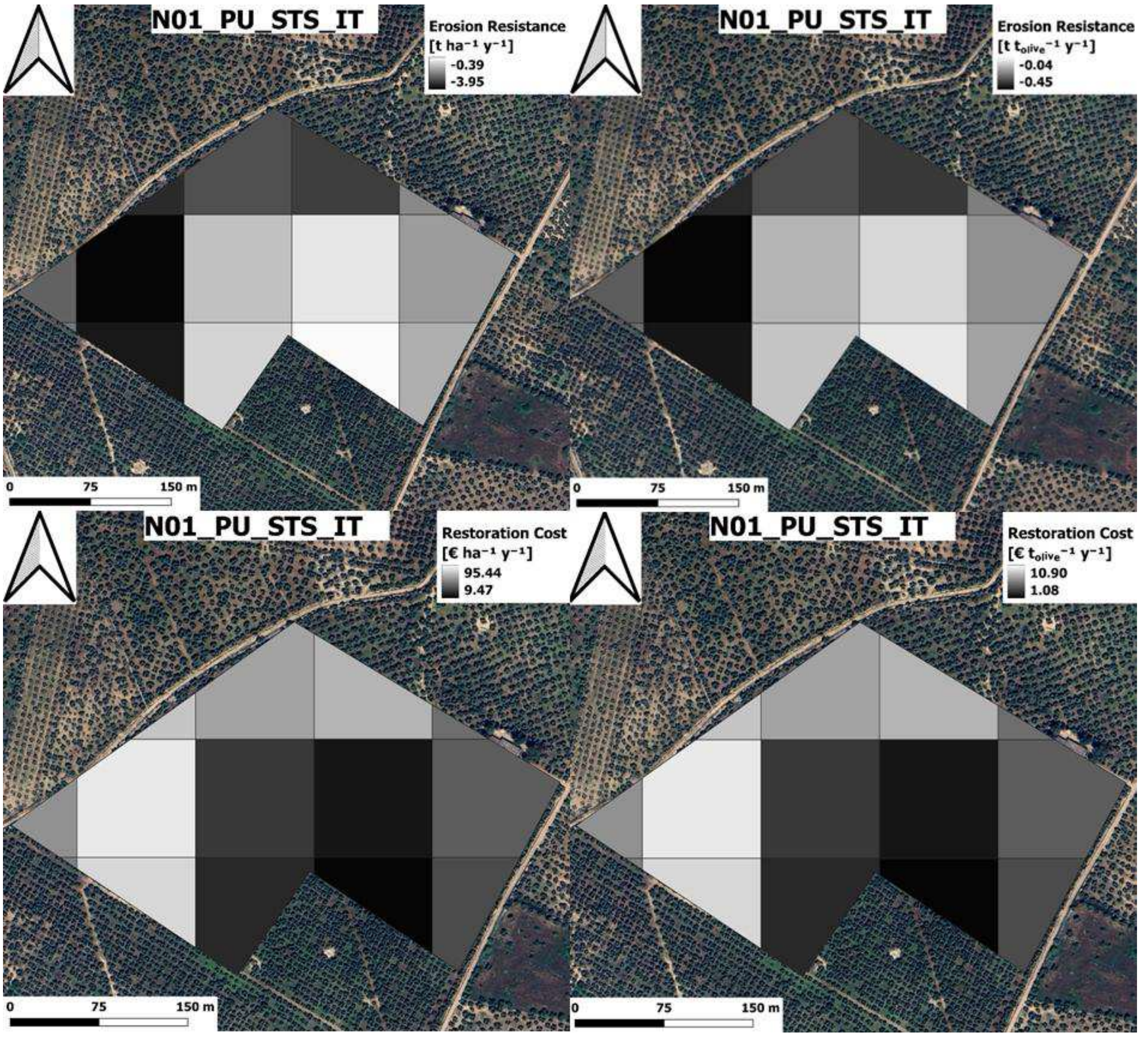
In Tunisia, the distinction between STS and non-STS is less clear-cut than in other countries, but STS practices demonstrate on average greater resistance to erosion than non-STS practices. Restoration costs are higher for non-STS practices, confirming that the increased soil loss has a significant economic impact.

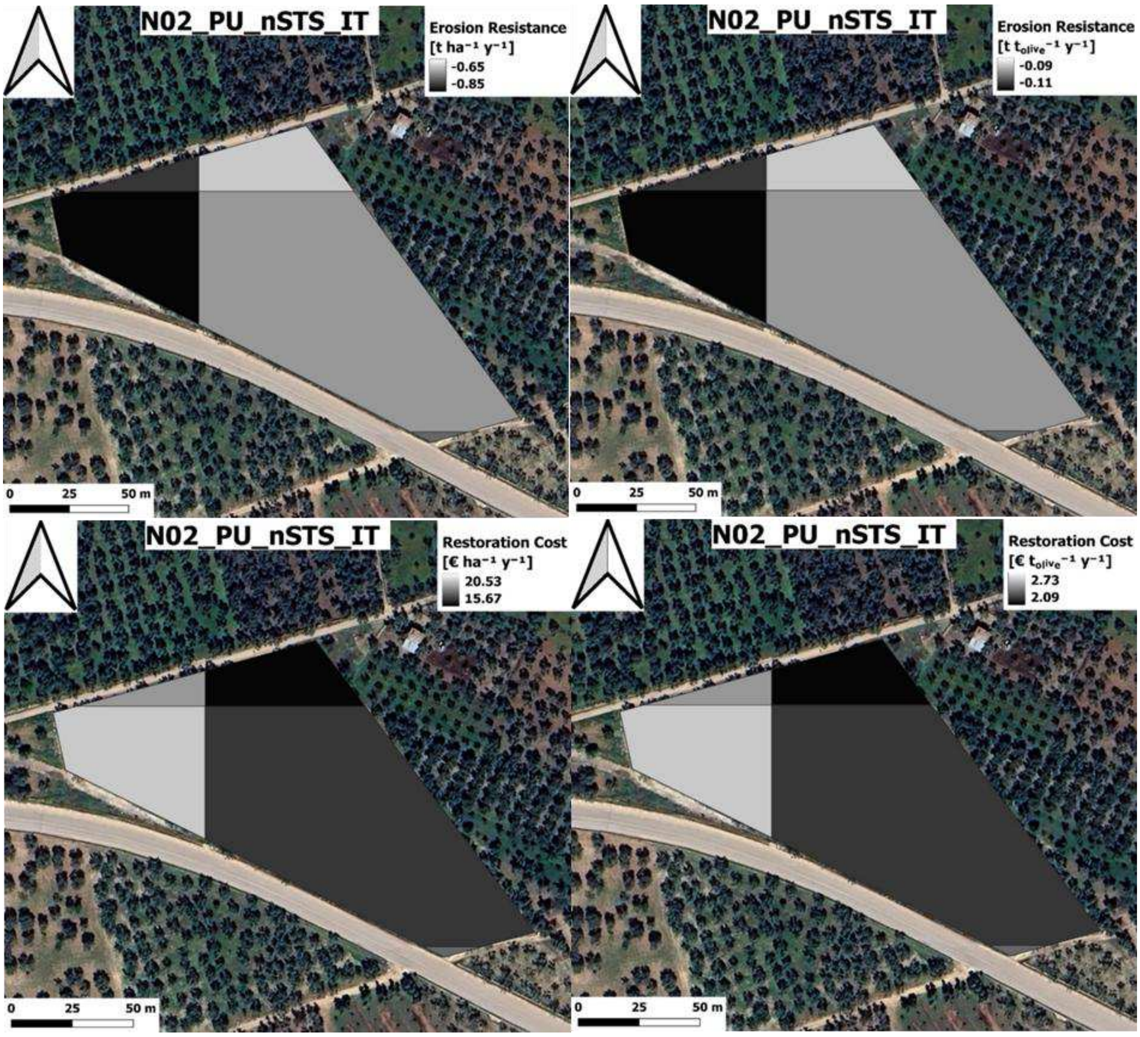
At the inter-country level, data show that Greece and Spain benefit more from STS practices, with reduced erosion and lower restoration costs. In Italy and Morocco, the balance between the two practices is more balanced, while in Portugal and Tunisia the differences between STS and non-STS are less marked, although confirming a slight superiority of STS.

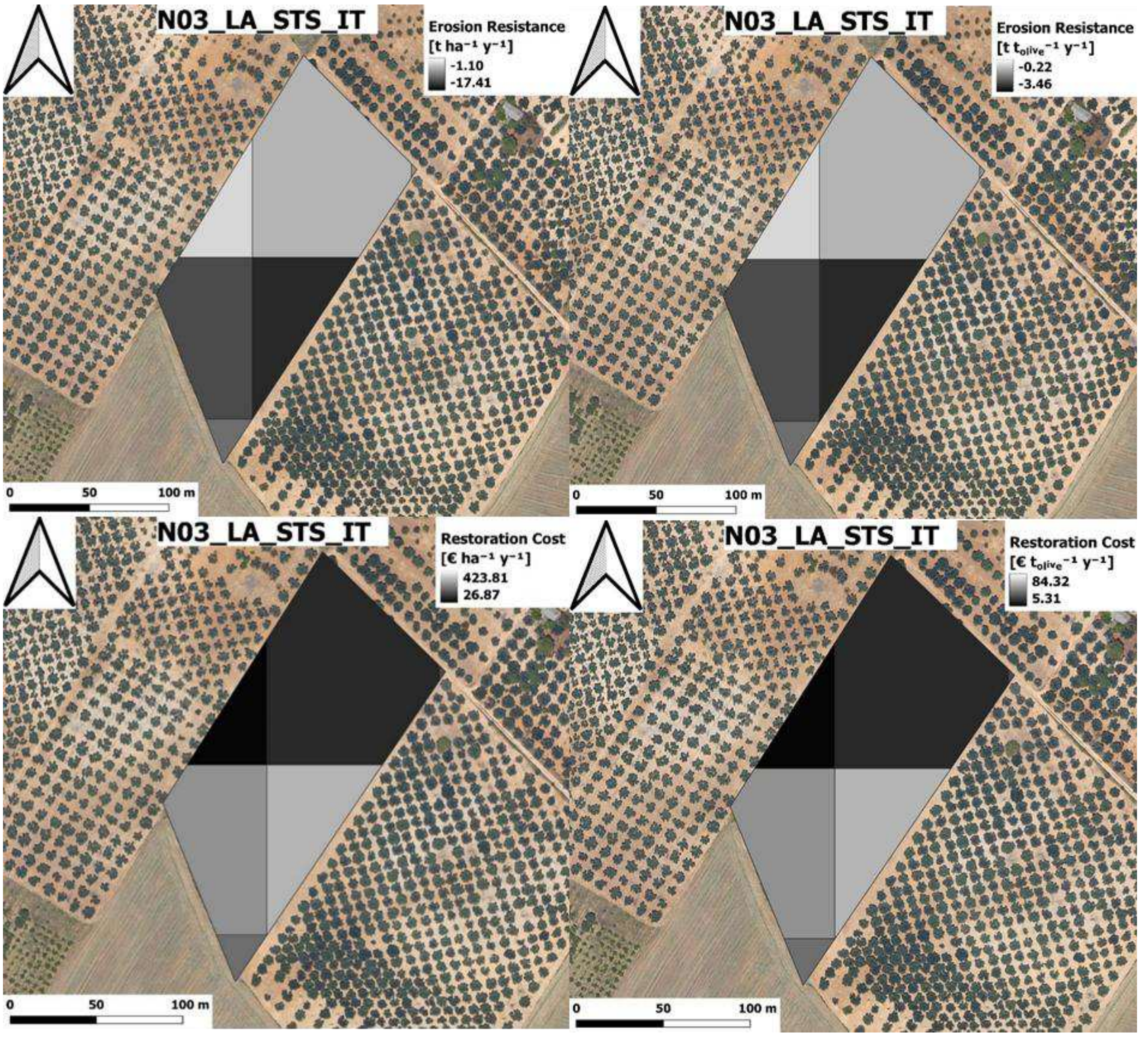
These results suggest that soil management and local environmental conditions strongly influence the effectiveness of the practices adopted. Although in many cases STS practices demonstrate a clear superiority in containing erosion, in others the differences are less evident or even reversed, depending on the context and the indicator considered. Maps that show environmental and economic outcomes are reported in the following sub-paragraphs.

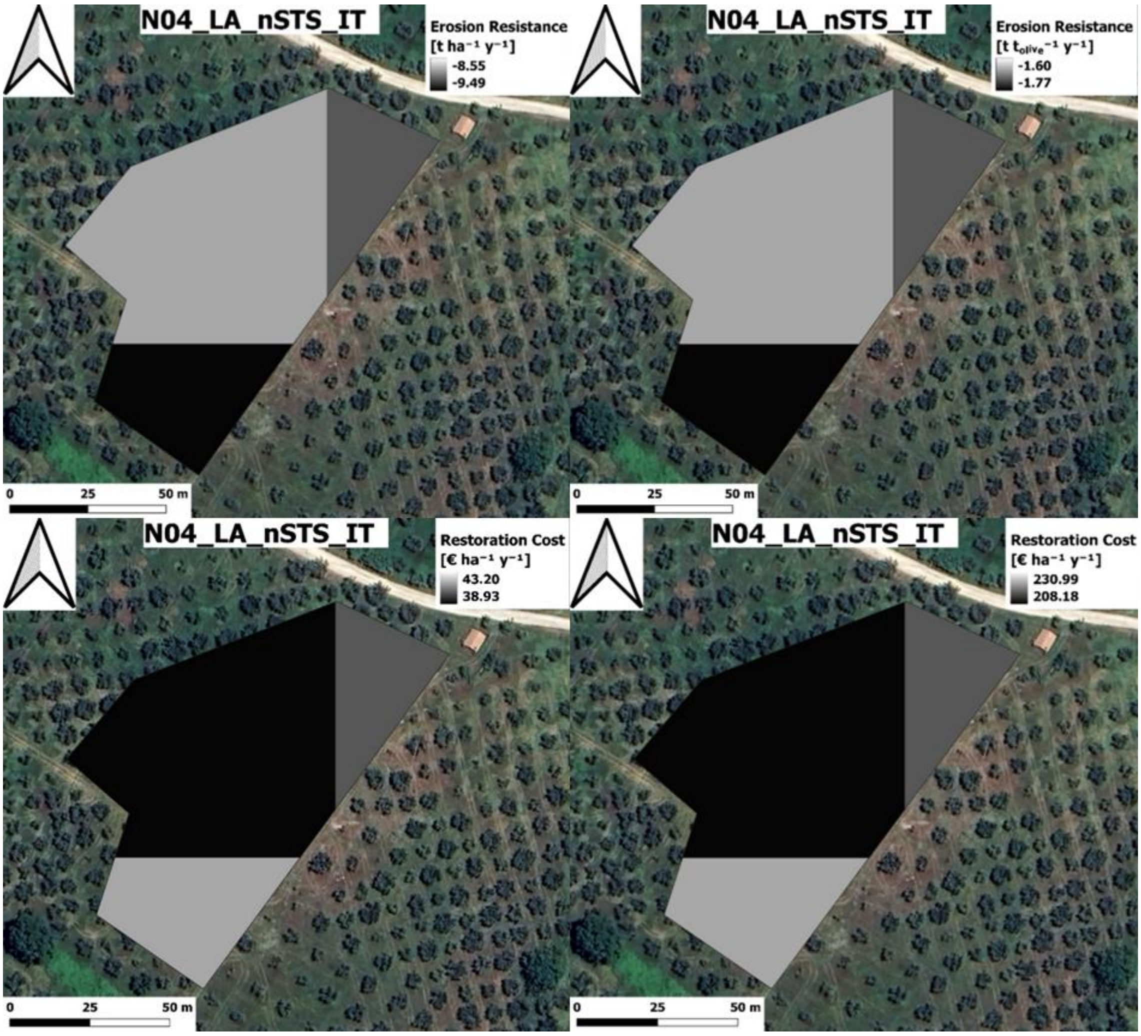
Italy

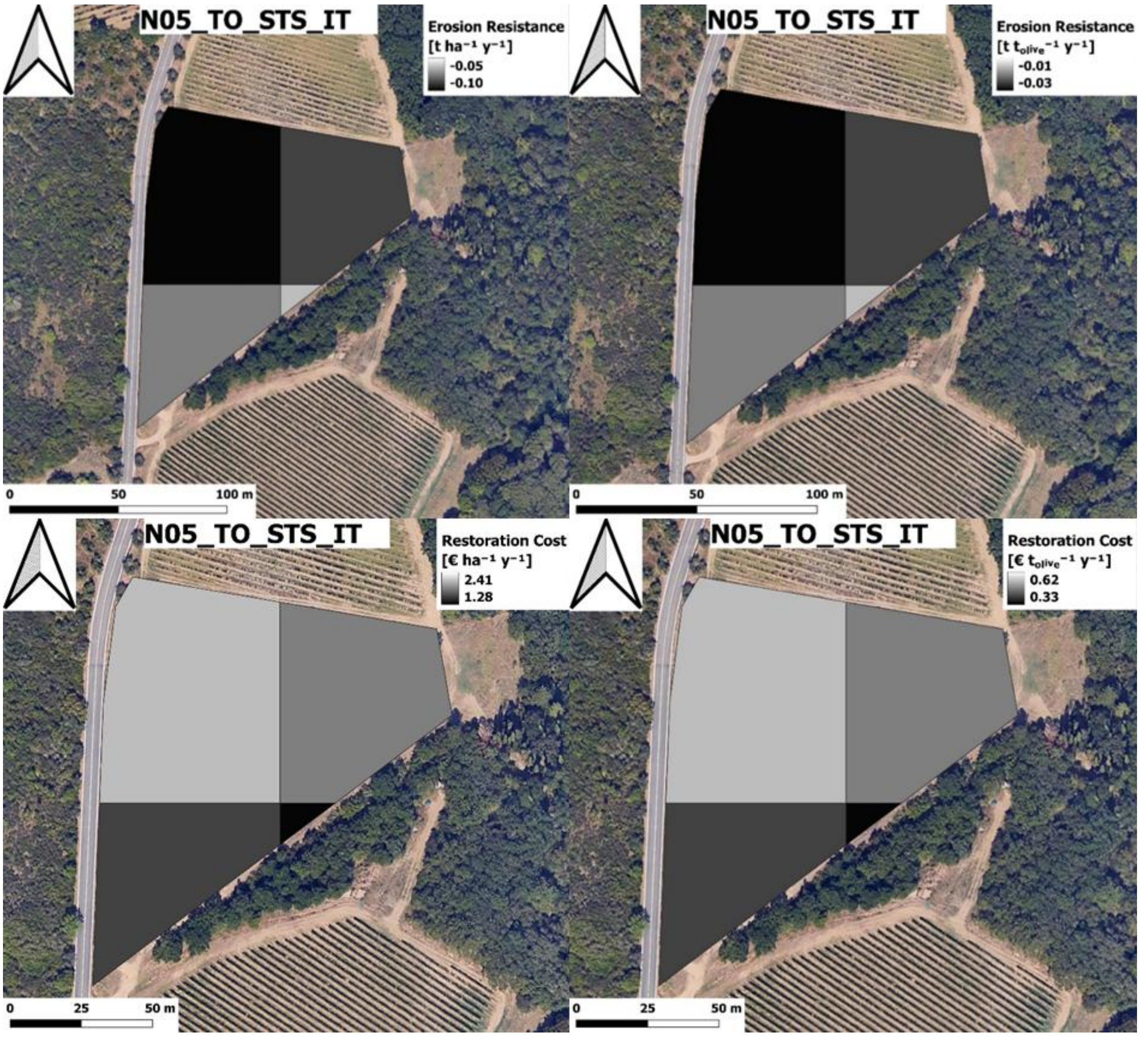
In this section are collected Erosion Resistance maps for Italy. The Coordinate Reference System (CSR) is WGS-84 UTM 33N for all olive farms except for N05_TO_STS_IT, N06_TO_nSTS_IT and N09_LI_STS_IT which the CSR is WGS-84 UTM 32N.

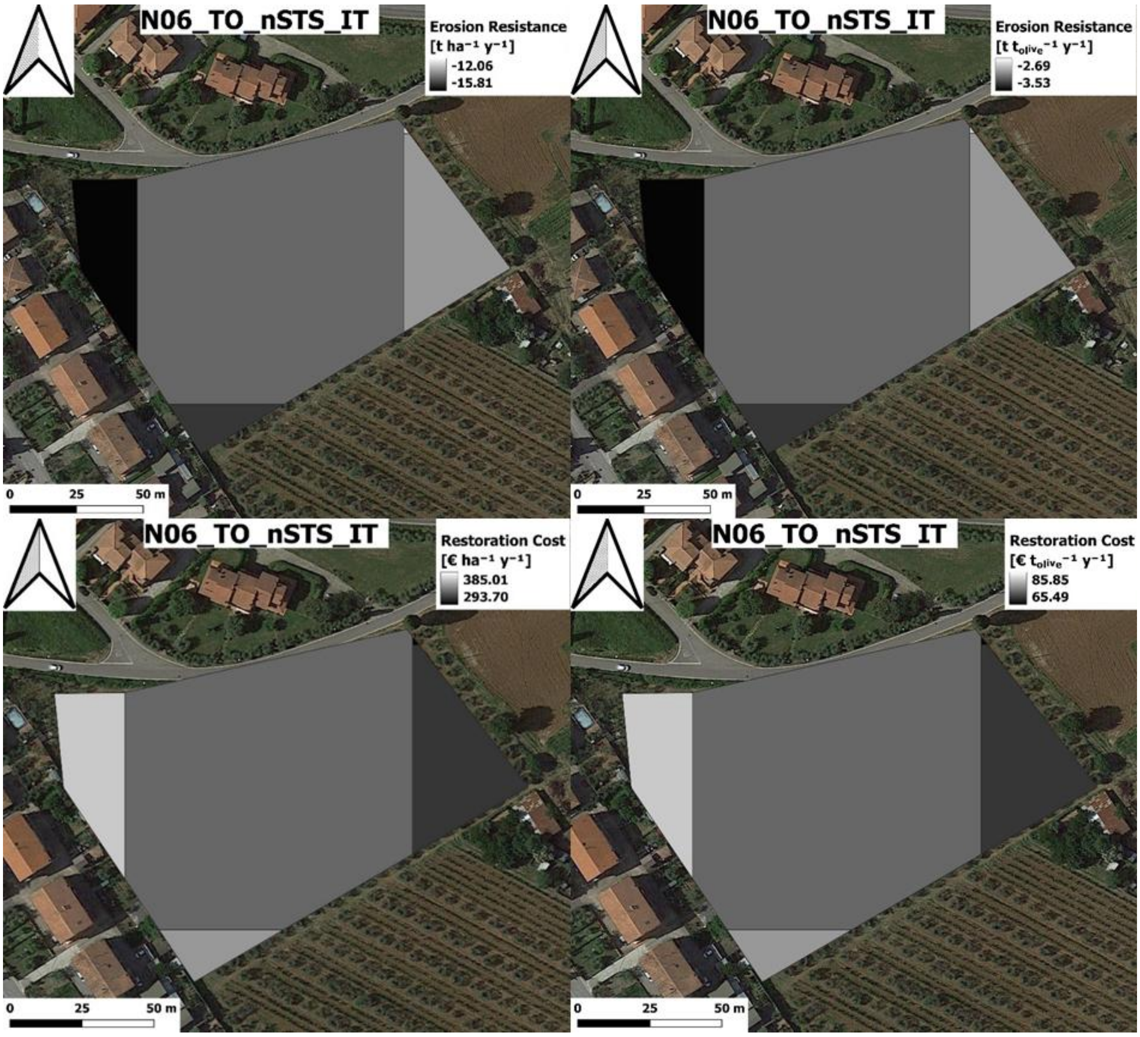

N01_PU_STS_IT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.39
-3.95
0
75
150 m
N01_PU_STS_IT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.04
-0.45
N01_PU_STS_IT
Restoration Cost
[€ ha⁻¹ y⁻¹]
95.44
9.47
N01_PU_STS_IT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
10.90
1.08

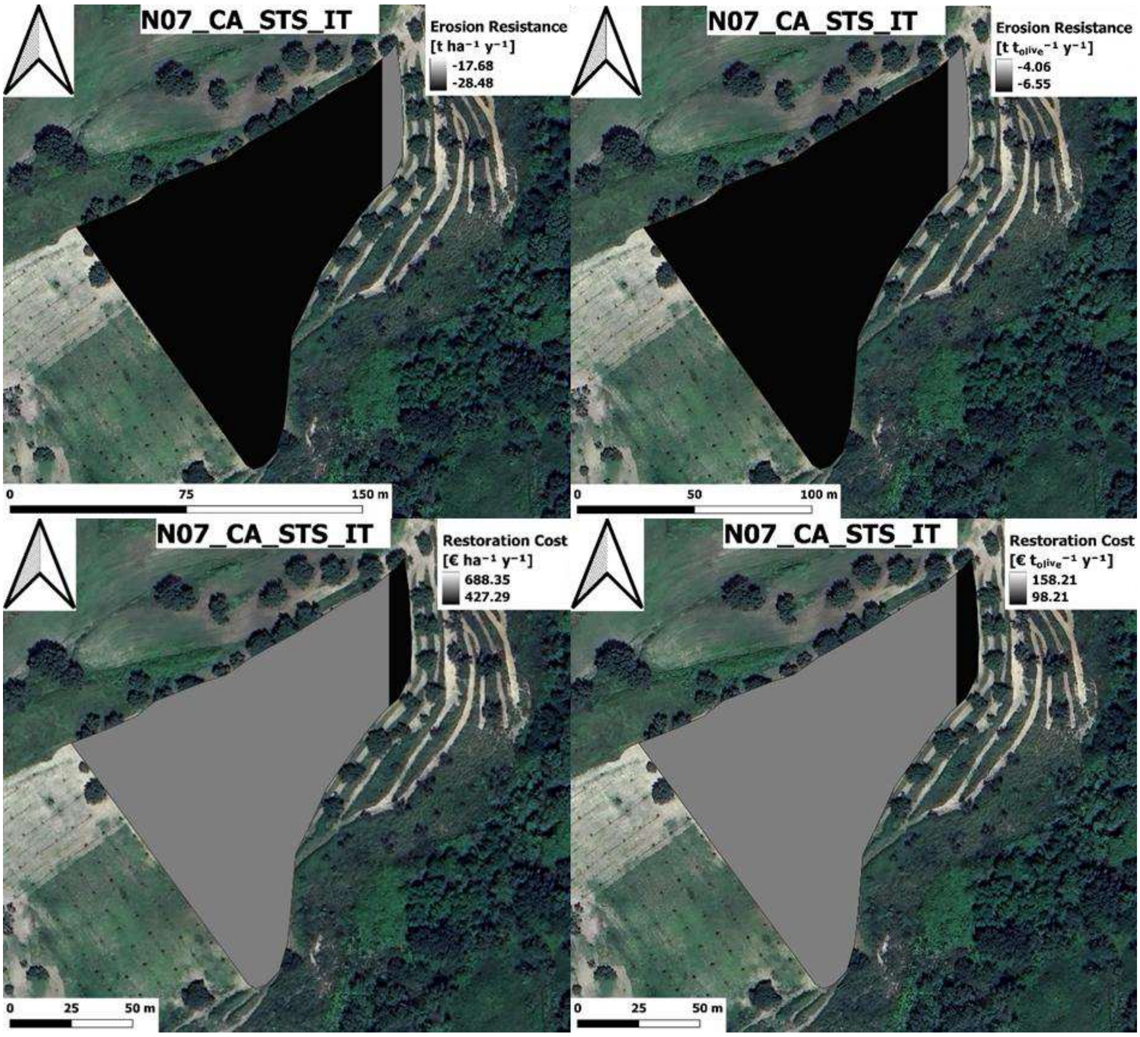

N02_PU_nSTS_IT
Erosion Resistance
[t ha^{-1} y^{-1}]
-0.65
-0.85
0
25
50 m
N02_PU_nSTS_IT
Erosion Resistance
[t t_{olive}^{-1} y^{-1}]
-0.09
-0.11
0
25
50 m
N02_PU_nSTS_IT
Restoration Cost
[€ ha^{-1} y^{-1}]
20.53
15.67
0
25
50 m
N02_PU_nSTS_IT
Restoration Cost
[€ t_{olive}^{-1} y^{-1}]
2.73
2.09
0
25
50 m

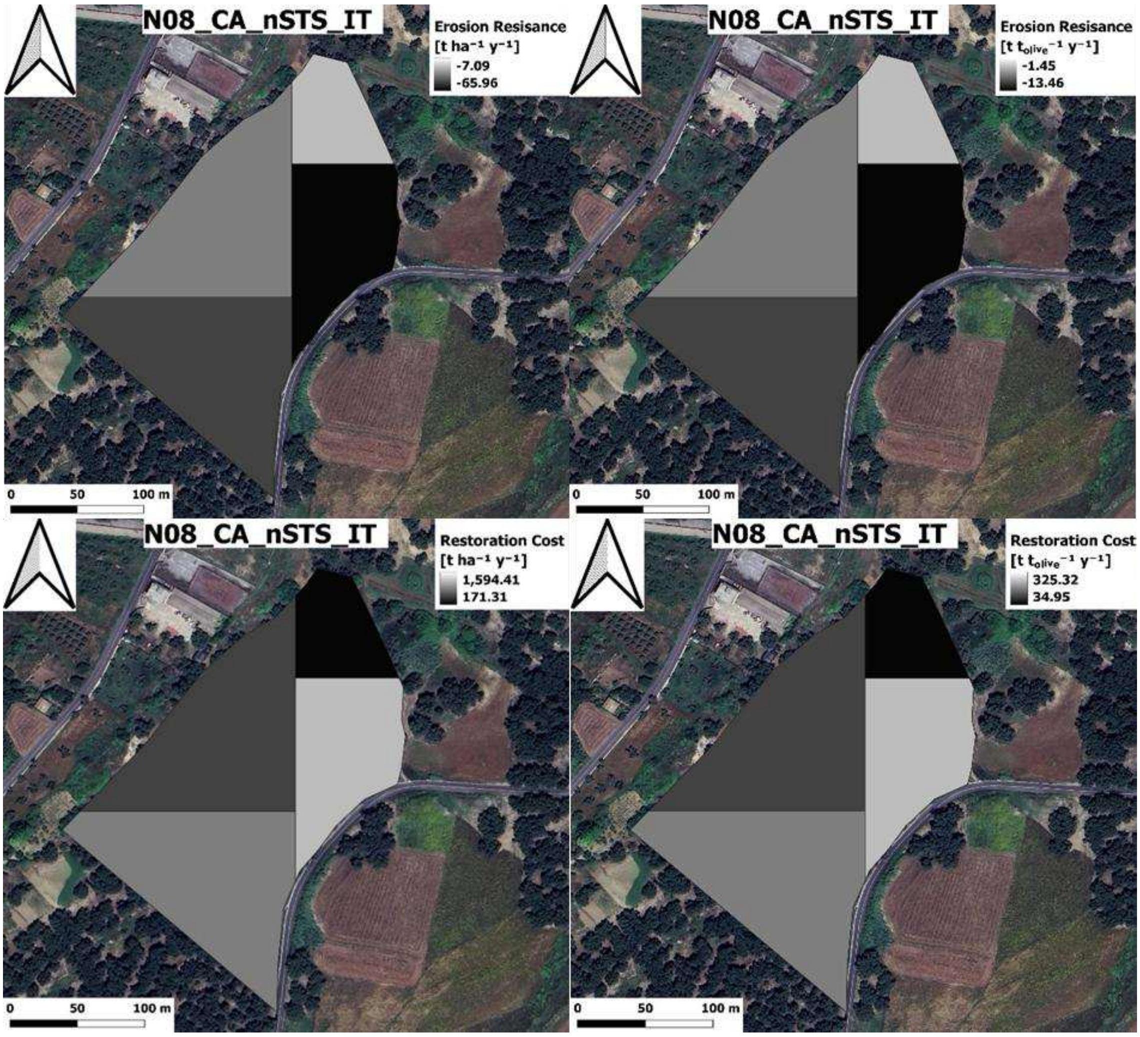

N03_LA_STS_IT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-1.10
-17.41
0 50 100 m
N03_LA_STS_IT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.22
-3.46
0 50 100 m
N03_LA_STS_IT
Restoration Cost
[€ ha⁻¹ y⁻¹]
423.81
26.87
0 50 100 m
N03_LA_STS_IT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
84.32
5.31
0 50 100 m

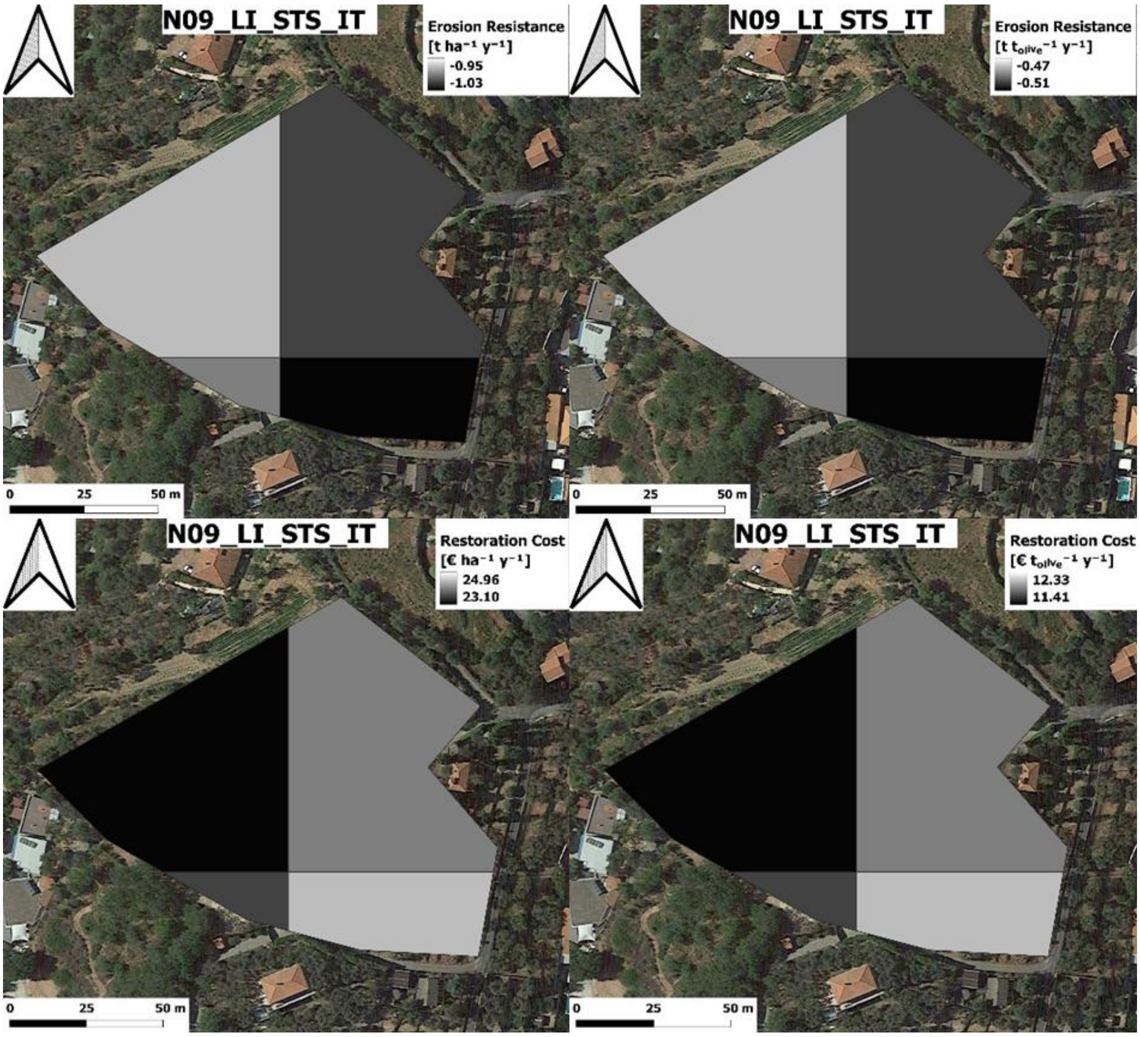

N04_LA_nSTS_IT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-8.55
-9.49
0 25 50 m
N04_LA_nSTS_IT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-1.60
-1.77
0 25 50 m
N04_LA_nSTS_IT
Restoration Cost
[€ ha⁻¹ y⁻¹]
43.20
38.93
0 25 50 m
N04_LA_nSTS_IT
Restoration Cost
[€ ha⁻¹ y⁻¹]
230.99
208.18
0 25 50 m


N05_TO_STS_IT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.05
-0.10
0
50
100 m
N05_TO_STS_IT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.01
-0.03
0
50
100 m
N05_TO_STS_IT
Restoration Cost
[€ ha⁻¹ y⁻¹]
2.41
1.28
0
25
50 m
N05_TO_STS_IT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
0.62
0.33
0
25
50 m


N06_TO_nSTS_IT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-12.06
-15.81
0 25 50 m
N06_TO_nSTS_IT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-2.69
-3.53
0 25 50 m
N06_TO_nSTS_IT
Restoration Cost
[€ ha⁻¹ y⁻¹]
385.01
293.70
0 25 50 m
N06_TO_nSTS_IT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
85.85
65.49
0 25 50 m

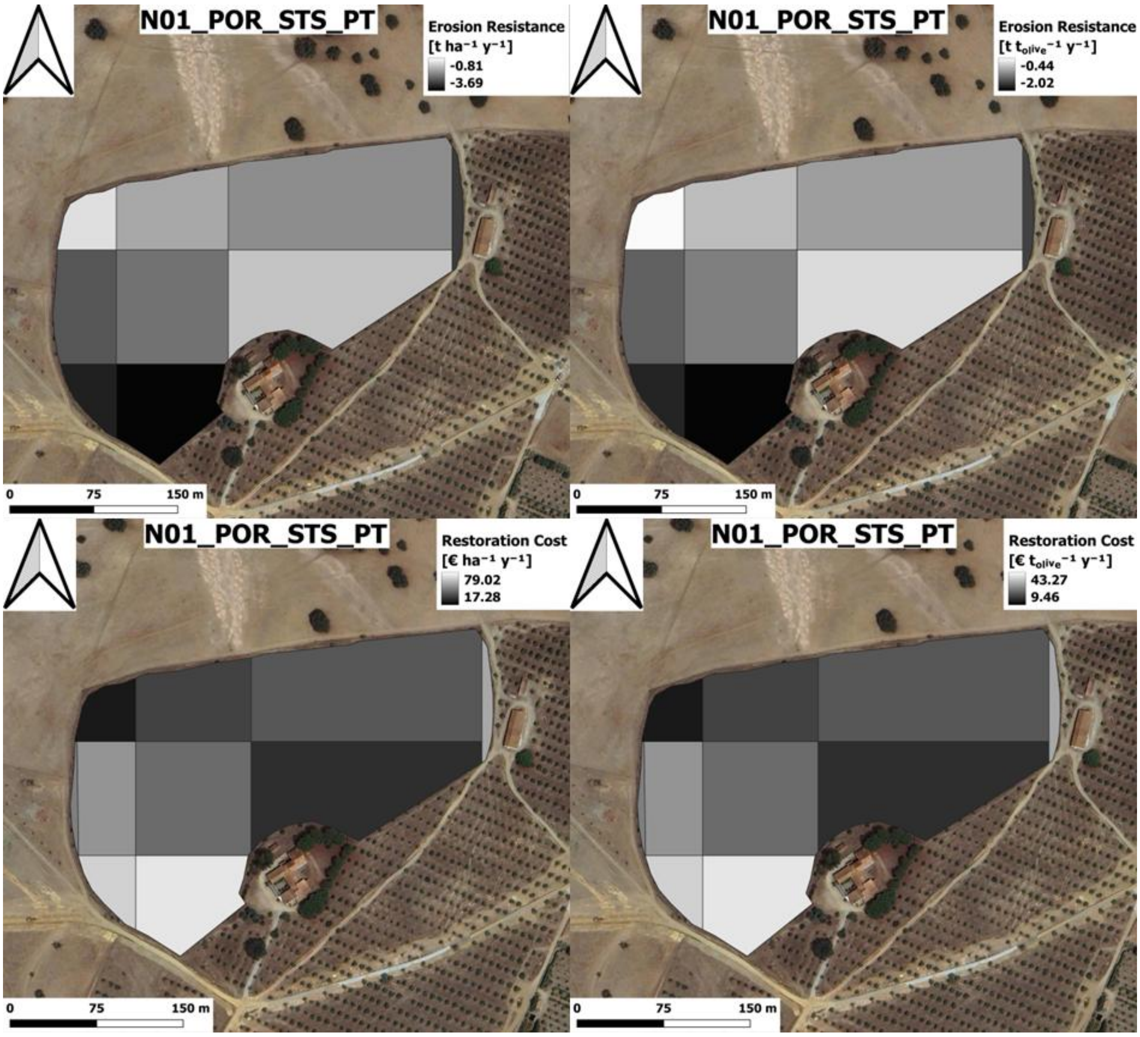

N07_CA_STS_IT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-17.68
-28.48
0
75
150 m
N07_CA_STS_IT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-4.06
-6.55
0
50
100 m
N07_CA_STS_IT
Restoration Cost
[€ ha⁻¹ y⁻¹]
688.35
427.29
0
25
50 m
N07_CA_STS_IT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
158.21
98.21
0
25
50 m

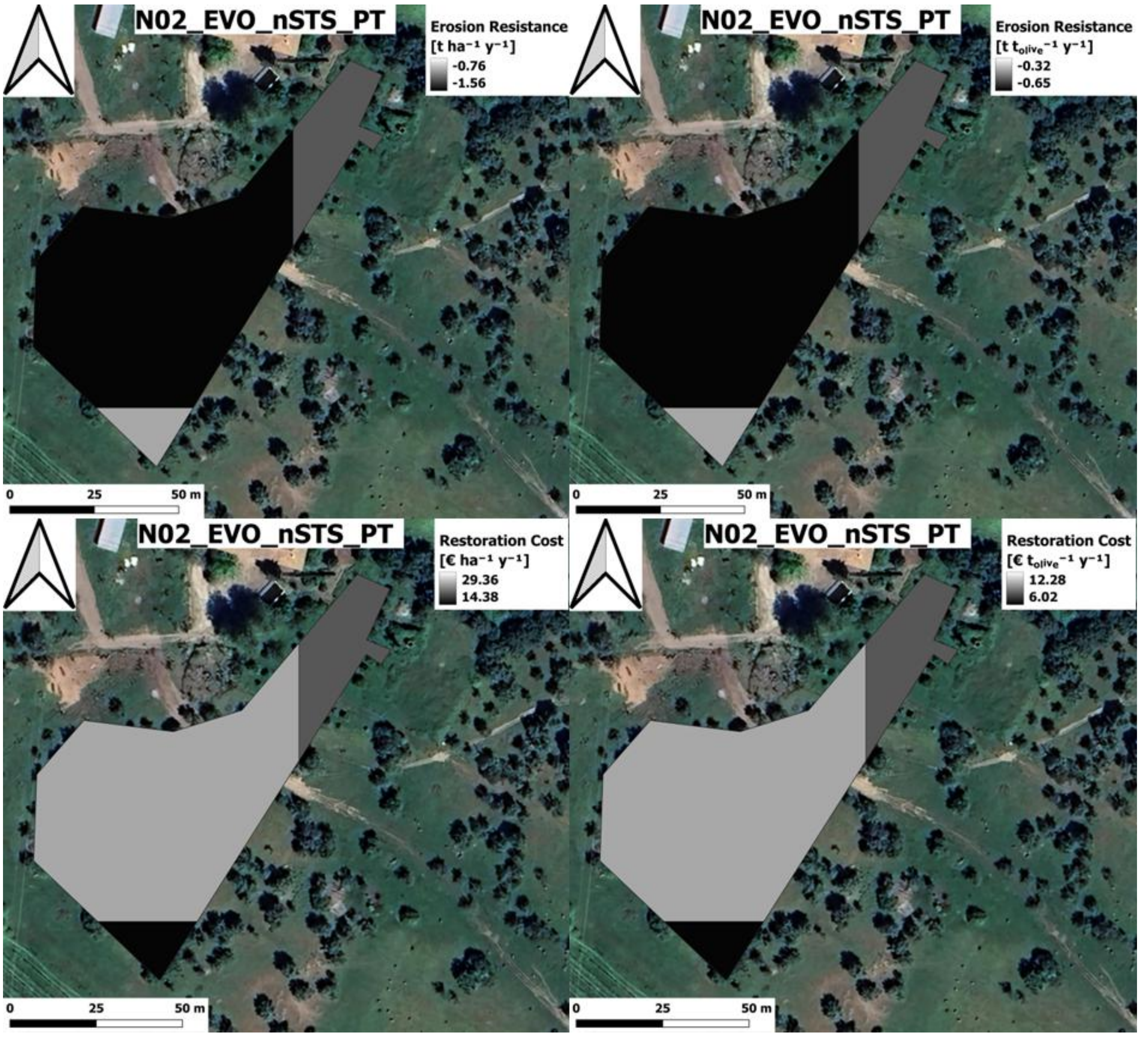

N08_CA_nSTS_IT
Erosion Resisance
[t ha⁻¹ y⁻¹]
-7.09
-65.96
0 50 100 m
N08_CA_nSTS_IT
Erosion Resisance
[t t_olive⁻¹ y⁻¹]
-1.45
-13.46
0 50 100 m
N08_CA_nSTS_IT
Restoration Cost
[t ha⁻¹ y⁻¹]
1,594.41
171.31
0 50 100 m
N08_CA_nSTS_IT
Restoration Cost
[t t_olive⁻¹ y⁻¹]
325.32
34.95
0 50 100 m

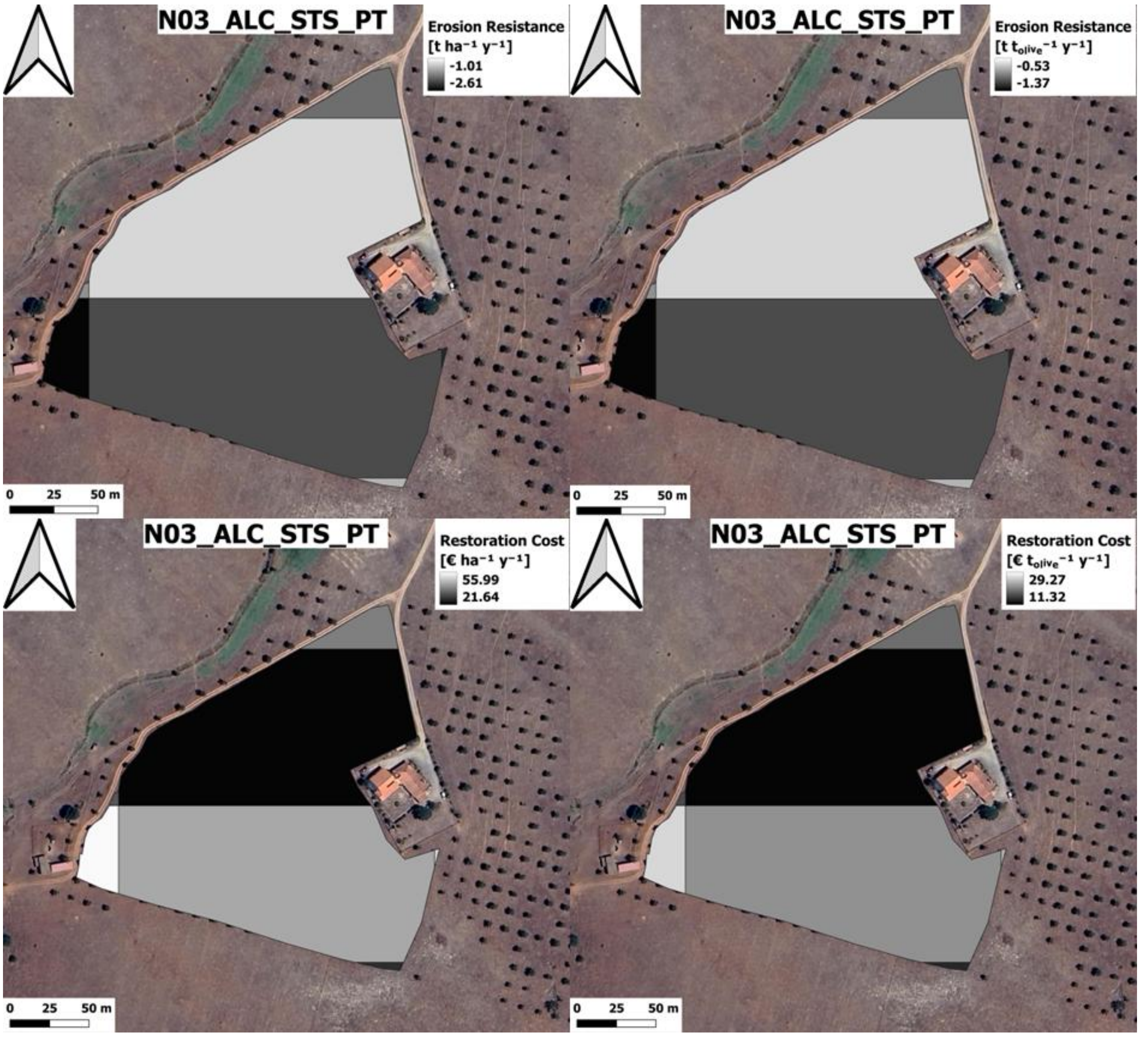

N09_LI_STS_IT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.95
-1.03
0 25 50 m
N09_LI_STS_IT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.47
-0.51
0 25 50 m
N09_LI_STS_IT
Restoration Cost
[€ ha⁻¹ y⁻¹]
24.96
23.10
0 25 50 m
N09_LI_STS_IT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
12.33
11.41
0 25 50 m

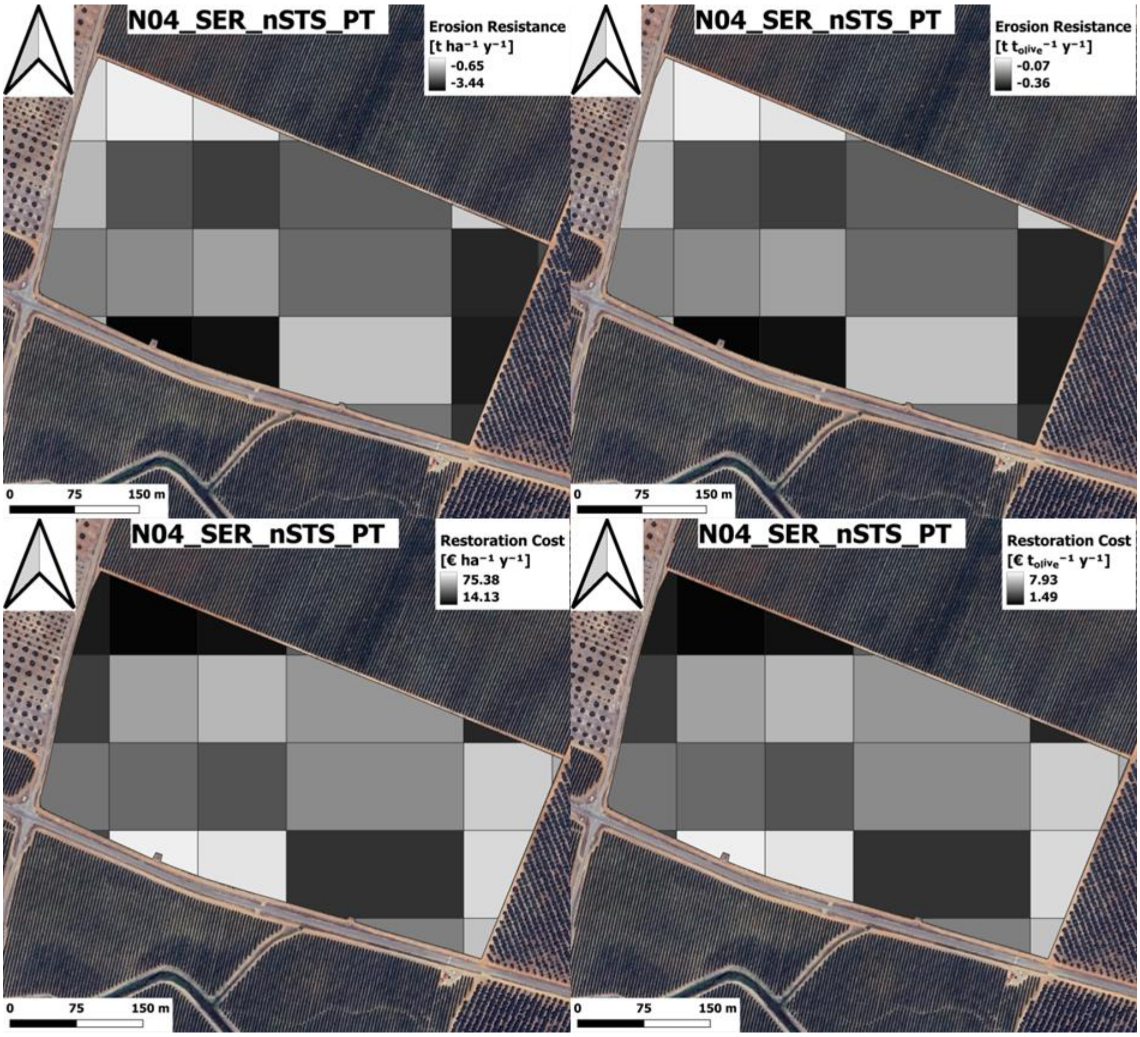
Portugal

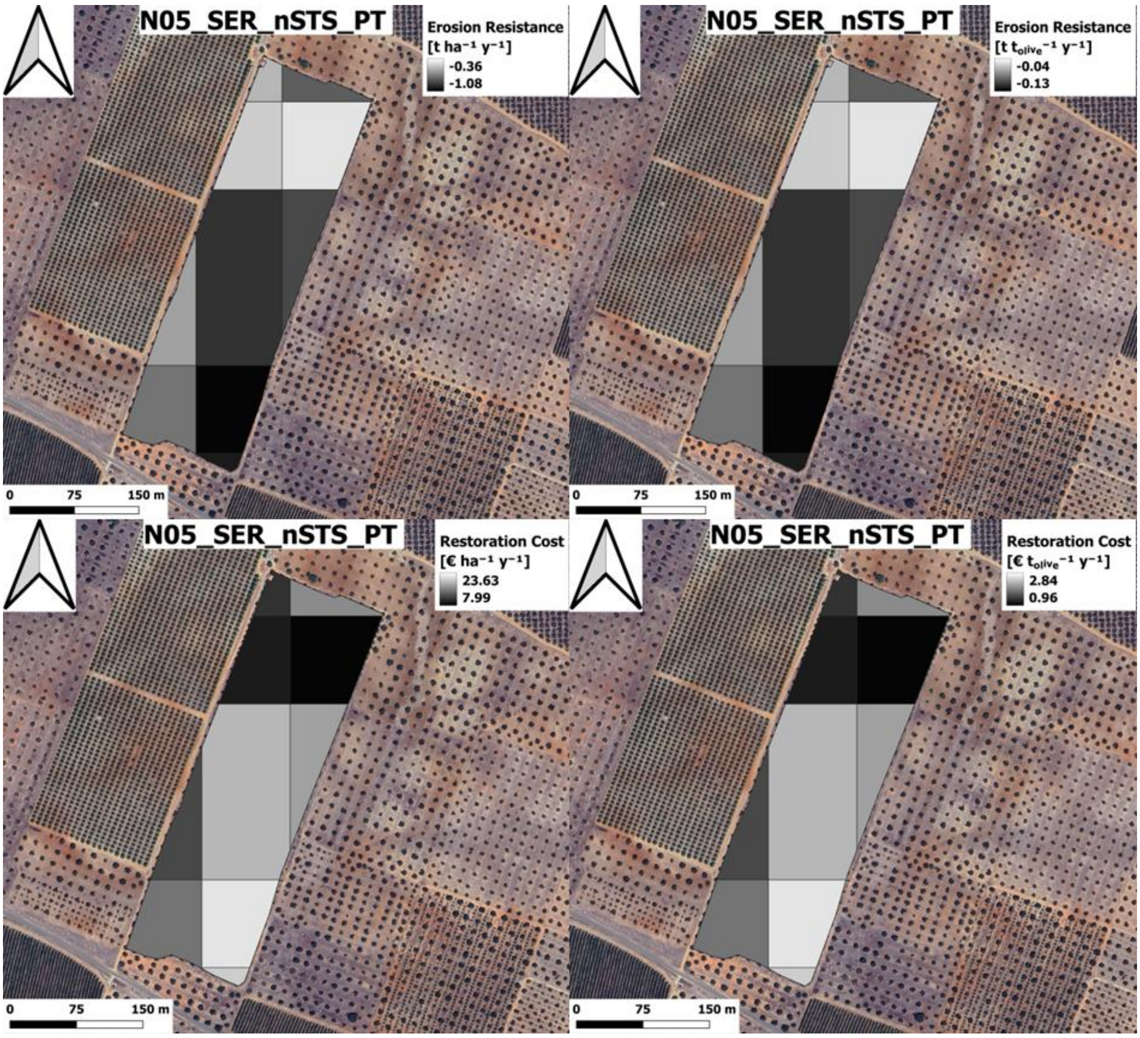
In this section are collected Erosion Resistance maps for Portugal. The Coordinate Reference System (CSR) is WGS-84 UTM 29N for all olive farms.

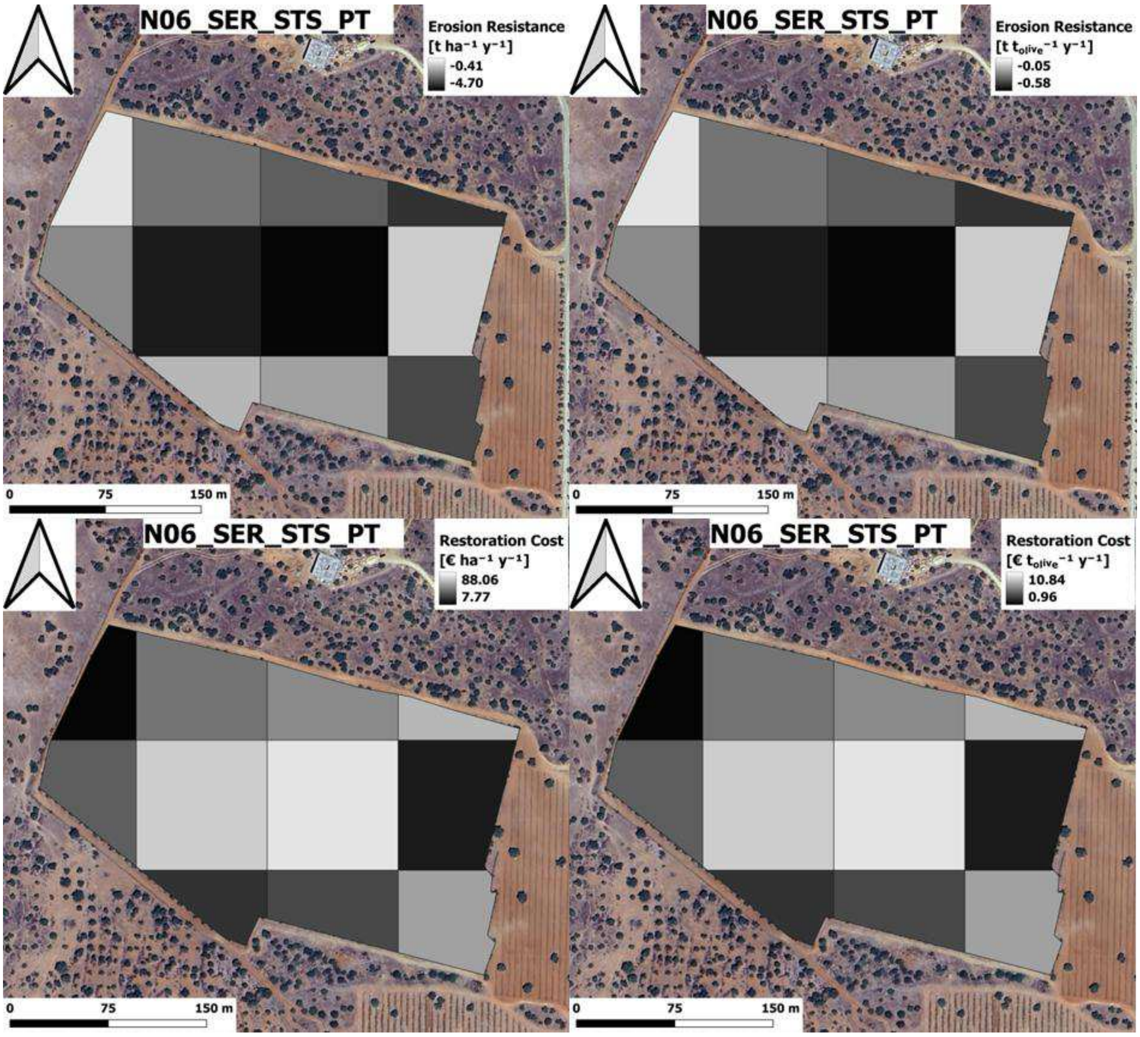



N02_EVO_nSTS_PT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.76
-1.56
0 25 50 m
N02_EVO_nSTS_PT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.32
-0.65
0 25 50 m
N02_EVO_nSTS_PT
Restoration Cost
[€ ha⁻¹ y⁻¹]
29.36
14.38
0 25 50 m
N02_EVO_nSTS_PT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
12.28
6.02
0 25 50 m


N03_ALC_STS_PT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-1.01
-2.61
0 25 50 m
N03_ALC_STS_PT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.53
-1.37
0 25 50 m
N03_ALC_STS_PT
Restoration Cost
[€ ha⁻¹ y⁻¹]
55.99
21.64
0 25 50 m
N03_ALC_STS_PT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
29.27
11.32
0 25 50 m

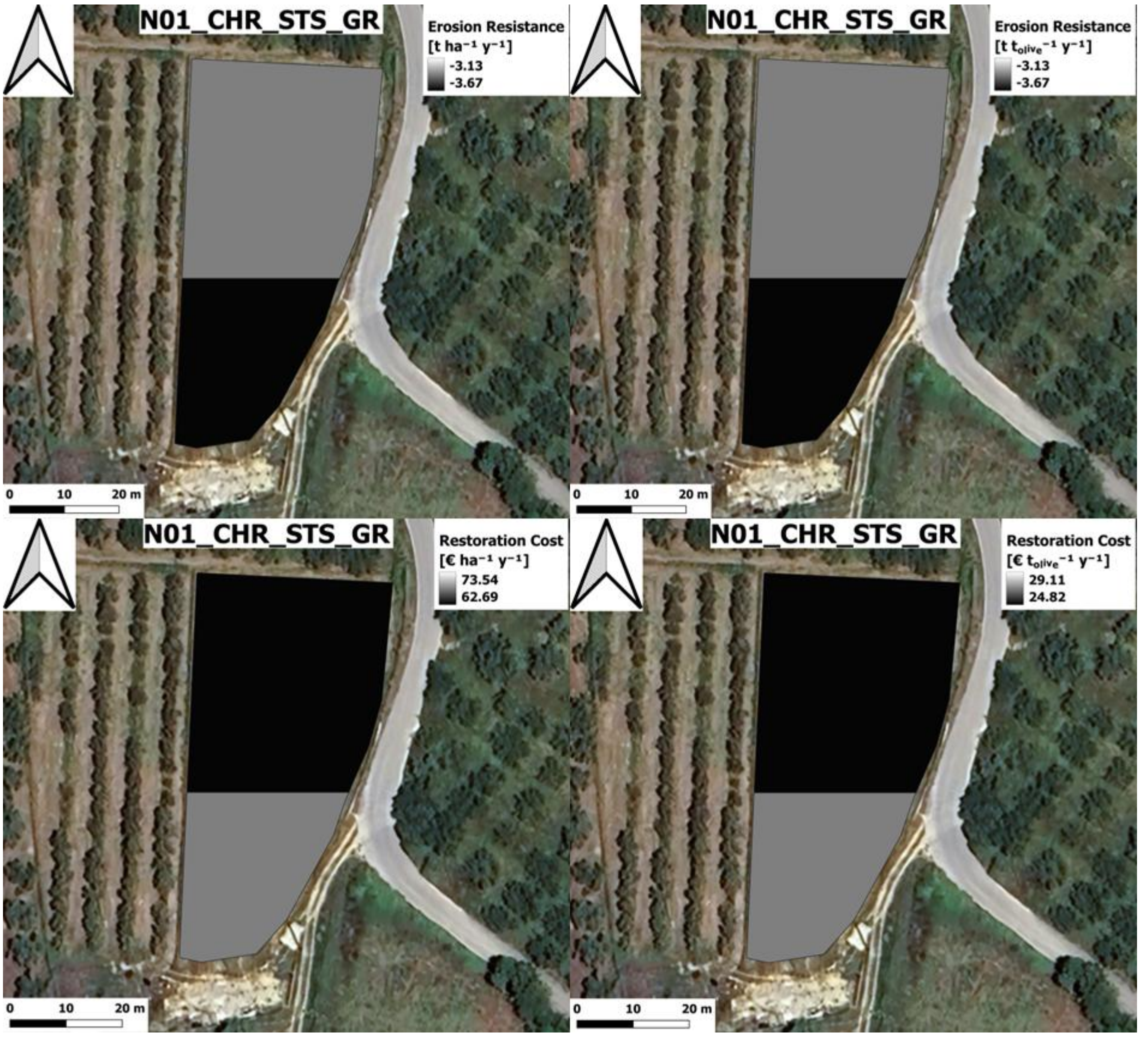

N04_SER_nSTS_PT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.65
-3.44
0 75 150 m
N04_SER_nSTS_PT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.07
-0.36
0 75 150 m
N04_SER_nSTS_PT
Restoration Cost
[€ ha⁻¹ y⁻¹]
75.38
14.13
0 75 150 m
N04_SER_nSTS_PT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
7.93
1.49
0 75 150 m

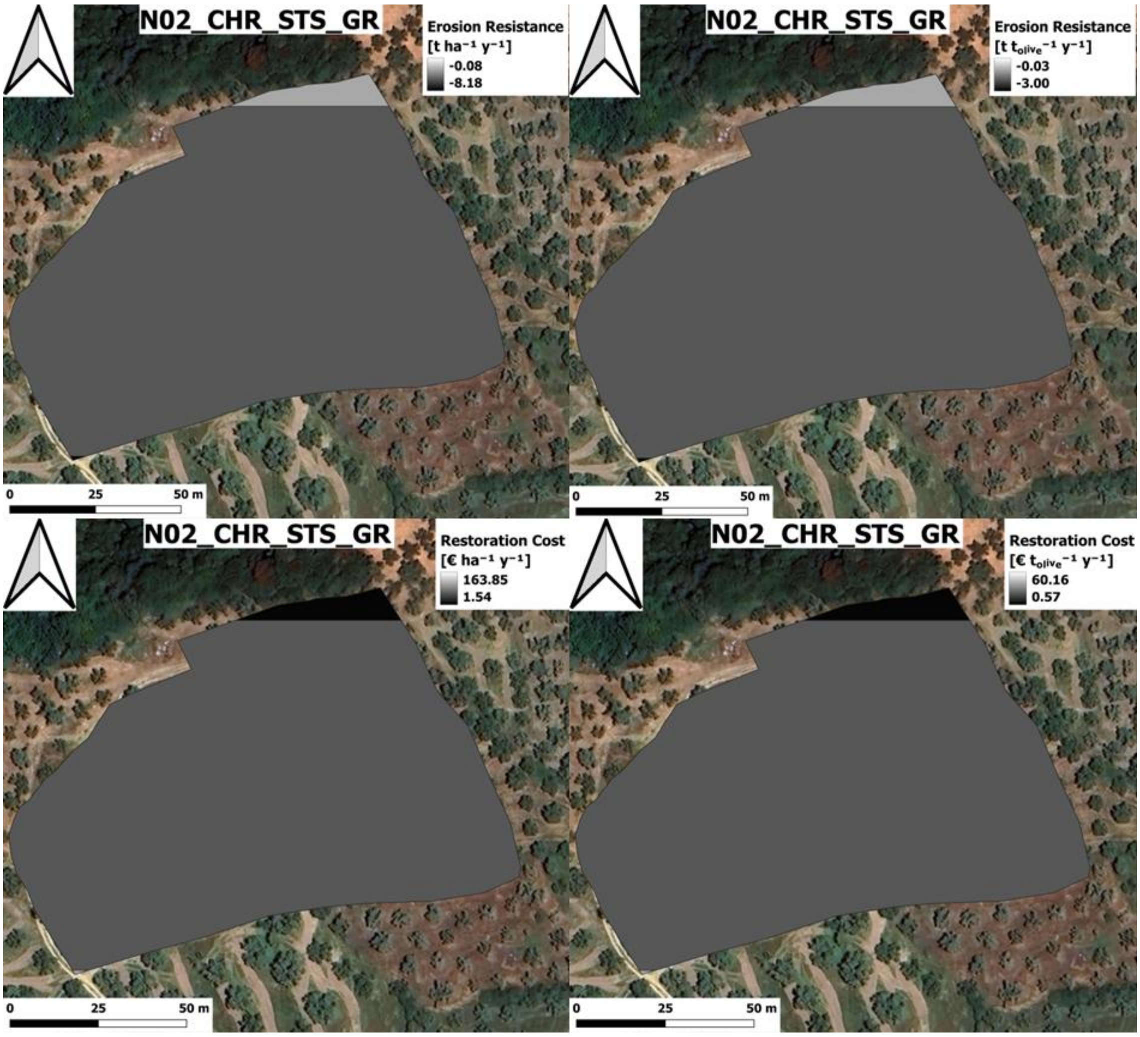

N05_SER_nSTS_PT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.36
-1.08
0
75
150 m
N05_SER_nSTS_PT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.04
-0.13
0
75
150 m
N05_SER_nSTS_PT
Restoration Cost
[€ ha⁻¹ y⁻¹]
23.63
7.99
0
75
150 m
N05_SER_nSTS_PT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
2.84
0.96
0
75
150 m

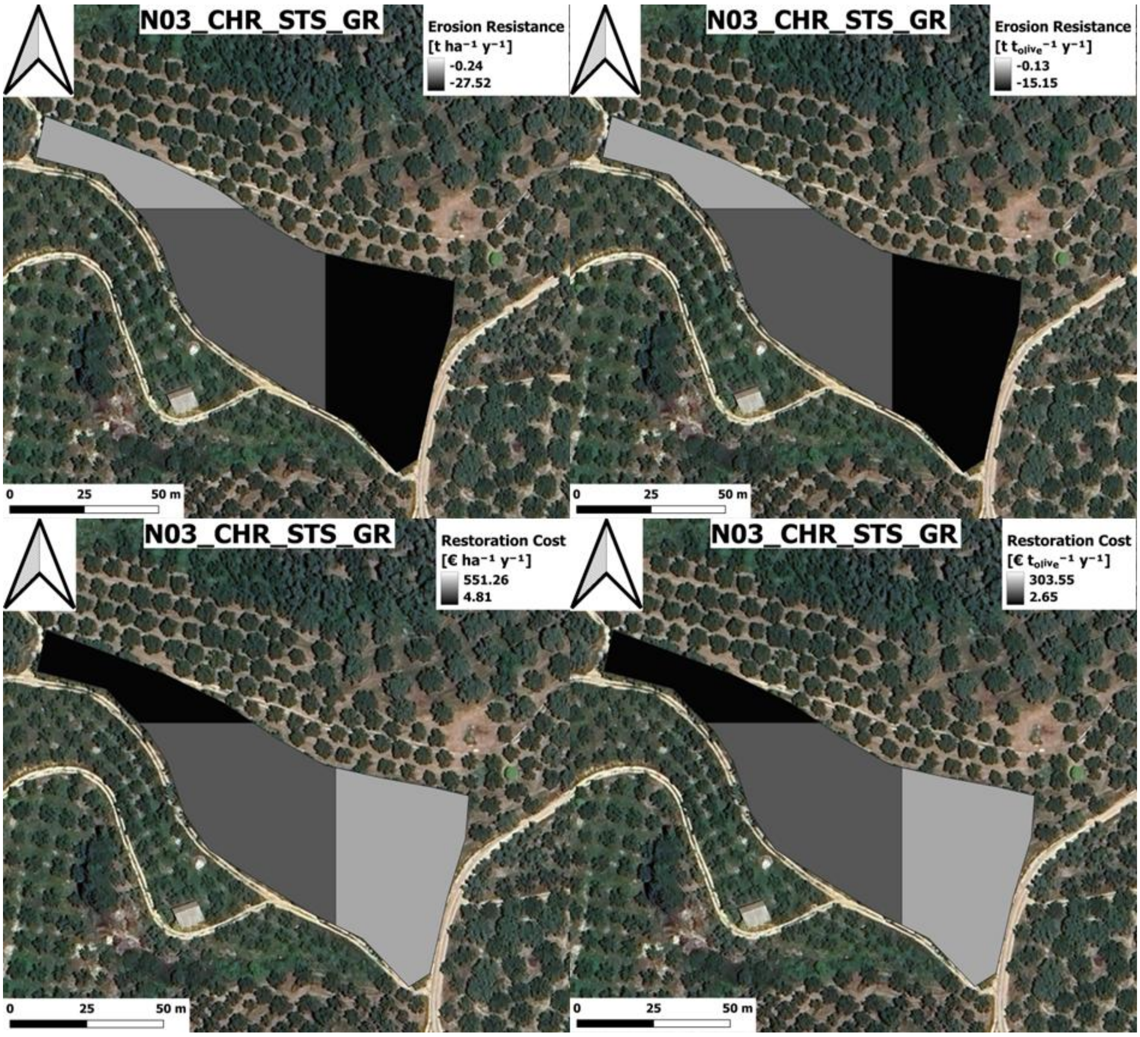

N06_SER_STS_PT
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.41
-4.70
0 75 150 m
N06_SER_STS_PT
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.05
-0.58
0 75 150 m
N06_SER_STS_PT
Restoration Cost
[€ ha⁻¹ y⁻¹]
88.06
7.77
0 75 150 m
N06_SER_STS_PT
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
10.84
0.96
0 75 150 m

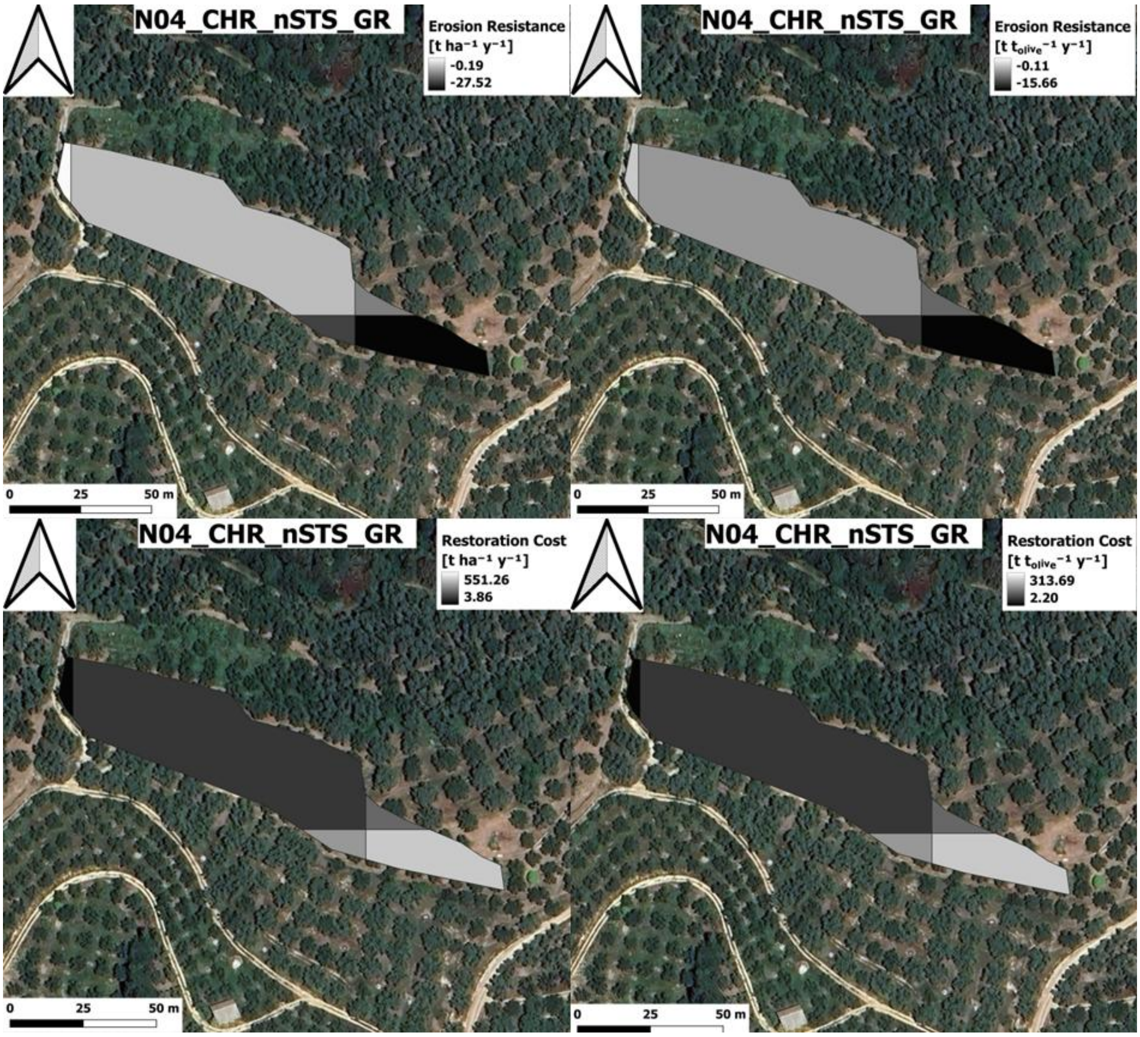
Greece

In this section are collected Erosion Resistance maps for Italy. The Coordinate Reference System (CSR) is WGS-84 UTM 34N for all olive farms.

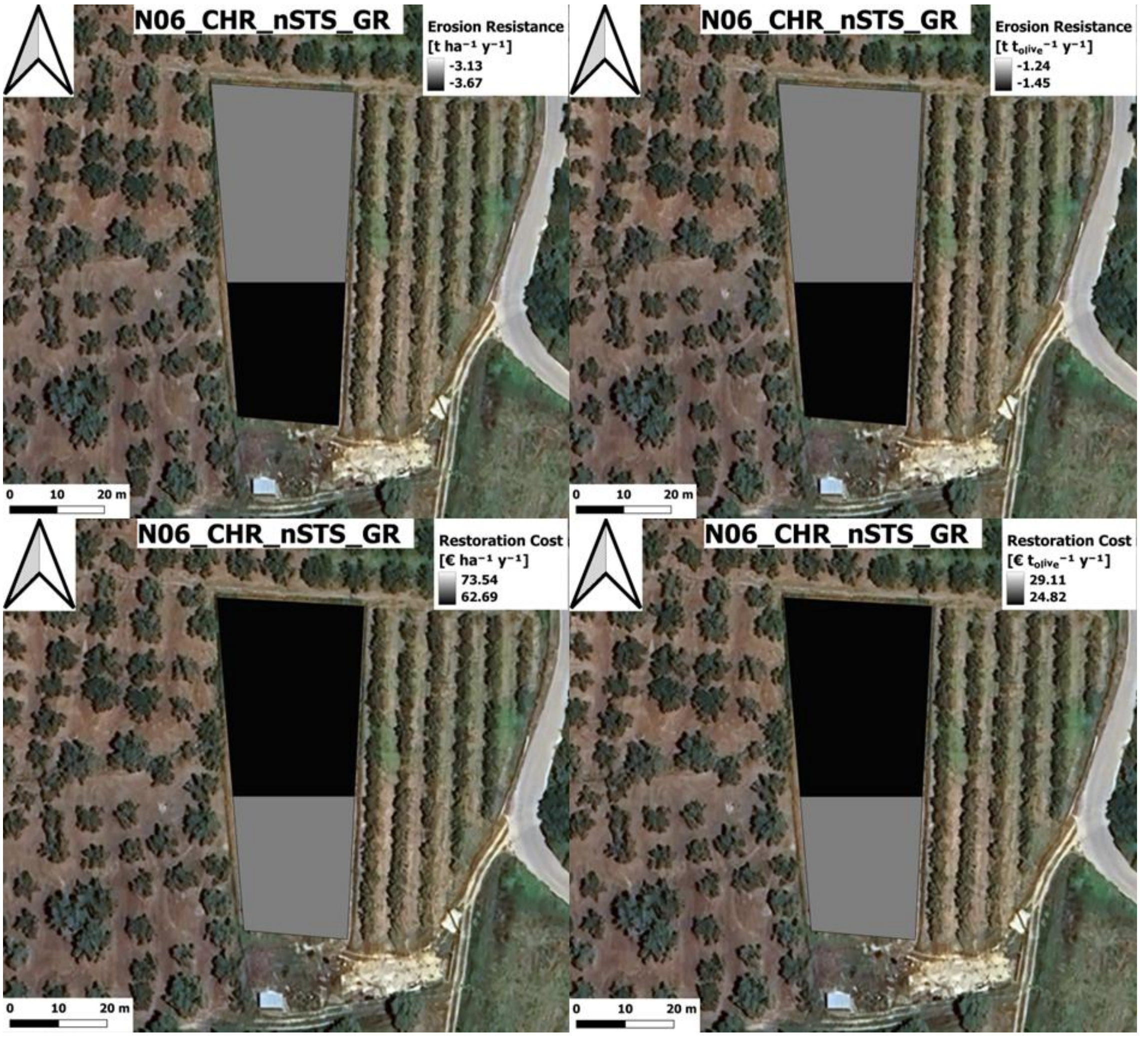


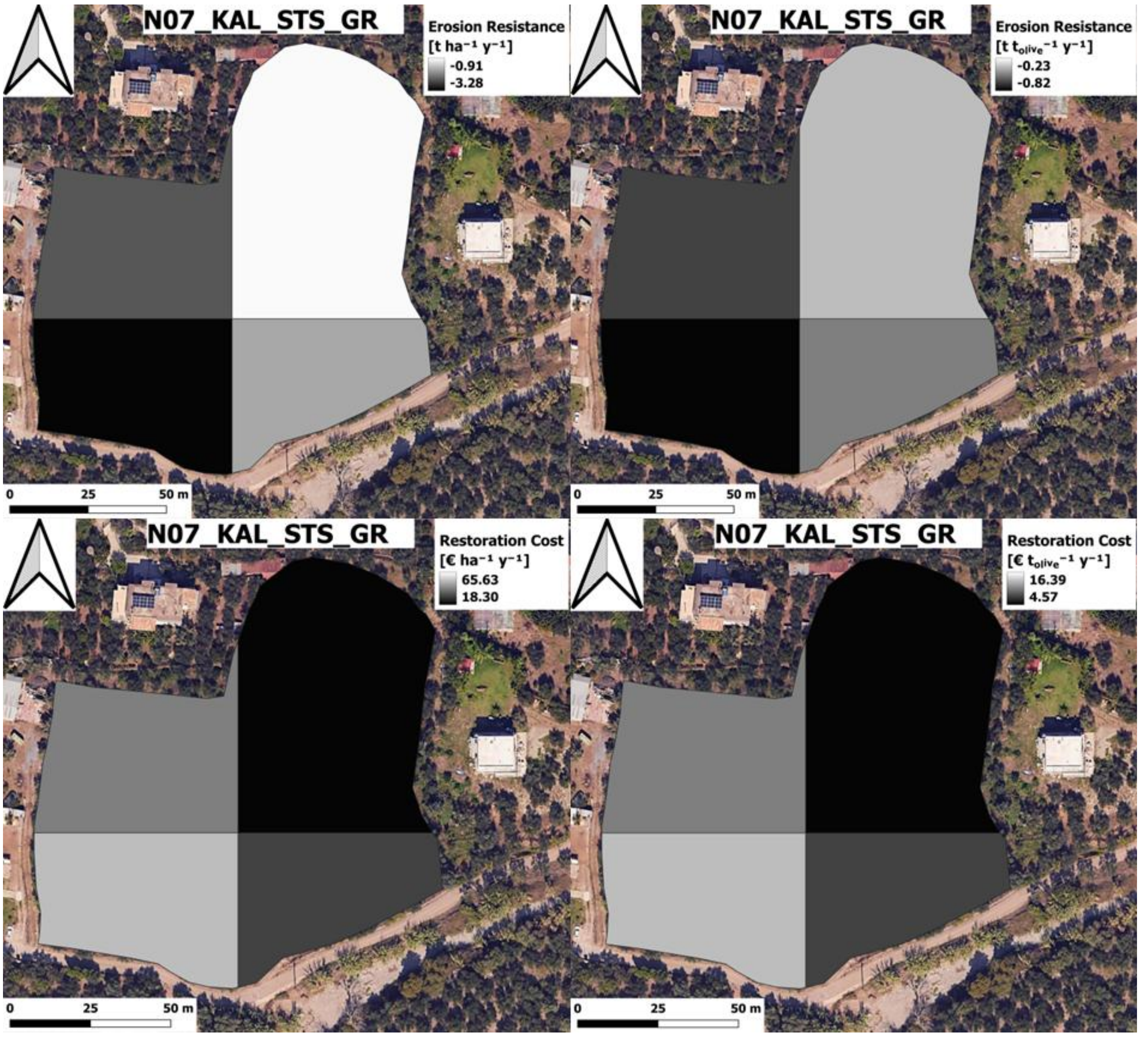

N02_CHR_STS_GR
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.08
-8.18
0 25 50 m
N02_CHR_STS_GR
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.03
-3.00
0 25 50 m
N02_CHR_STS_GR
Restoration Cost
[€ ha⁻¹ y⁻¹]
163.85
1.54
0 25 50 m
N02_CHR_STS_GR
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
60.16
0.57
0 25 50 m

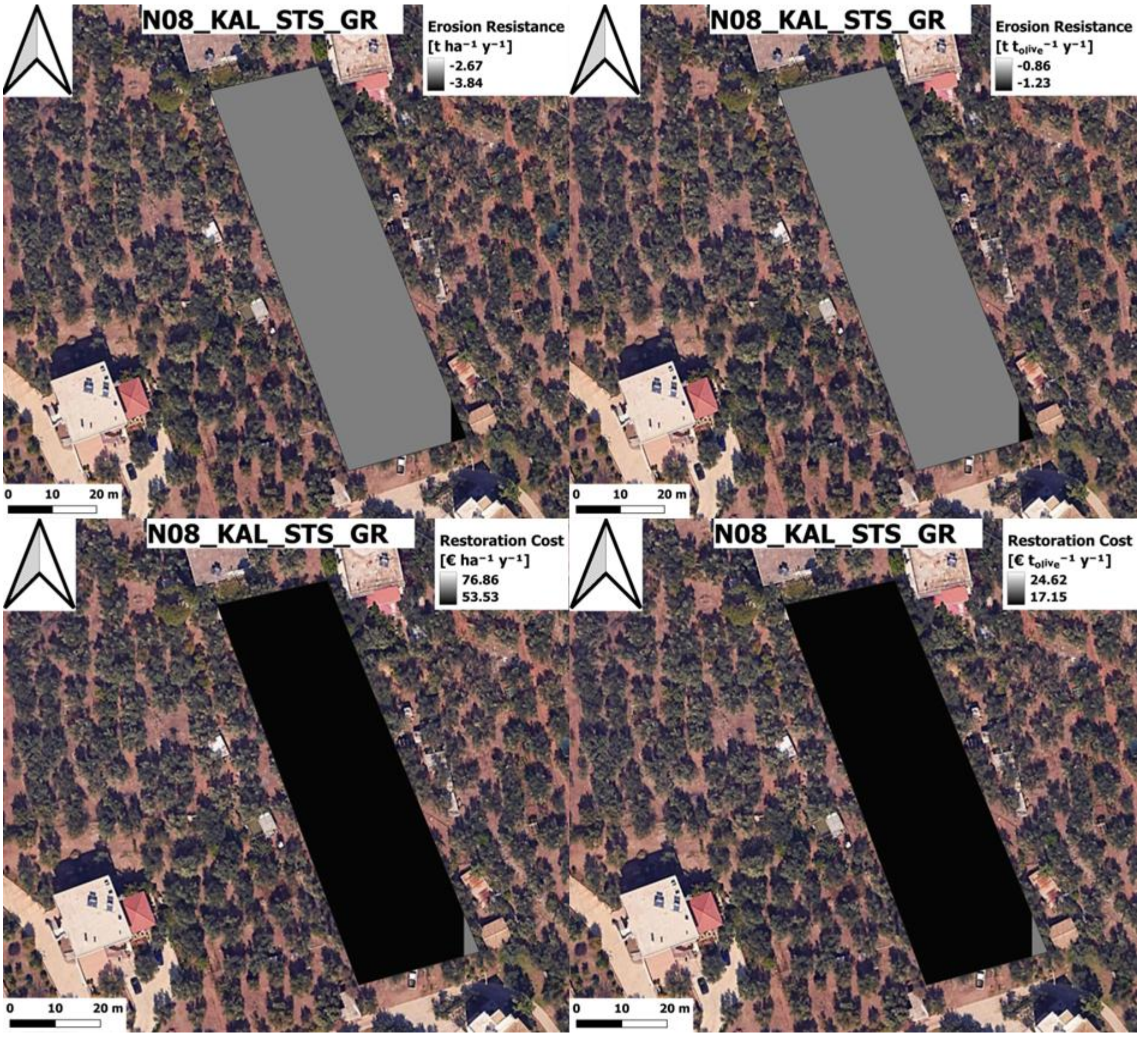

N03_CHR_STS_GR
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.24
-27.52
0 25 50 m
N03_CHR_STS_GR
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.13
-15.15
0 25 50 m
N03_CHR_STS_GR
Restoration Cost
[€ ha⁻¹ y⁻¹]
551.26
4.81
0 25 50 m
N03_CHR_STS_GR
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
303.55
2.65
0 25 50 m


N04_CHR_nSTS_GR
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.19
-27.52
0 25 50 m
N04_CHR_nSTS_GR
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.11
-15.66
0 25 50 m
N04_CHR_nSTS_GR
Restoration Cost
[t ha⁻¹ y⁻¹]
551.26
3.86
0 25 50 m
N04_CHR_nSTS_GR
Restoration Cost
[t t_olive⁻¹ y⁻¹]
313.69
2.20
0 25 50 m

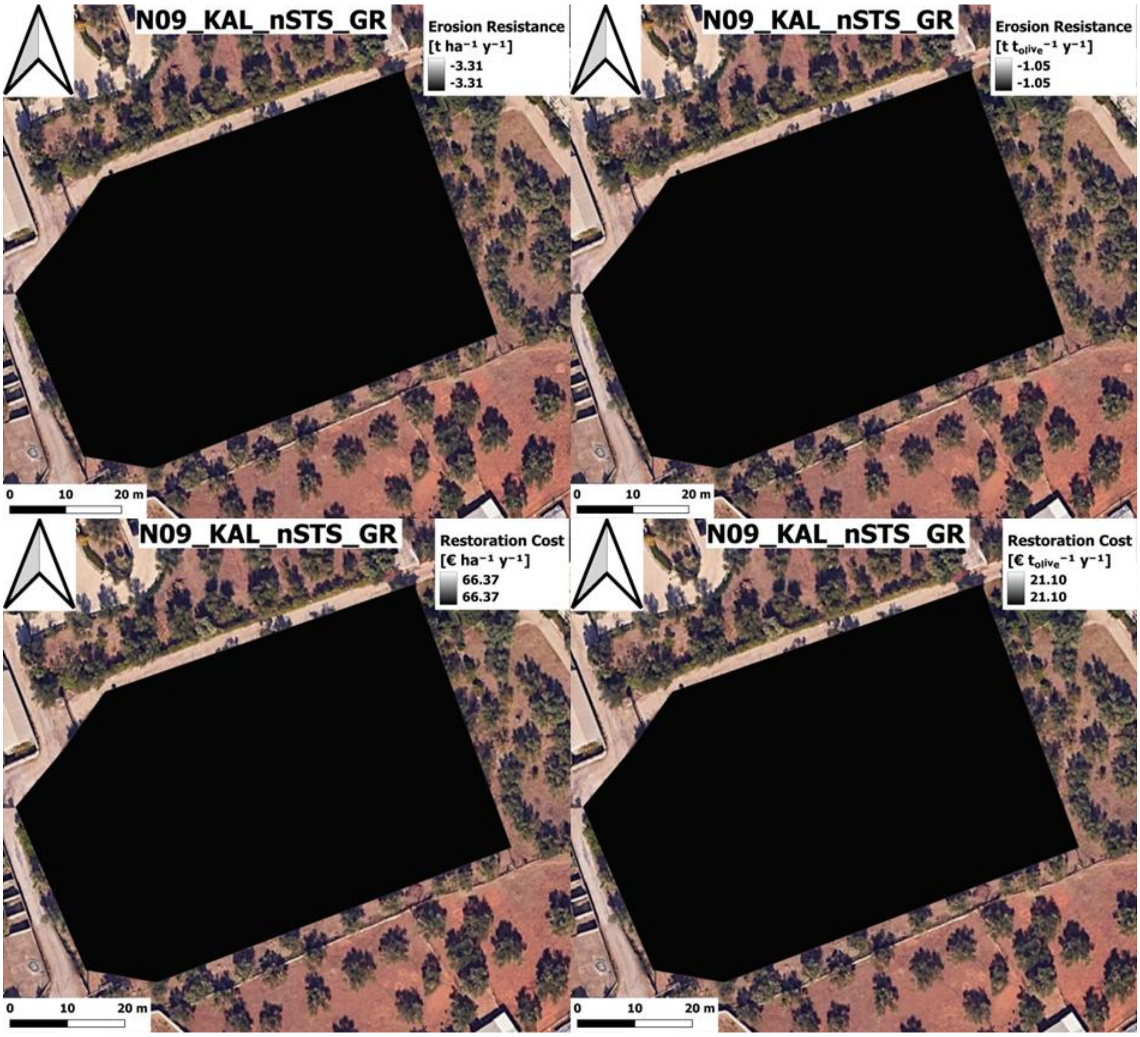

N05_CHR_ nSTS_GR
Erosion Resistance
[t ha⁻¹ y⁻¹]
-3.16
-5.38
0 25 50 m
N05_CHR_ nSTS_GR
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-1.72
-2.92
0 25 50 m
N05_CHR_ nSTS_GR
Restoration Cost
[€ ha⁻¹ y⁻¹]
107.75
63.24
0 25 50 m
N05_CHR_ nSTS_GR
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
58.59
34.39
0 25 50 m

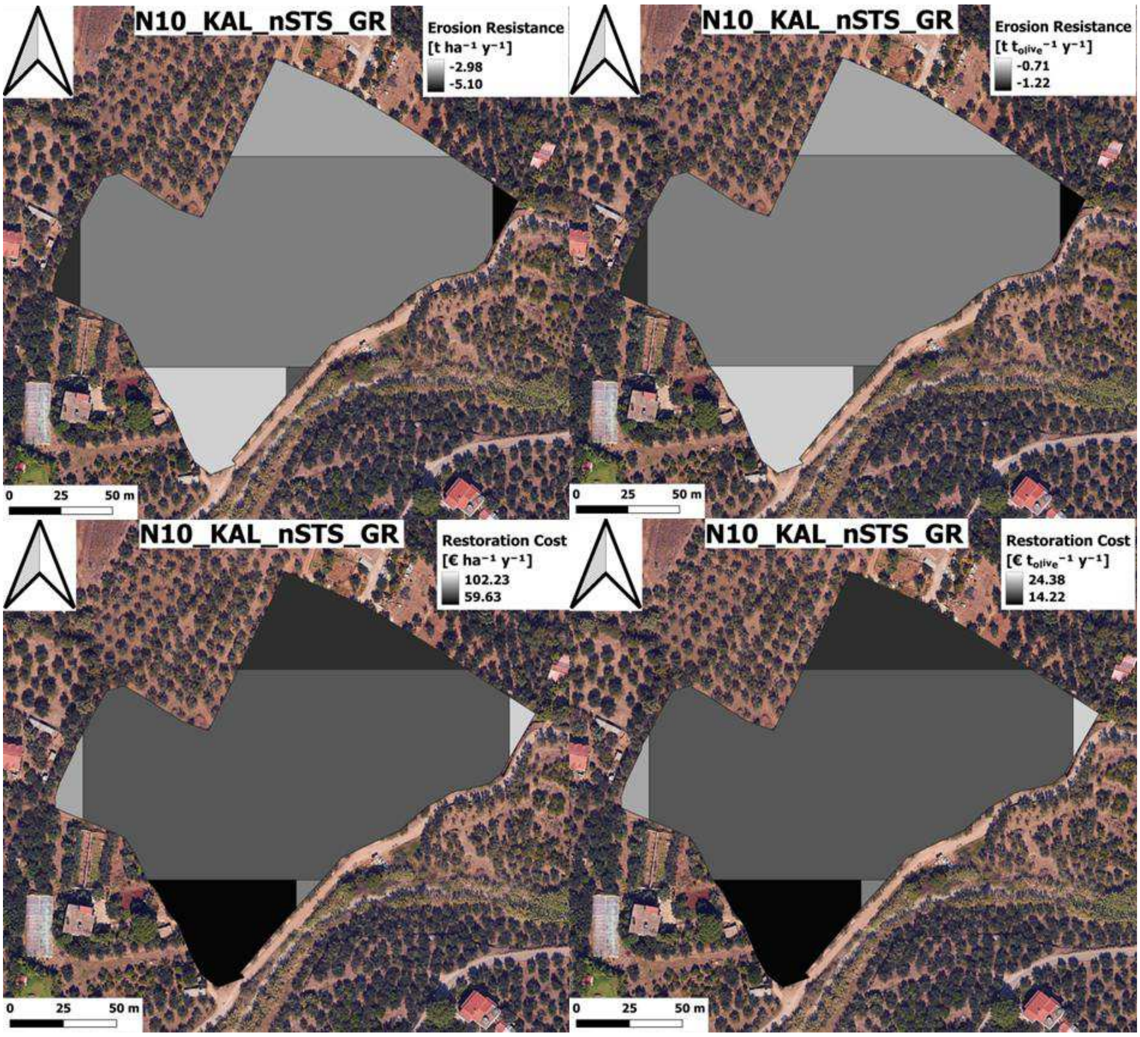

N06_CHR_nSTS_GR
Erosion Resistance
[t ha⁻¹ y⁻¹]
-3.13
-3.67
0 10 20 m
N06_CHR_nSTS_GR
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-1.24
-1.45
0 10 20 m
N06_CHR_nSTS_GR
Restoration Cost
[€ ha⁻¹ y⁻¹]
73.54
62.69
0 10 20 m
N06_CHR_nSTS_GR
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
29.11
24.82
0 10 20 m


N07_KAL_STS_GR
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.91
-3.28
0 25 50 m
N07_KAL_STS_GR
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.23
-0.82
0 25 50 m
N07_KAL_STS_GR
Restoration Cost
[€ ha⁻¹ y⁻¹]
65.63
18.30
0 25 50 m
N07_KAL_STS_GR
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
16.39
4.57
0 25 50 m



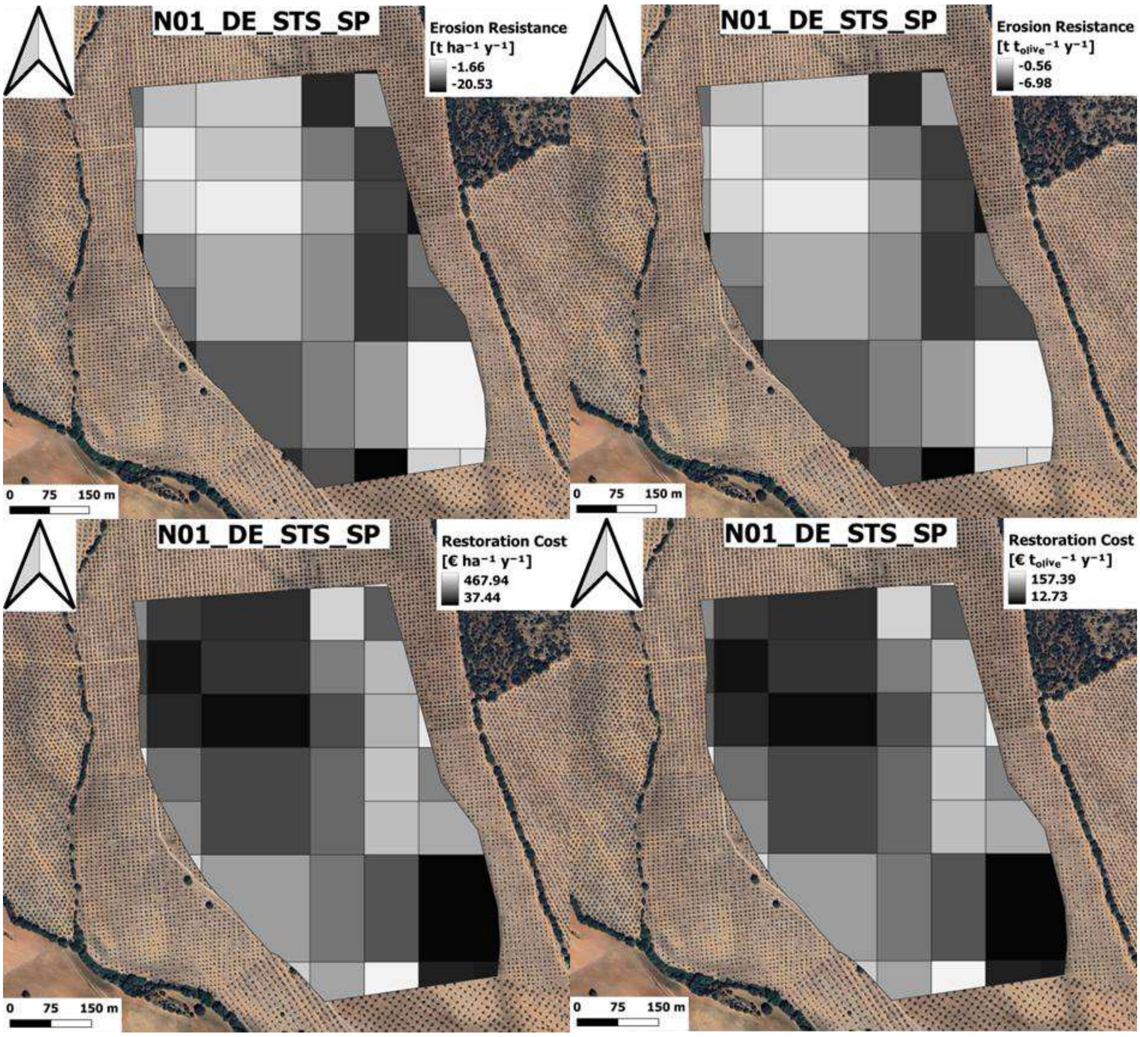
N08_KAL_STS_GR
Erosion Resistance
[t ha⁻¹ y⁻¹]
-2.67
-3.84
0 10 20 m
N08_KAL_STS_GR
Erosion Resistance
[t t$_{olive}$⁻¹ y⁻¹]
-0.86
-1.23
0 10 20 m
N08_KAL_STS_GR
Restoration Cost
[€ ha⁻¹ y⁻¹]
76.86
53.53
0 10 20 m
N08_KAL_STS_GR
Restoration Cost
[€ t$_{olive}$⁻¹ y⁻¹]
24.62
17.15
0 10 20 m

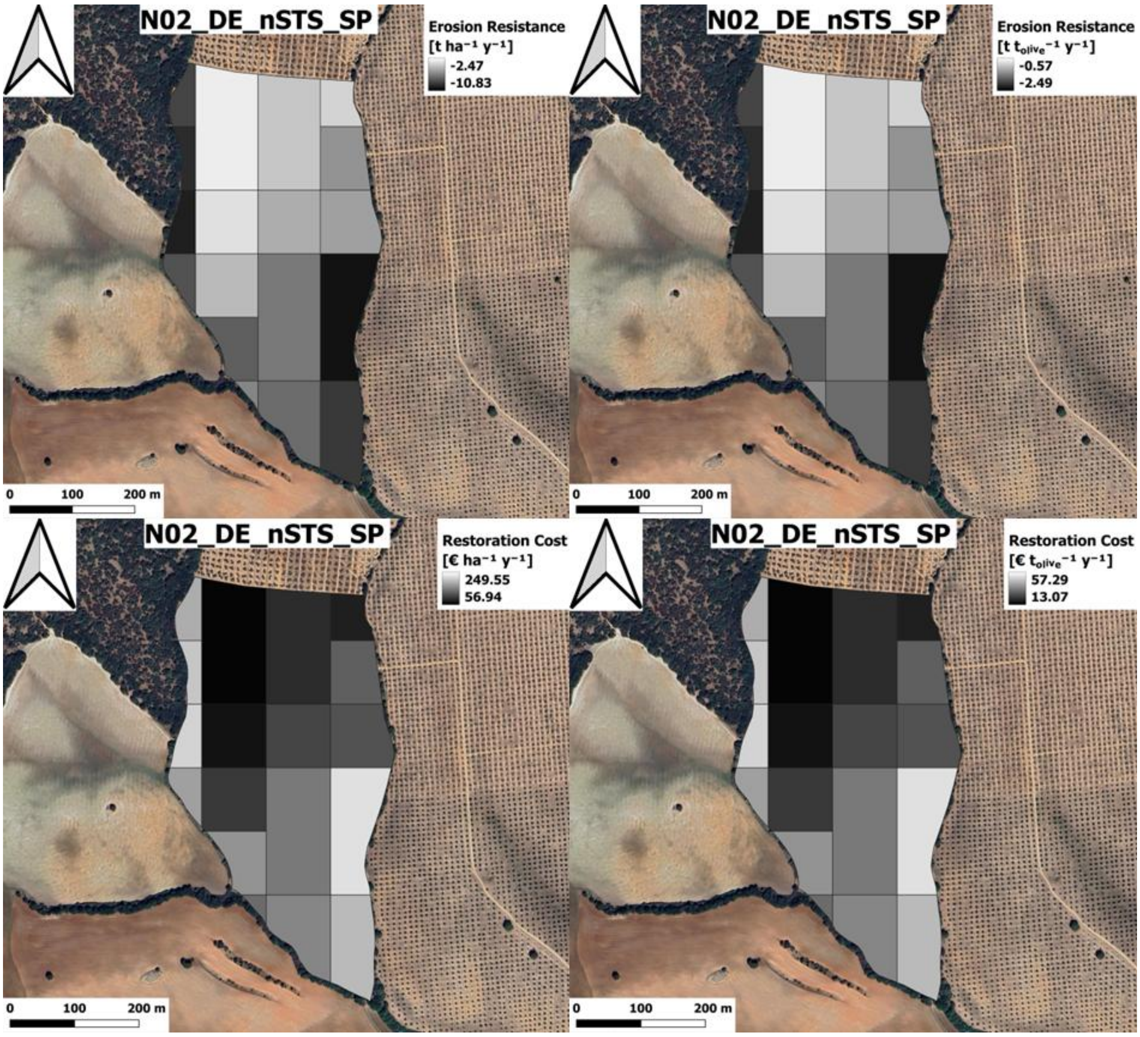

N09_KAL_nSTS_GR
Erosion Resistance
[t ha^{-1} y^{-1}]
-3.31
-3.31
0 10 20 m
N09_KAL_nSTS_GR
Erosion Resistance
[t t_{olive}^{-1} y^{-1}]
-1.05
-1.05
0 10 20 m
N09_KAL_nSTS_GR
Restoration Cost
[€ ha^{-1} y^{-1}]
66.37
66.37
0 10 20 m
N09_KAL_nSTS_GR
Restoration Cost
[€ t_{olive}^{-1} y^{-1}]
21.10
21.10
0 10 20 m

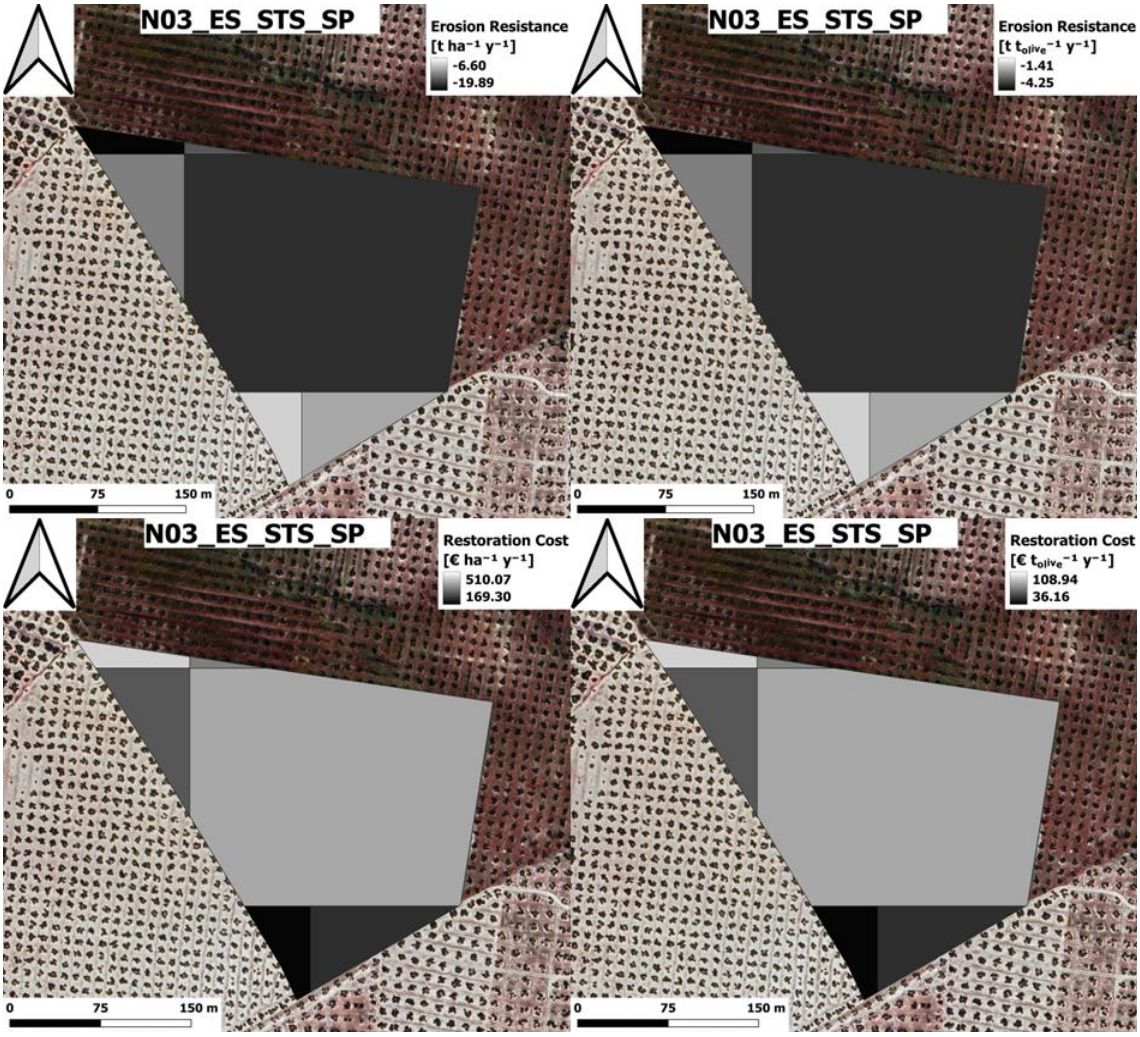

N10_KAL_nSTS_GR
Erosion Resistance
[t ha⁻¹ y⁻¹]
-2.98
-5.10
0 25 50 m
N10_KAL_nSTS_GR
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.71
-1.22
0 25 50 m
N10_KAL_nSTS_GR
Restoration Cost
[€ ha⁻¹ y⁻¹]
102.23
59.63
0 25 50 m
N10_KAL_nSTS_GR
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
24.38
14.22
0 25 50 m

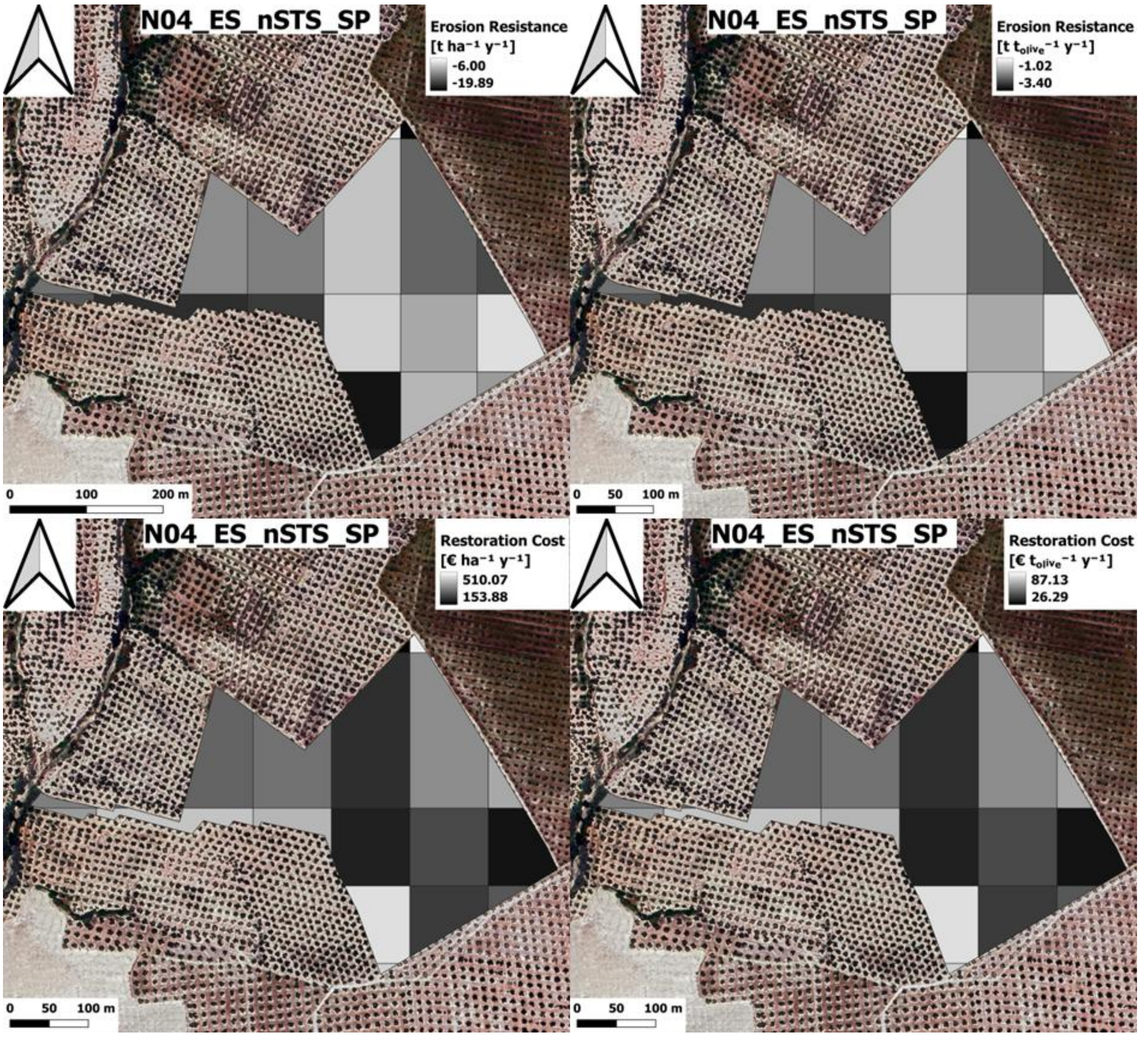
Spain

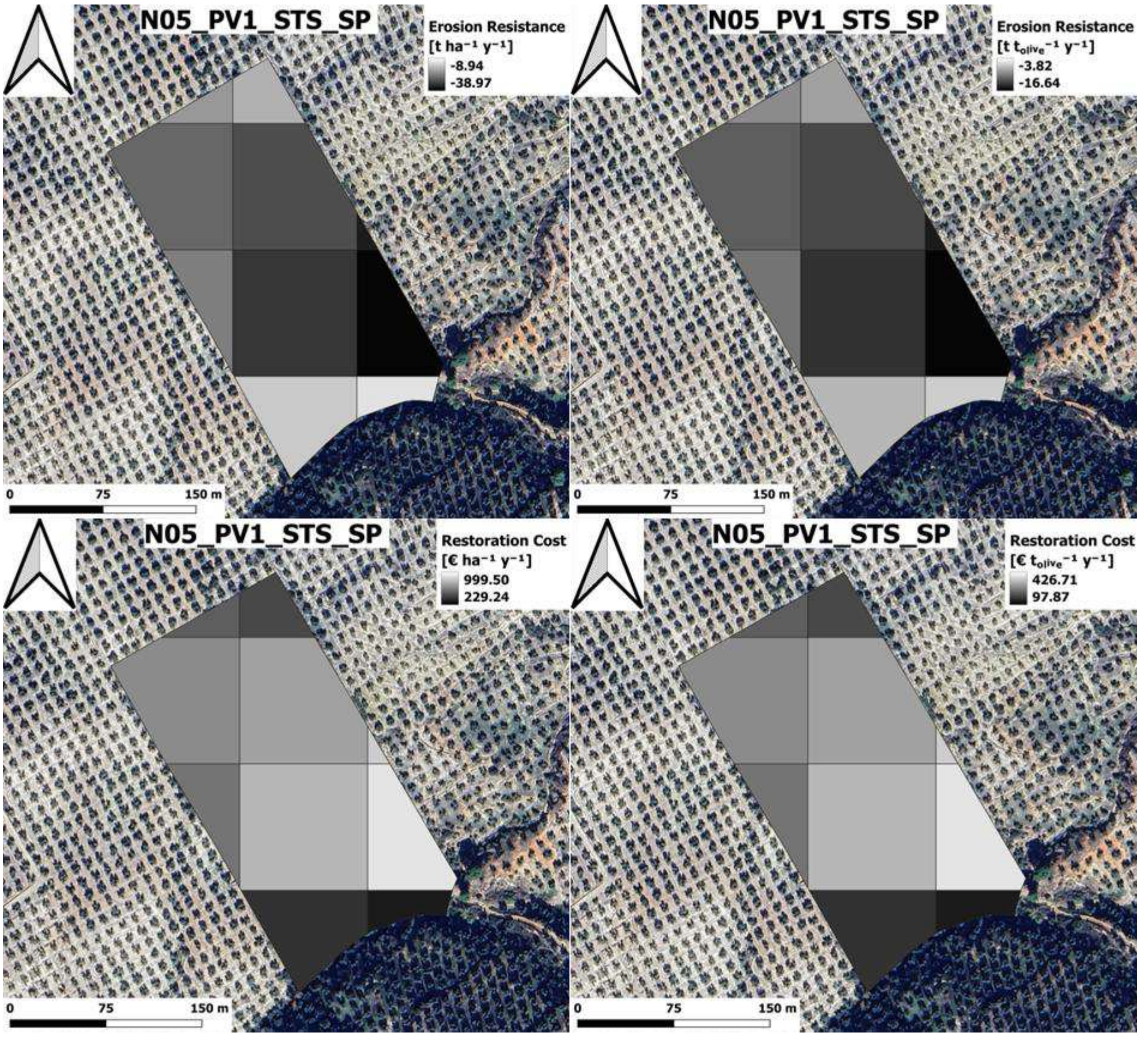
In this section are collected Erosion Resistance maps for Italy. The Coordinate Reference System (CSR) is WGS-84 UTM 30N for all olive farms.

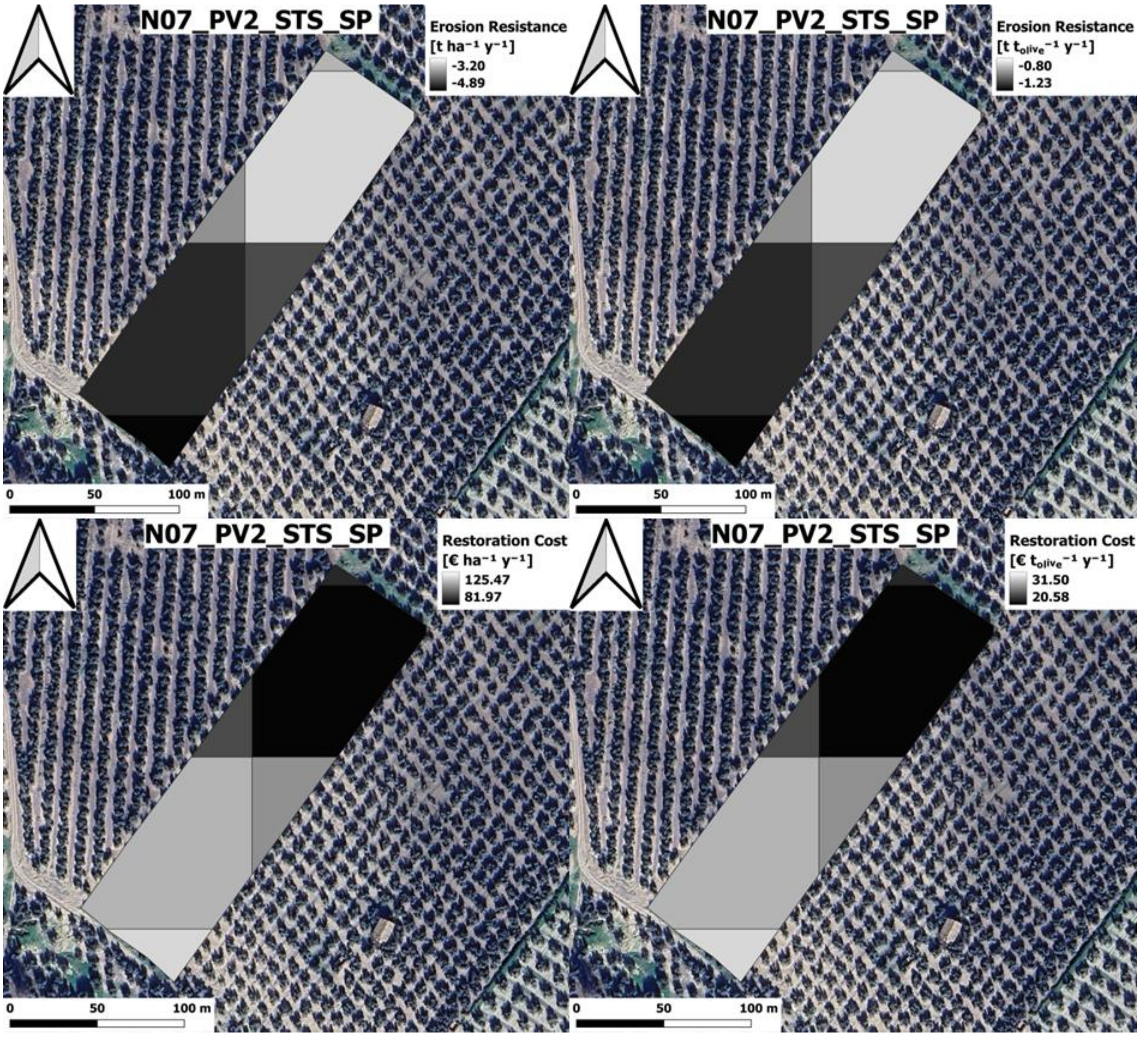


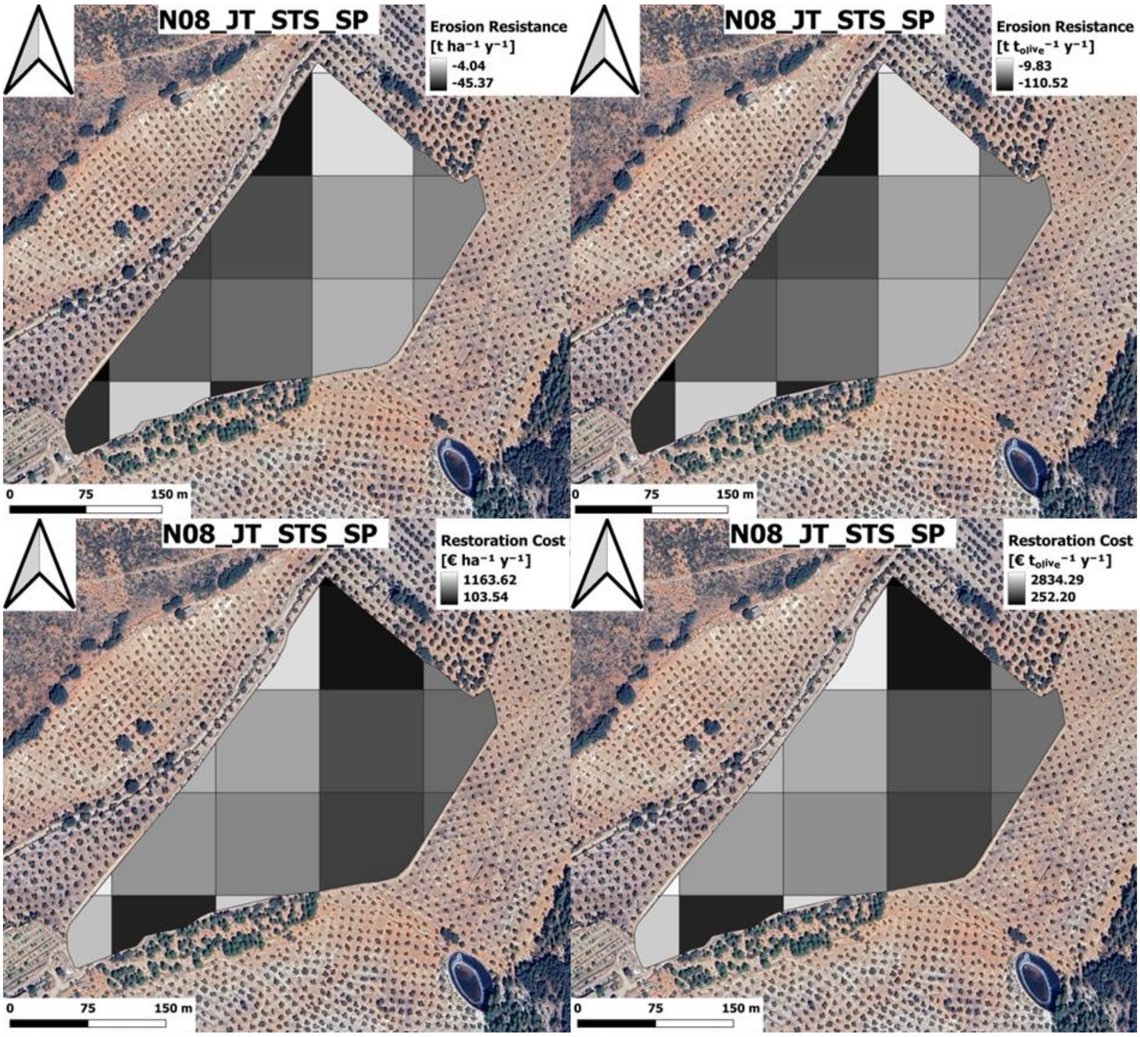

N02_DE_nSTS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-2.47
-10.83
0 100 200 m
N02_DE_nSTS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.57
-2.49
0 100 200 m
N02_DE_nSTS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
249.55
56.94
0 100 200 m
N02_DE_nSTS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
57.29
13.07
0 100 200 m

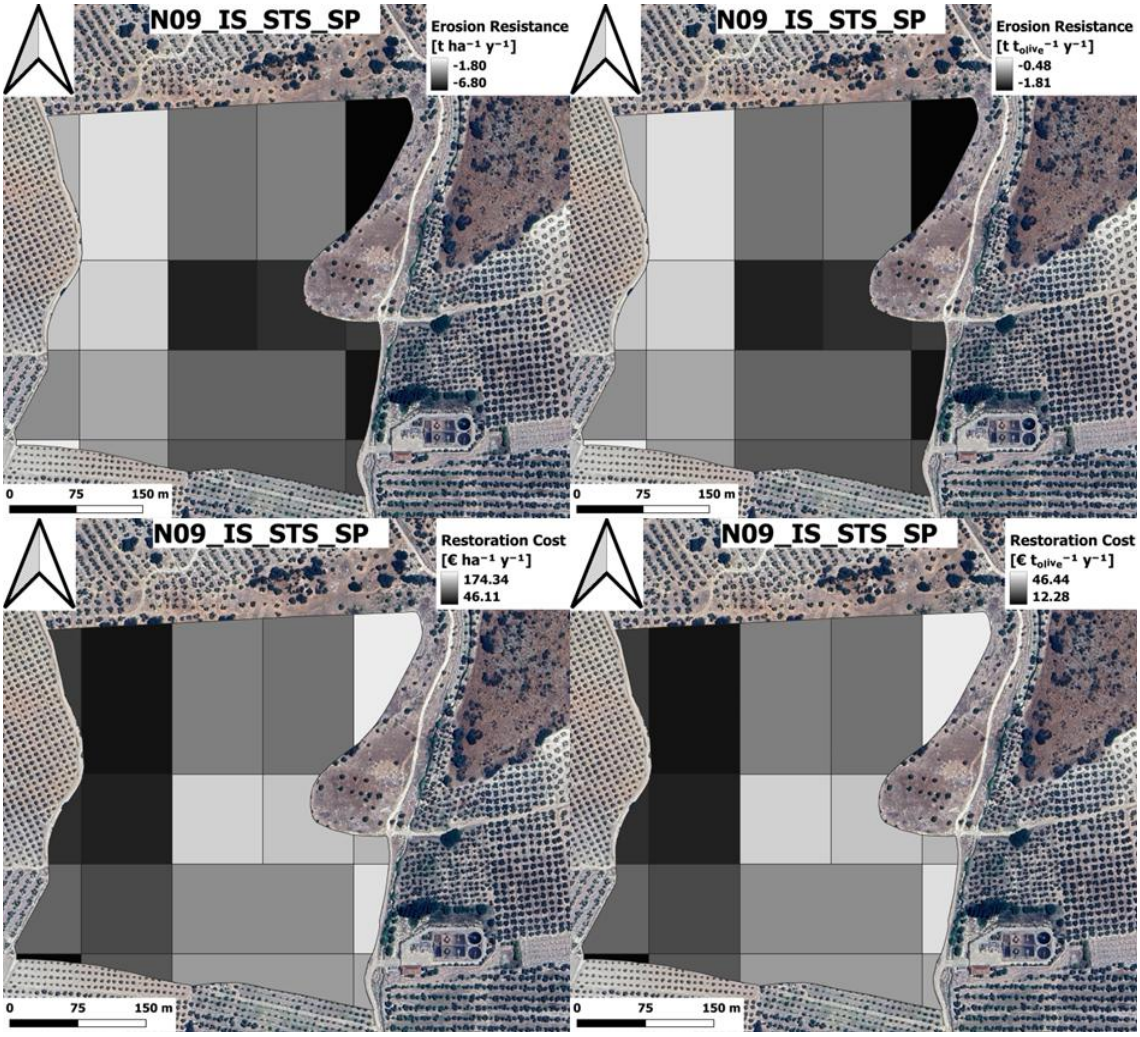

N03_ES_STS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-6.60
-19.89
0 75 150 m
N03_ES_STS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-1.41
-4.25
0 75 150 m
N03_ES_STS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
510.07
169.30
0 75 150 m
N03_ES_STS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
108.94
36.16
0 75 150 m

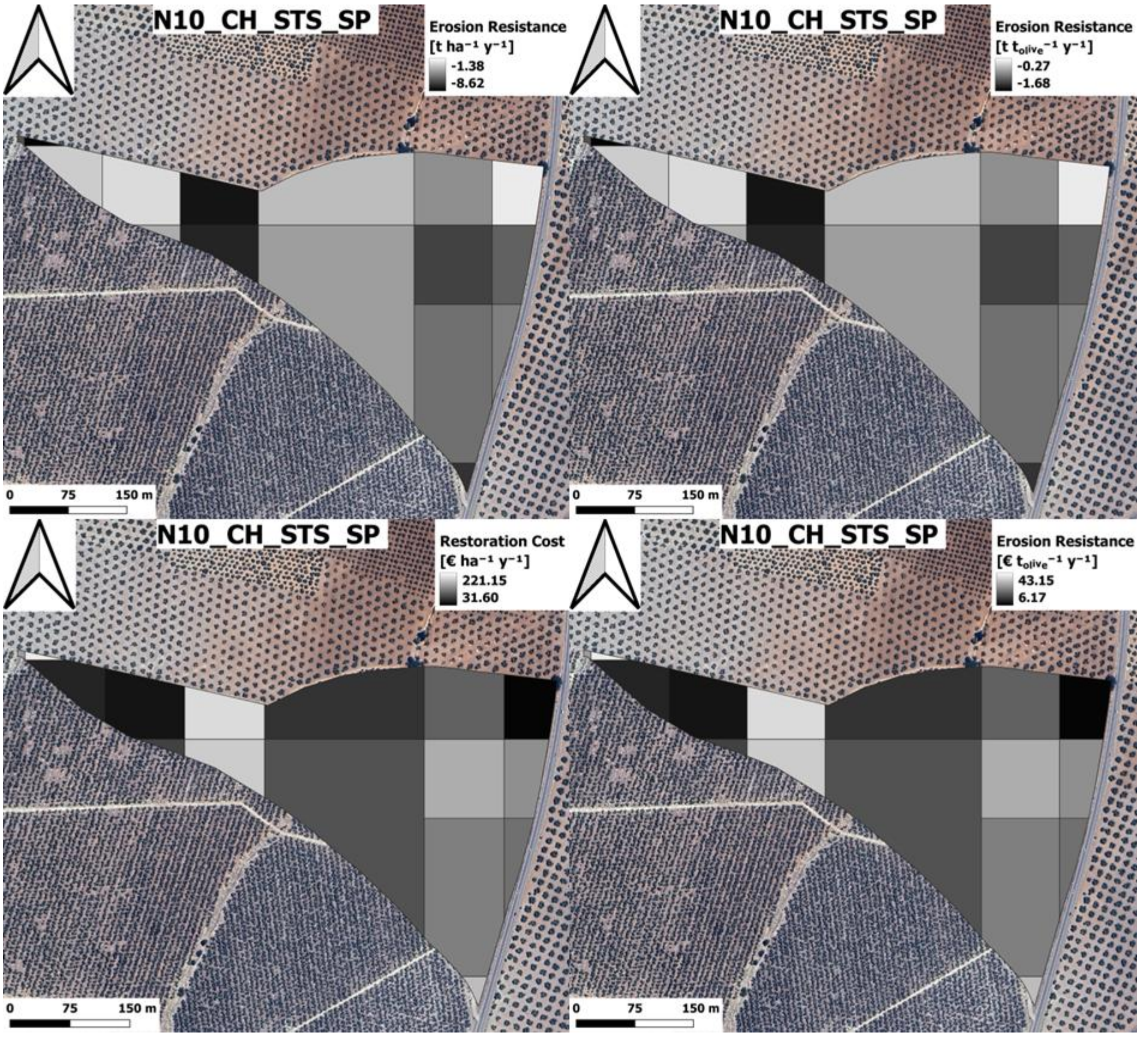

N04_ES_nSTS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-6.00
-19.89
0 100 200 m
N04_ES_nSTS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-1.02
-3.40
0 50 100 m
N04_ES_nSTS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
510.07
153.88
0 50 100 m
N04_ES_nSTS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
87.13
26.29
0 50 100 m

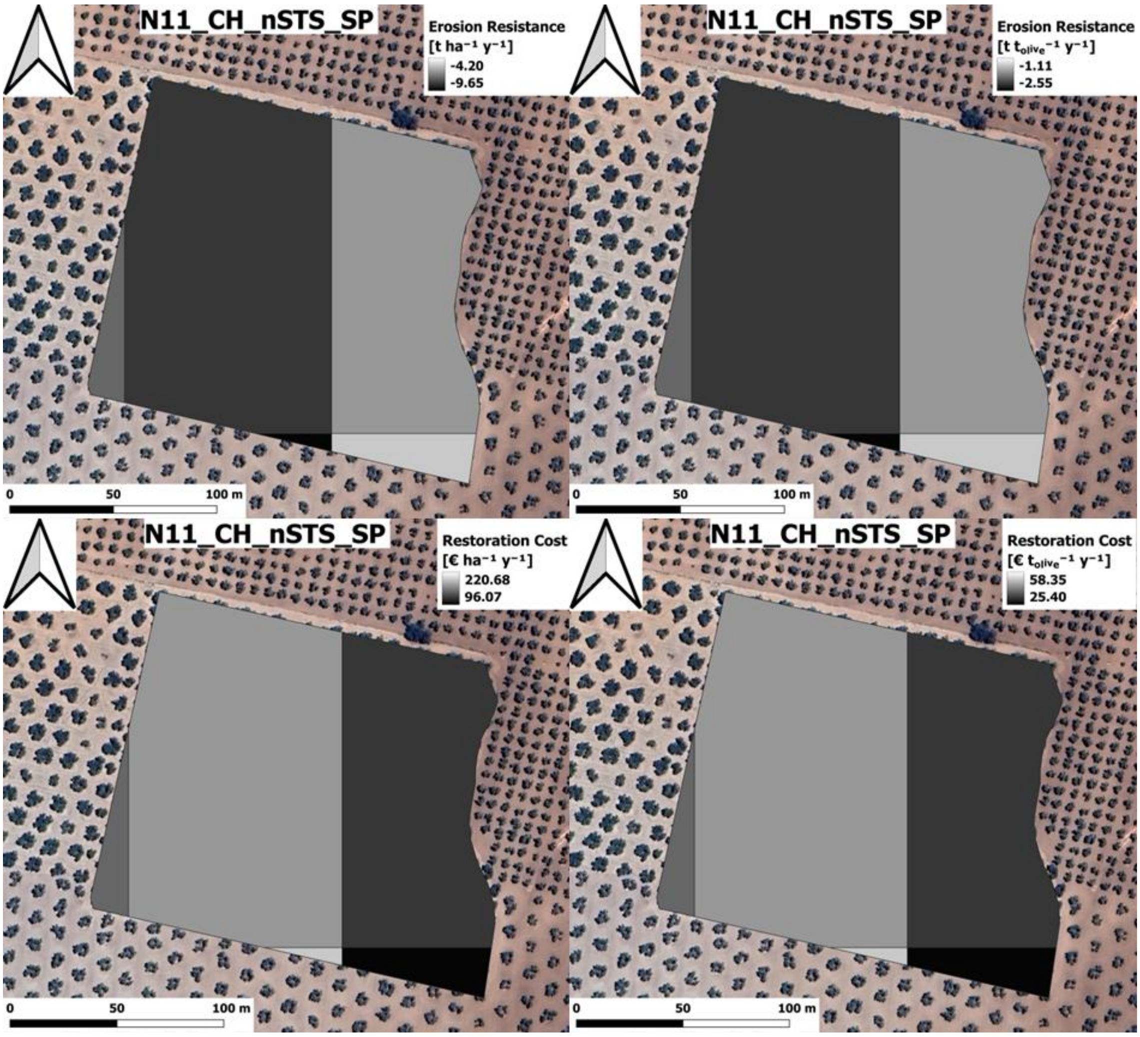

N05_PV1_STS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-8.94
-38.97
0 75 150 m
N05_PV1_STS_SP
Erosion Resistance
[t t$_{olive}$⁻¹ y⁻¹]
-3.82
-16.64
0 75 150 m
N05_PV1_STS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
999.50
229.24
0 75 150 m
N05_PV1_STS_SP
Restoration Cost
[€ t$_{olive}$⁻¹ y⁻¹]
426.71
97.87
0 75 150 m

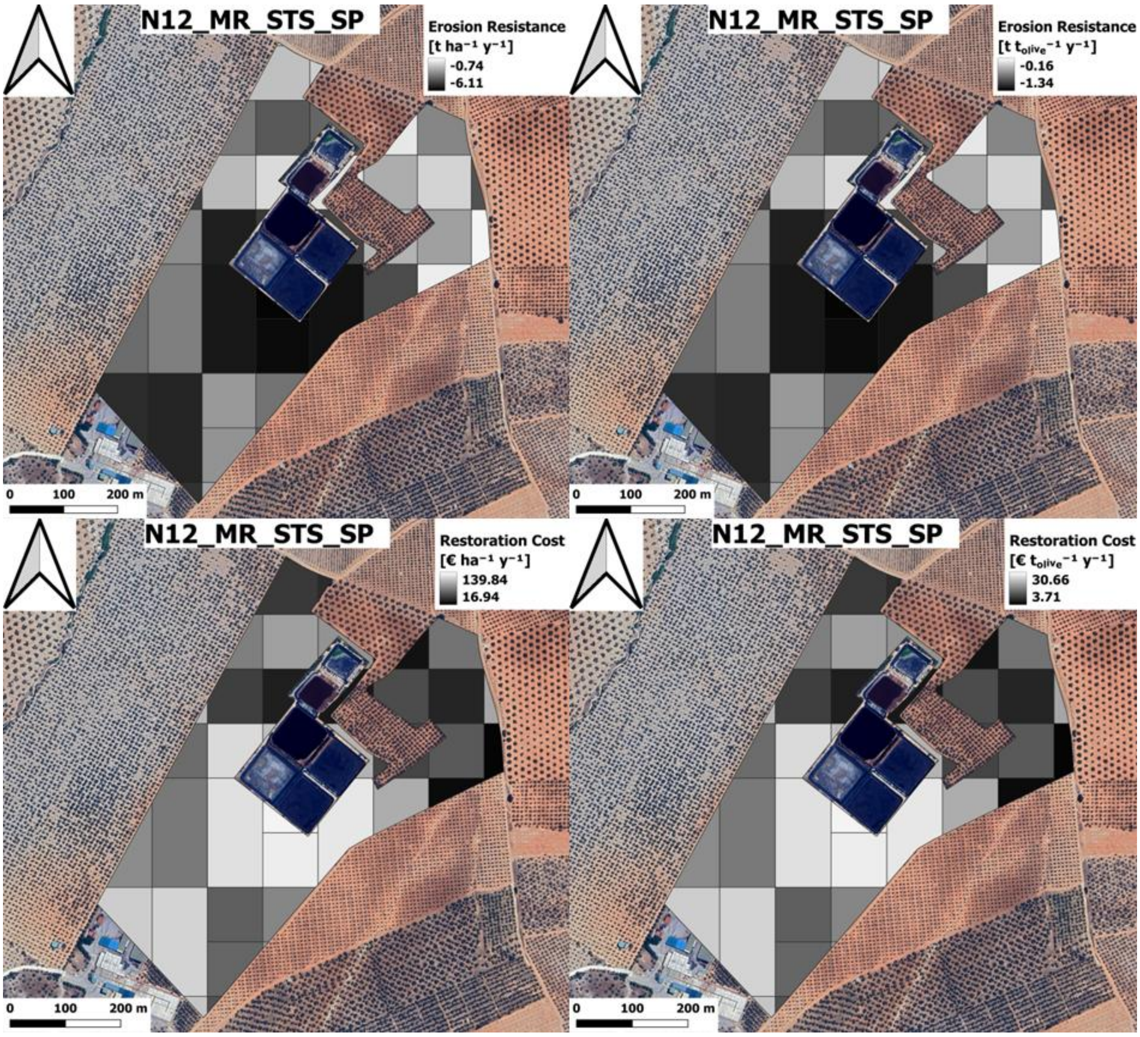

N07_PV2_STS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-3.20
-4.89
0 50 100 m
N07_PV2_STS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.80
-1.23
0 50 100 m
N07_PV2_STS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
125.47
81.97
0 50 100 m
N07_PV2_STS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
31.50
20.58
0 50 100 m

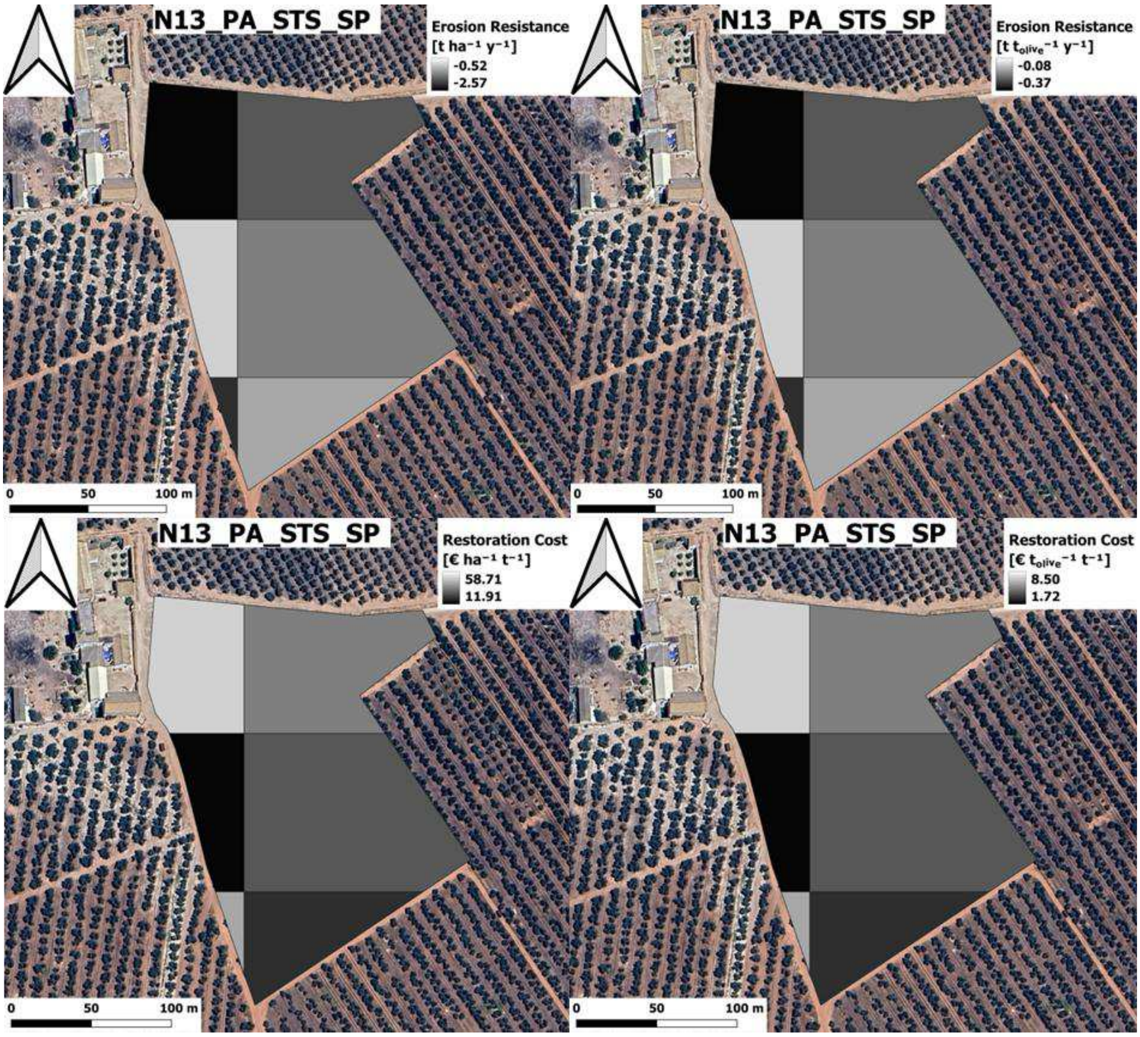

N08_JT_STS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-4.04
-45.37
0 75 150 m
N08_JT_STS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-9.83
-110.52
0 75 150 m
N08_JT_STS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
1163.62
103.54
0 75 150 m
N08_JT_STS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
2834.29
252.20
0 75 150 m

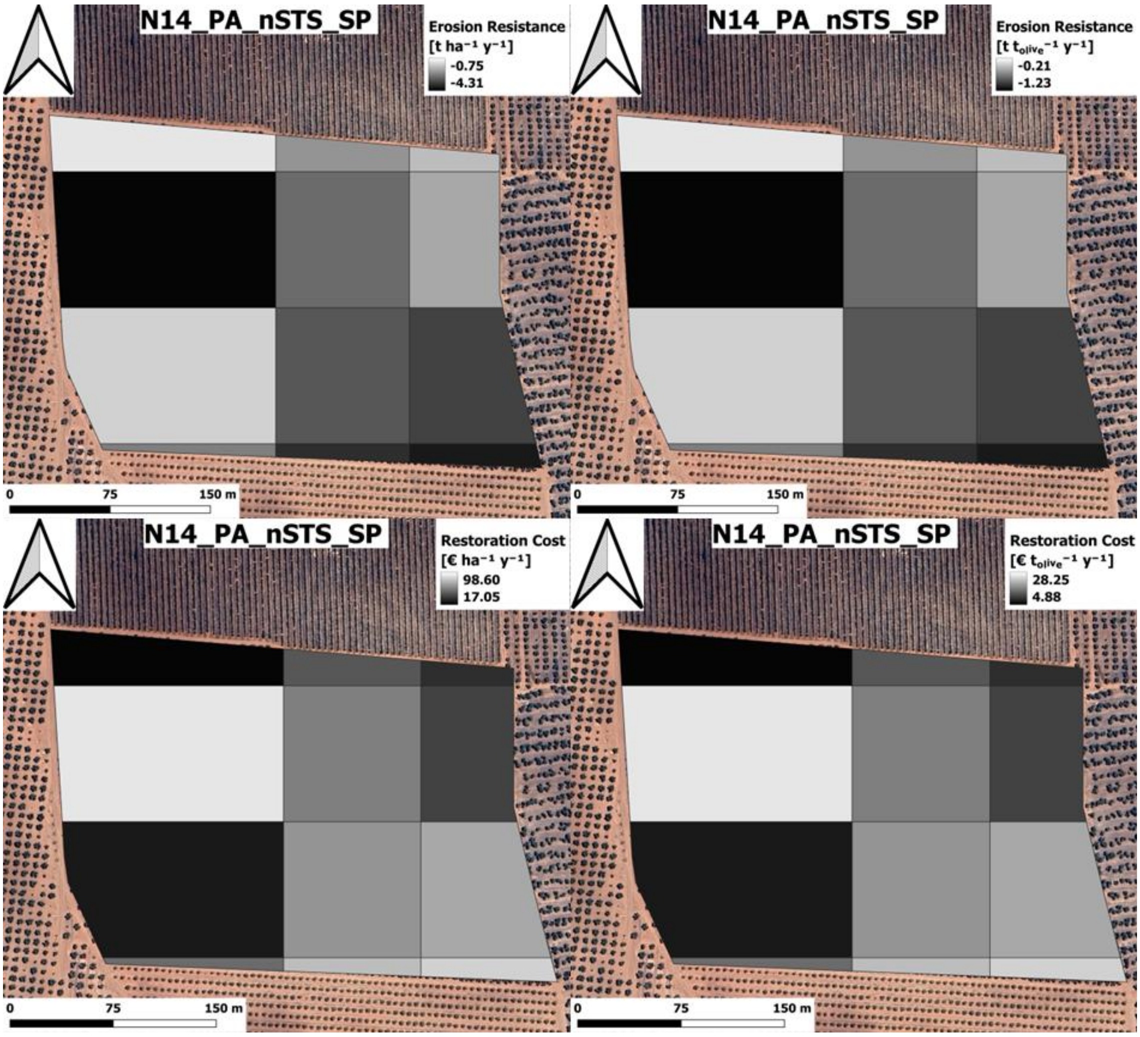

N09_IS_STS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-1.80
-6.80
0 75 150 m
N09_IS_STS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.48
-1.81
0 75 150 m
N09_IS_STS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
174.34
46.11
0 75 150 m
N09_IS_STS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
46.44
12.28
0 75 150 m

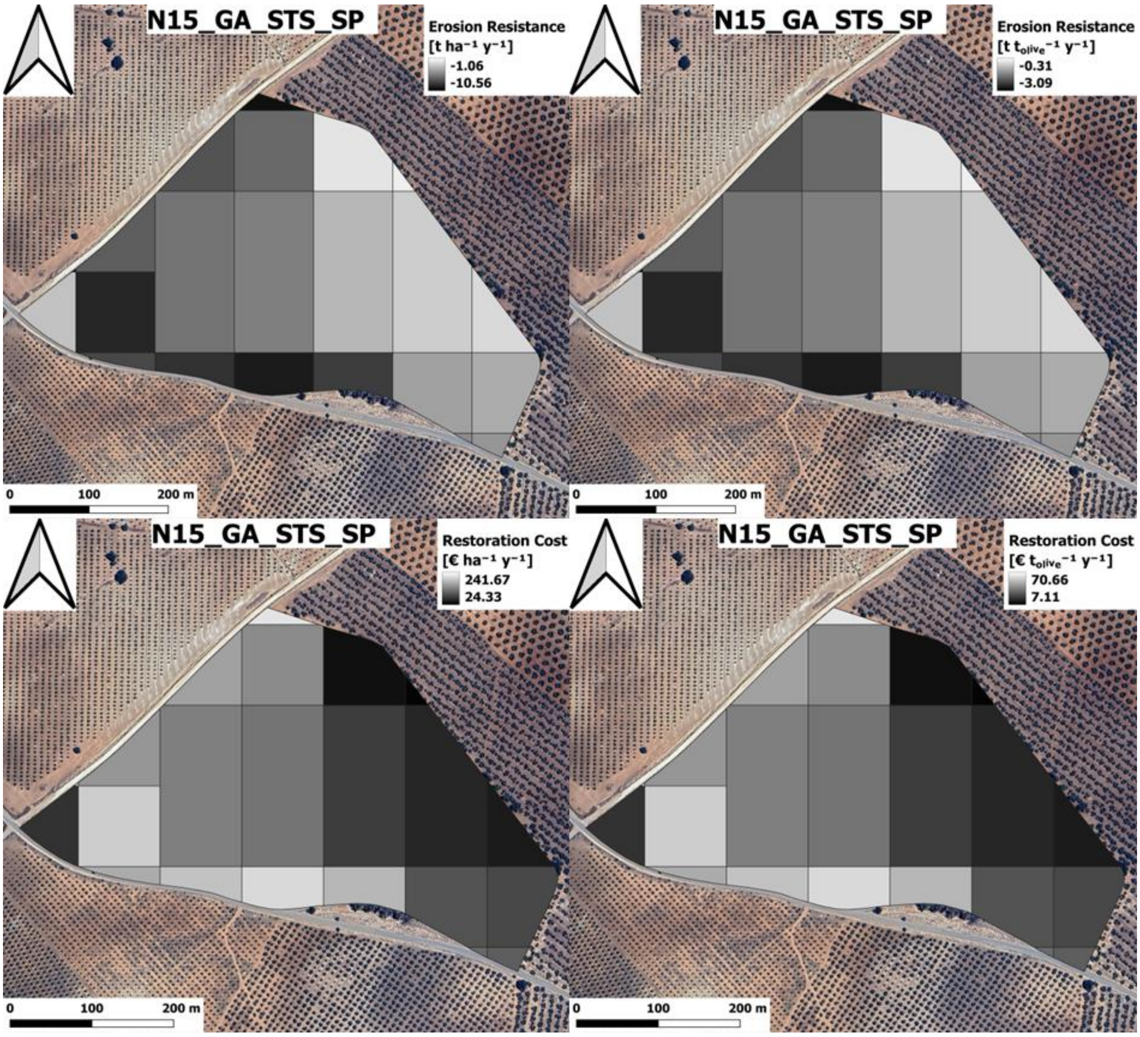

N10_CH_STS_SP
Erosion Resistance
[t ha^{-1} y^{-1}]
-1.38
-8.62
0 75 150 m
N10_CH_STS_SP
Erosion Resistance
[t t_{olive}^{-1} y^{-1}]
-0.27
-1.68
0 75 150 m
N10_CH_STS_SP
Restoration Cost
[€ ha^{-1} y^{-1}]
221.15
31.60
0 75 150 m
N10_CH_STS_SP
Erosion Resistance
[€ t_{olive}^{-1} y^{-1}]
43.15
6.17
0 75 150 m

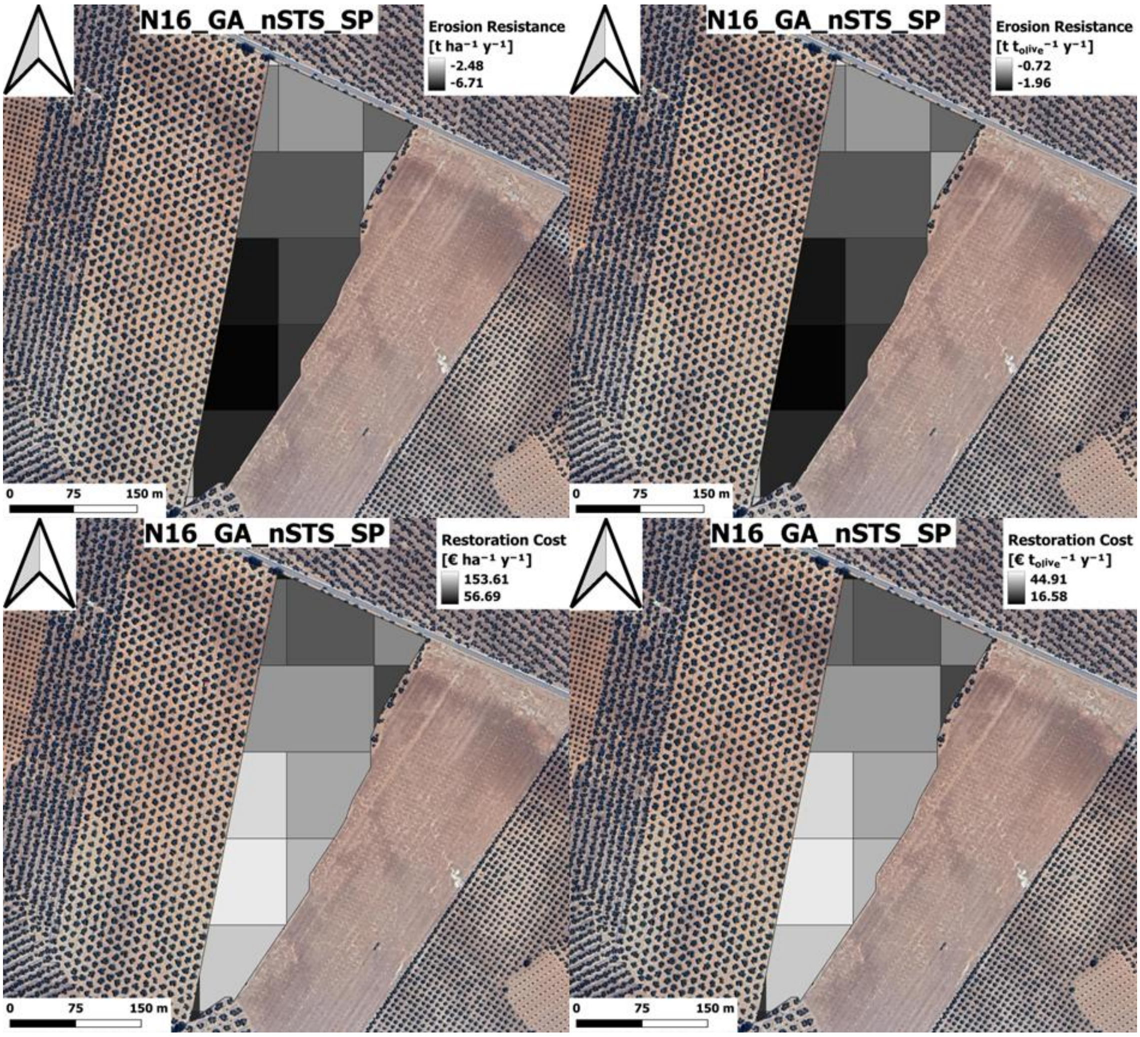

N11_CH_nSTS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-4.20
-9.65
0
50
100 m
N11_CH_nSTS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-1.11
-2.55
0
50
100 m
N11_CH_nSTS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
220.68
96.07
0
50
100 m
N11_CH_nSTS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
58.35
25.40
0
50
100 m

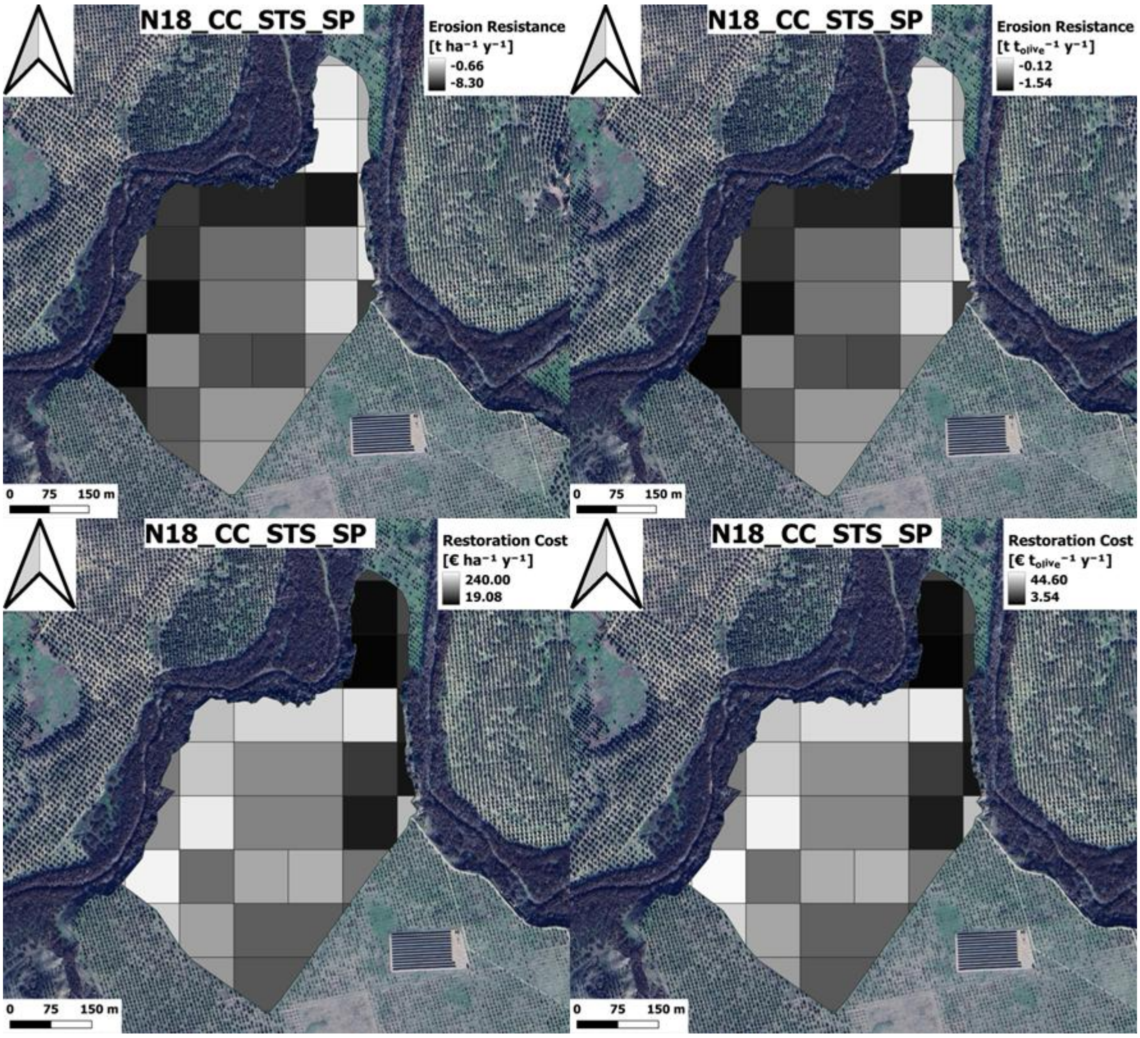

N12_MR_STS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.74
-6.11
0 100 200 m
N12_MR_STS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.16
-1.34
0 100 200 m
N12_MR_STS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
139.84
16.94
0 100 200 m
N12_MR_STS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
30.66
3.71
0 100 200 m


N13_PA_STS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.52
-2.57
0
50
100 m
N13_PA_STS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.08
-0.37
0
50
100 m
N13_PA_STS_SP
Restoration Cost
[€ ha⁻¹ t⁻¹]
58.71
11.91
0
50
100 m
N13_PA_STS_SP
Restoration Cost
[€ t_olive⁻¹ t⁻¹]
8.50
1.72
0
50
100 m


N14_PA_nSTS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.75
-4.31
0
75
150 m
N14_PA_nSTS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.21
-1.23
0
75
150 m
N14_PA_nSTS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
98.60
17.05
0
75
150 m
N14_PA_nSTS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
28.25
4.88
0
75
150 m

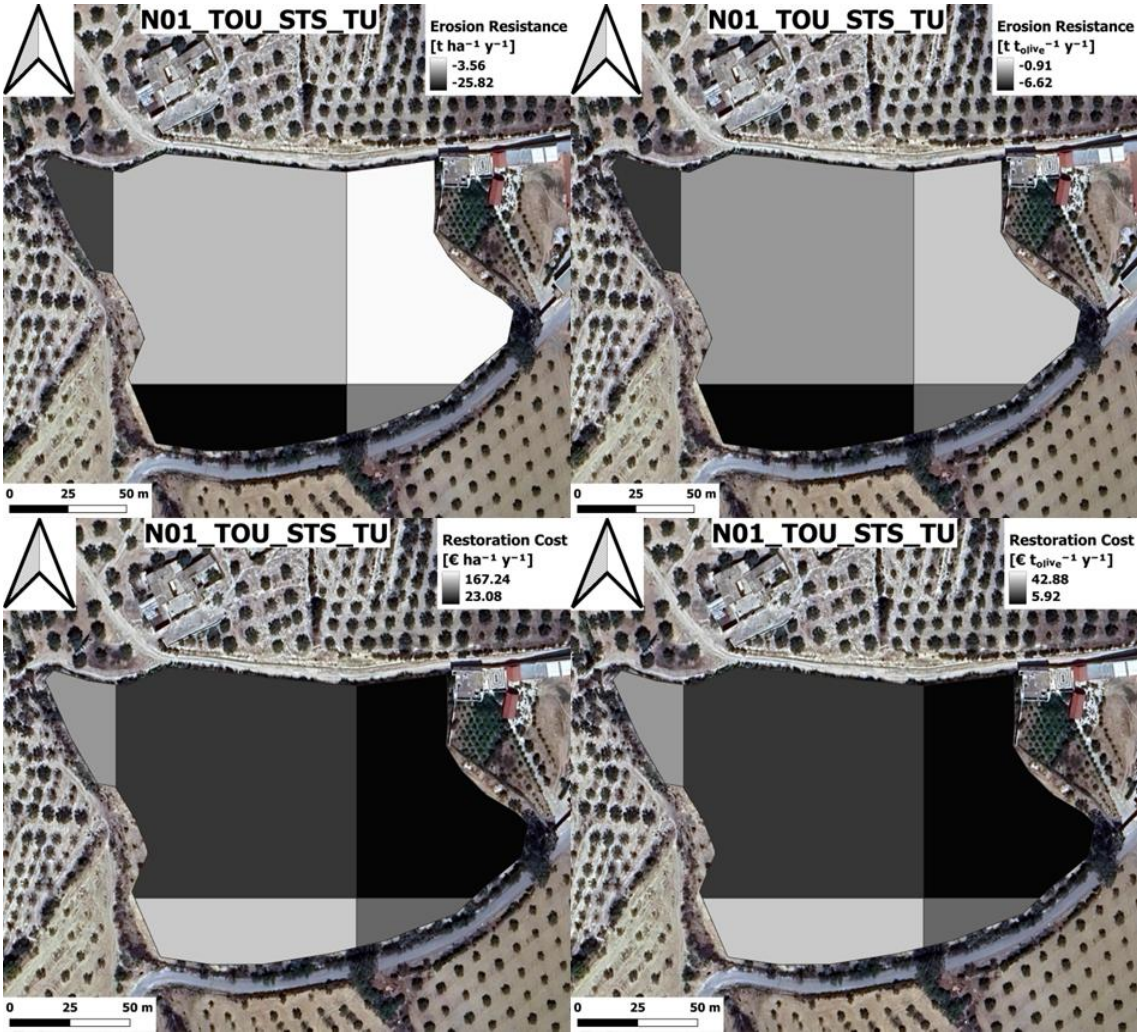

N15_GA_STS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-1.06
-10.56
0 100 200 m
N15_GA_STS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.31
-3.09
0 100 200 m
N15_GA_STS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
241.67
24.33
0 100 200 m
N15_GA_STS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
70.66
7.11
0 100 200 m

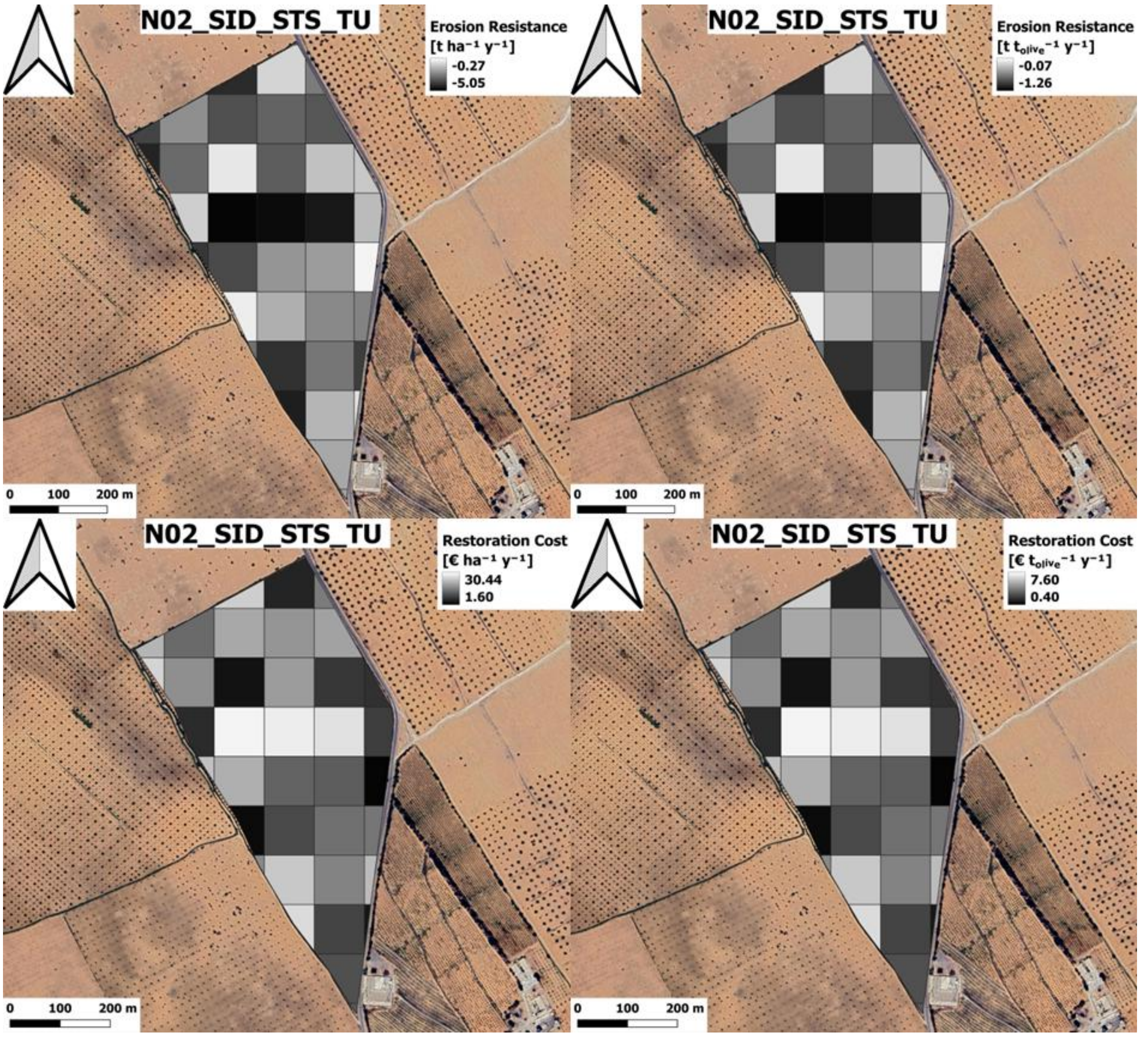

N16_GA_nSTS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-2.48
-6.71
0 75 150 m
N16_GA_nSTS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.72
-1.96
0 75 150 m
N16_GA_nSTS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
153.61
56.69
0 75 150 m
N16_GA_nSTS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
44.91
16.58
0 75 150 m

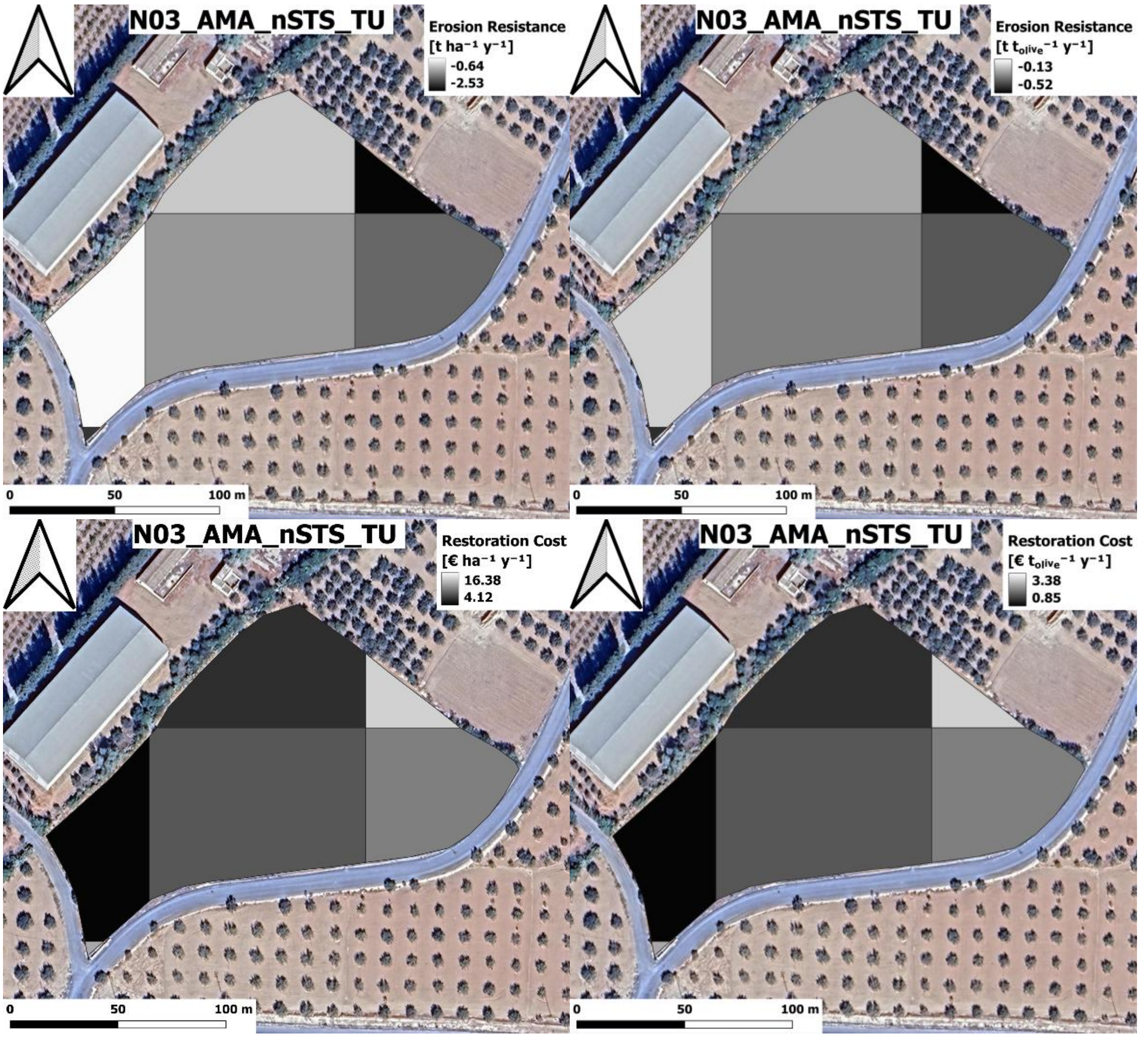

N18_CC_STS_SP
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.66
-8.30
0 75 150 m
N18_CC_STS_SP
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.12
-1.54
0 75 150 m
N18_CC_STS_SP
Restoration Cost
[€ ha⁻¹ y⁻¹]
240.00
19.08
0 75 150 m
N18_CC_STS_SP
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
44.60
3.54
0 75 150 m

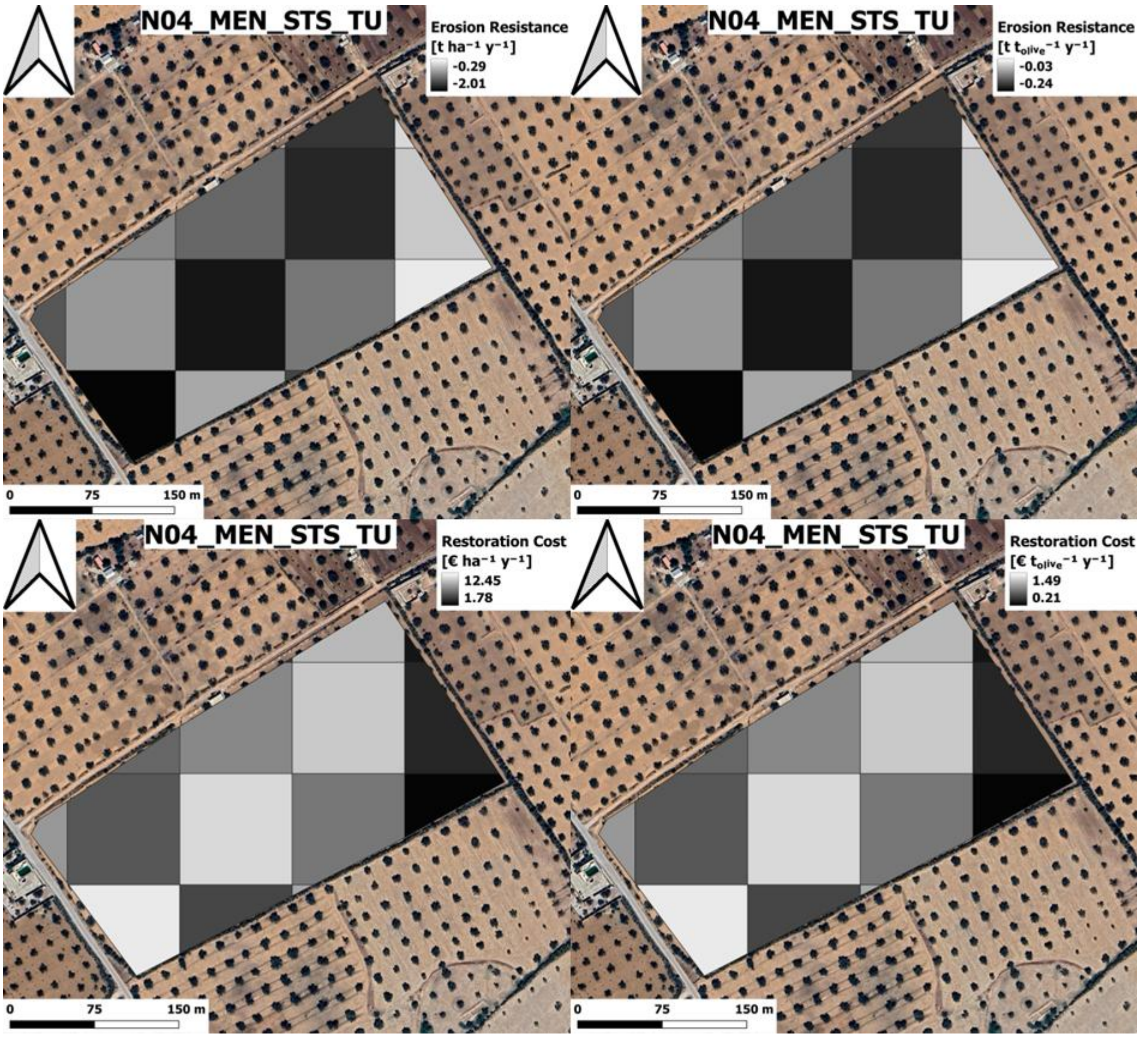
Tunisia

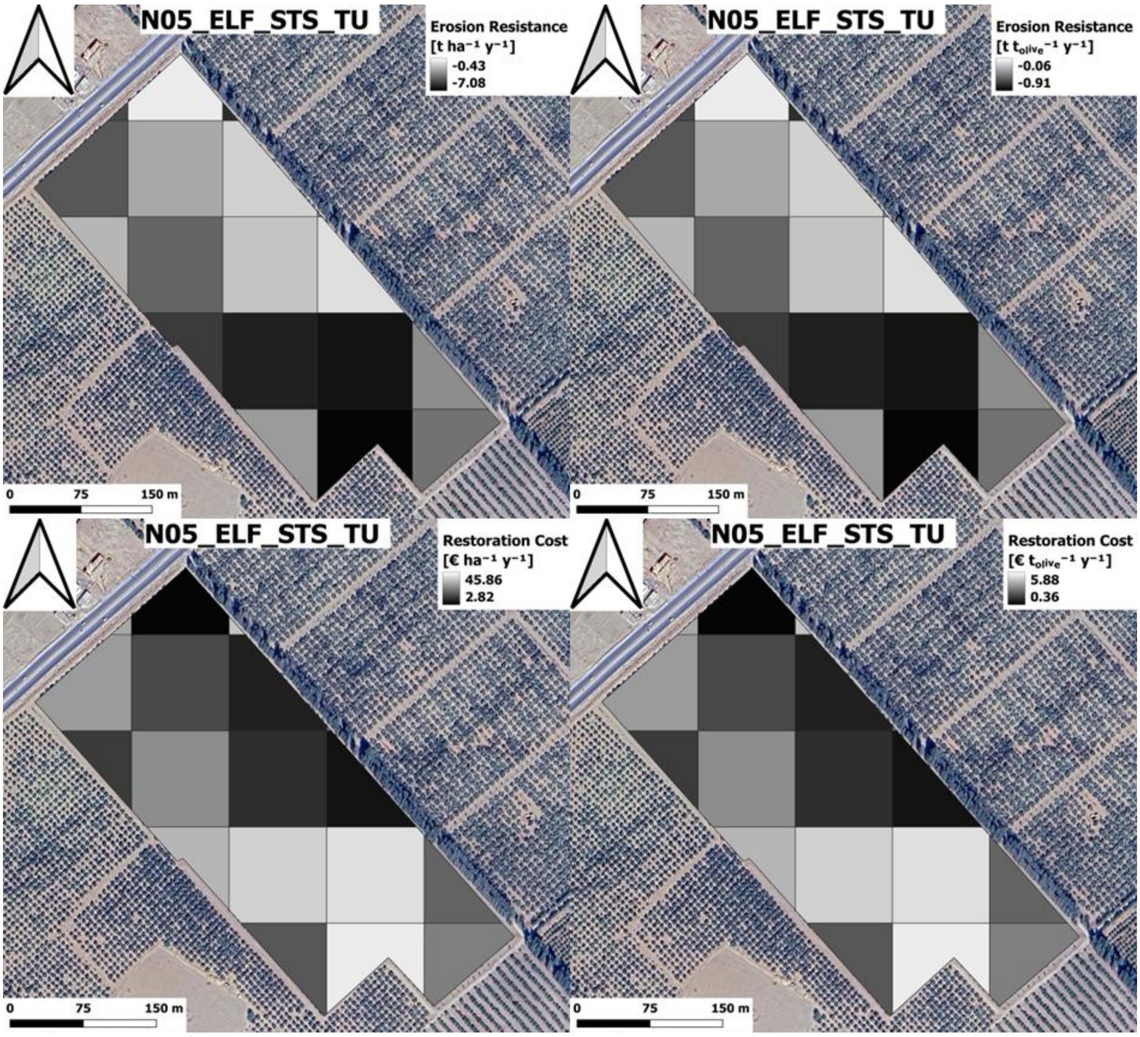
In this section are collected Erosion Resistance maps for Italy. The Coordinate Reference System (CSR) is WGS-84 UTM 32N for all olive farms.



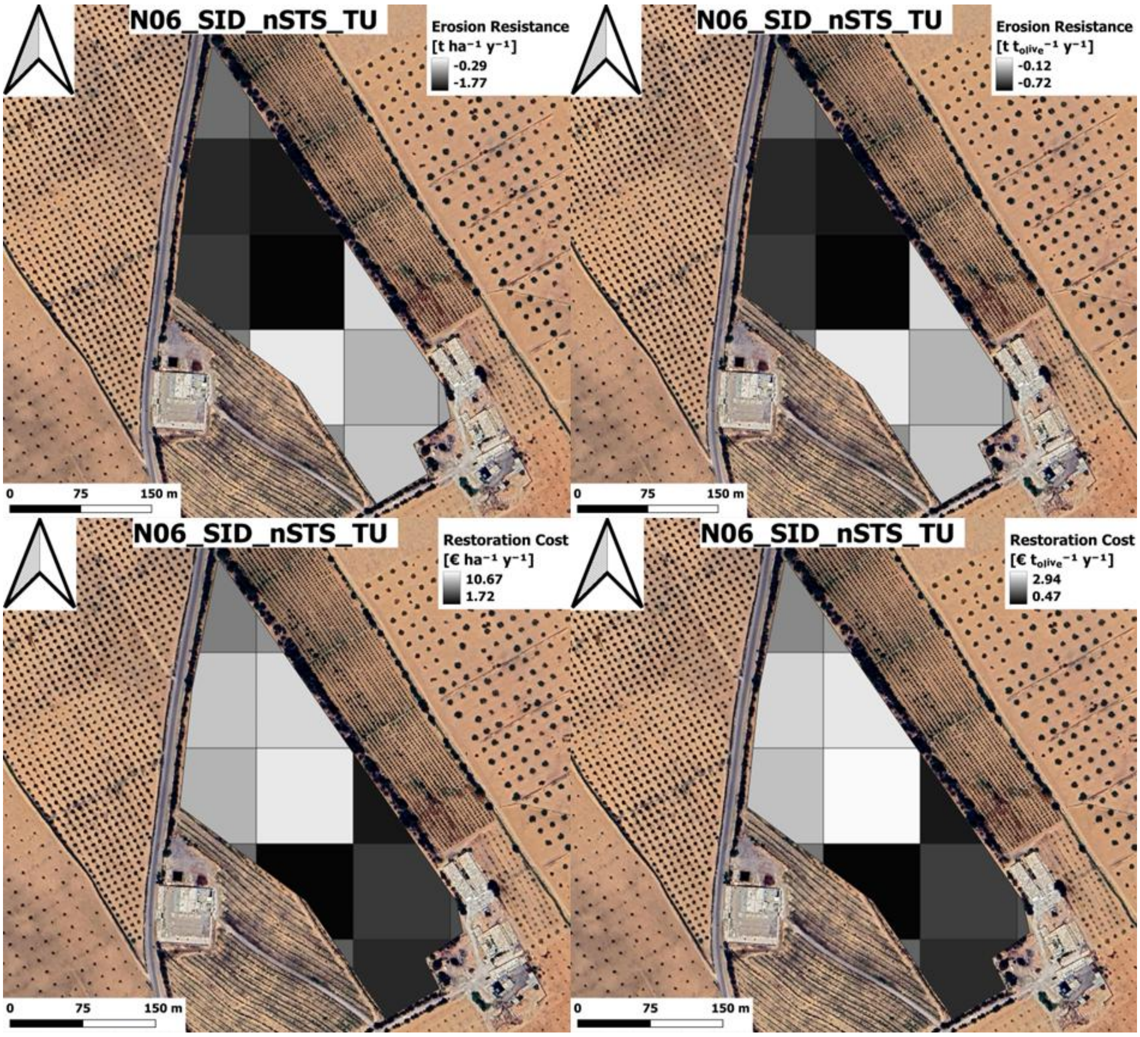

N02_SID_STS_TU
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.27
-5.05
0 100 200 m
N02_SID_STS_TU
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.07
-1.26
0 100 200 m
N02_SID_STS_TU
Restoration Cost
[€ ha⁻¹ y⁻¹]
30.44
1.60
0 100 200 m
N02_SID_STS_TU
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
7.60
0.40
0 100 200 m

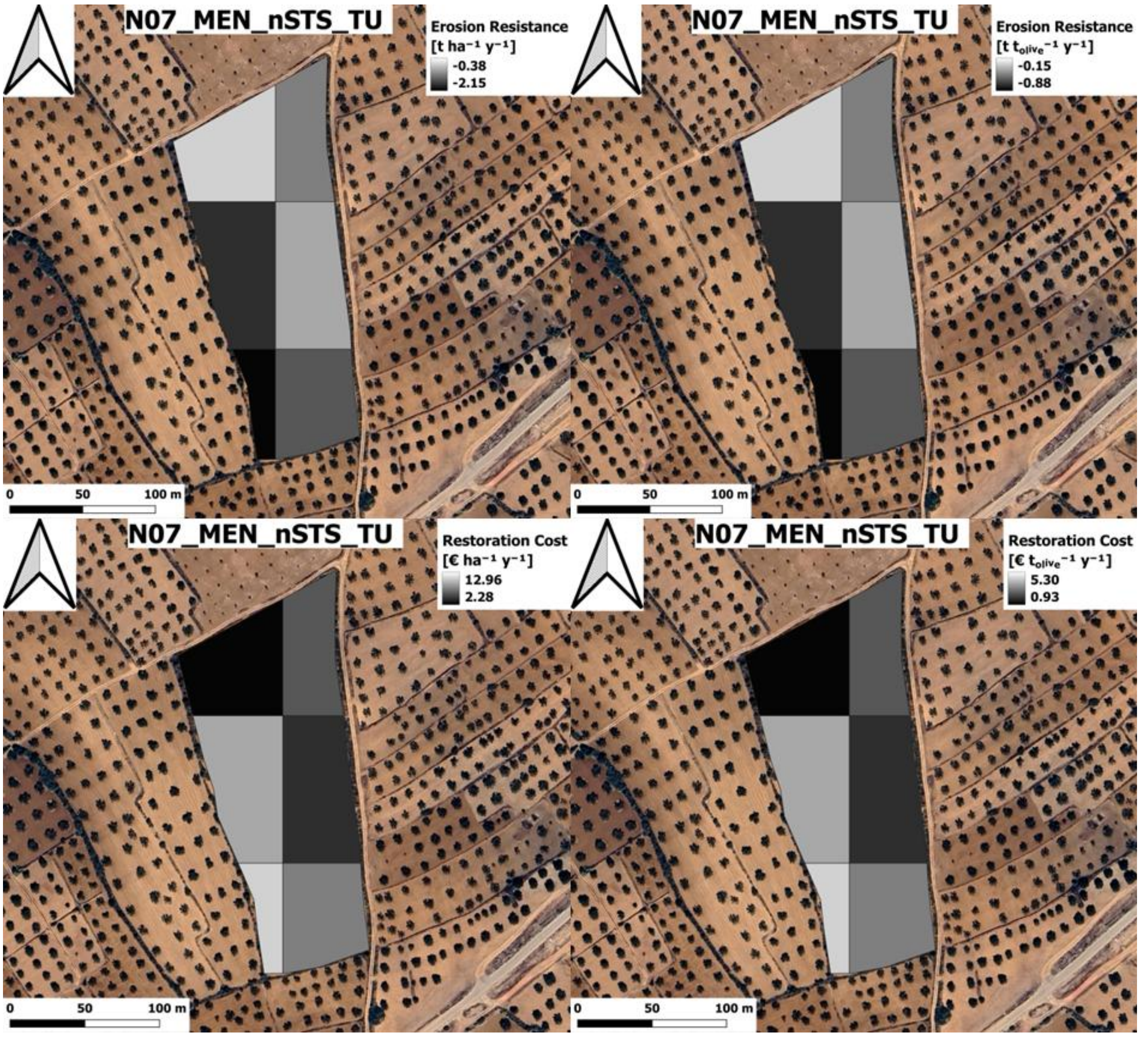

N03_AMA_nSTS_TU
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.64
-2.53
0
50
100 m
N03_AMA_nSTS_TU
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.13
-0.52
0
50
100 m
N03_AMA_nSTS_TU
Restoration Cost
[€ ha⁻¹ y⁻¹]
16.38
4.12
0
50
100 m
N03_AMA_nSTS_TU
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
3.38
0.85
0
50
100 m

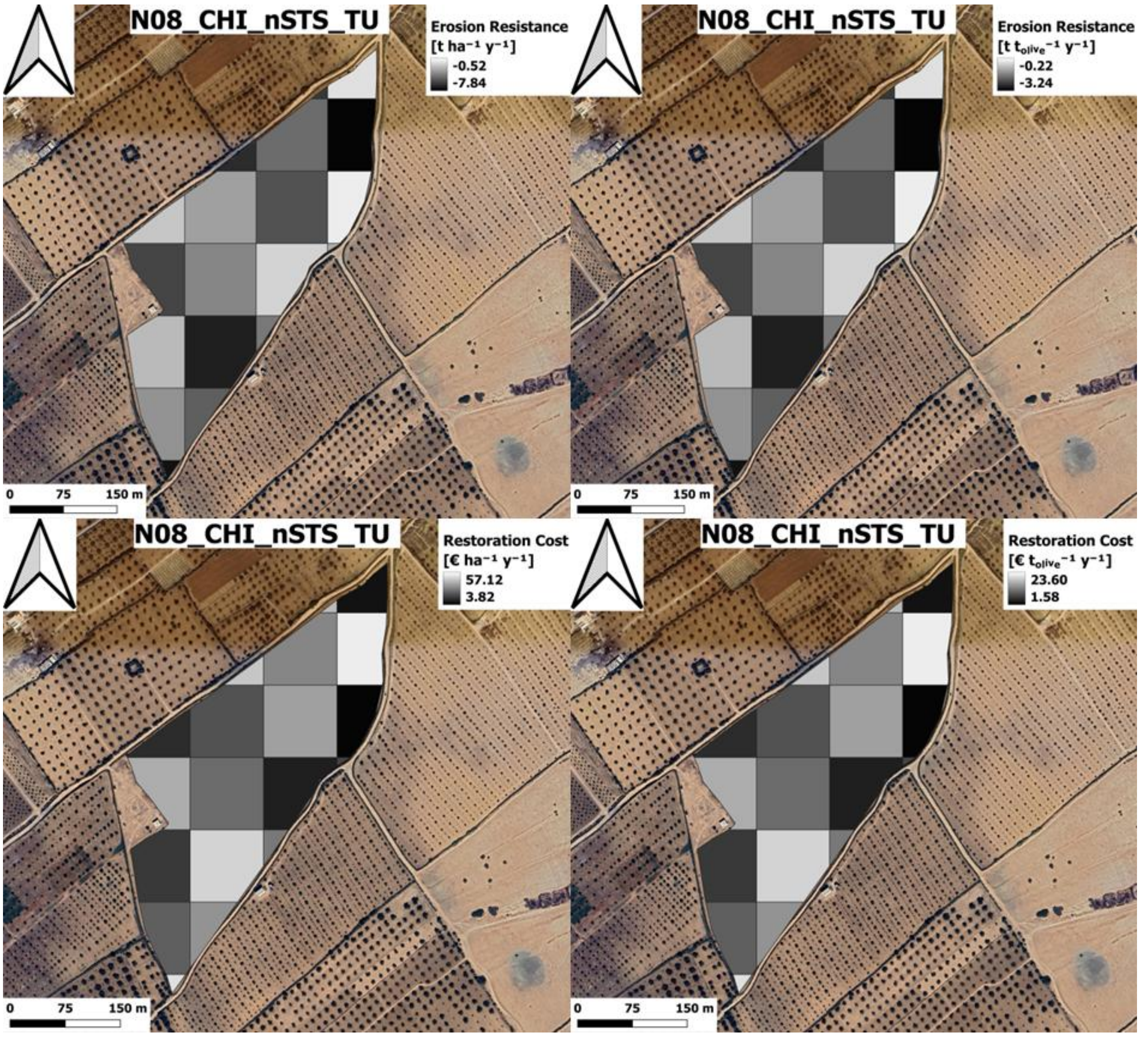

N04_MEN_STS_TU
Erosion Resistance
[t ha^{-1} y^{-1}]
-0.29
-2.01
0
75
150 m
N04_MEN_STS_TU
Erosion Resistance
[t t_{olive}^{-1} y^{-1}]
-0.03
-0.24
0
75
150 m
N04_MEN_STS_TU
Restoration Cost
[€ ha^{-1} y^{-1}]
12.45
1.78
0
75
150 m
N04_MEN_STS_TU
Restoration Cost
[€ t_{olive}^{-1} y^{-1}]
1.49
0.21
0
75
150 m

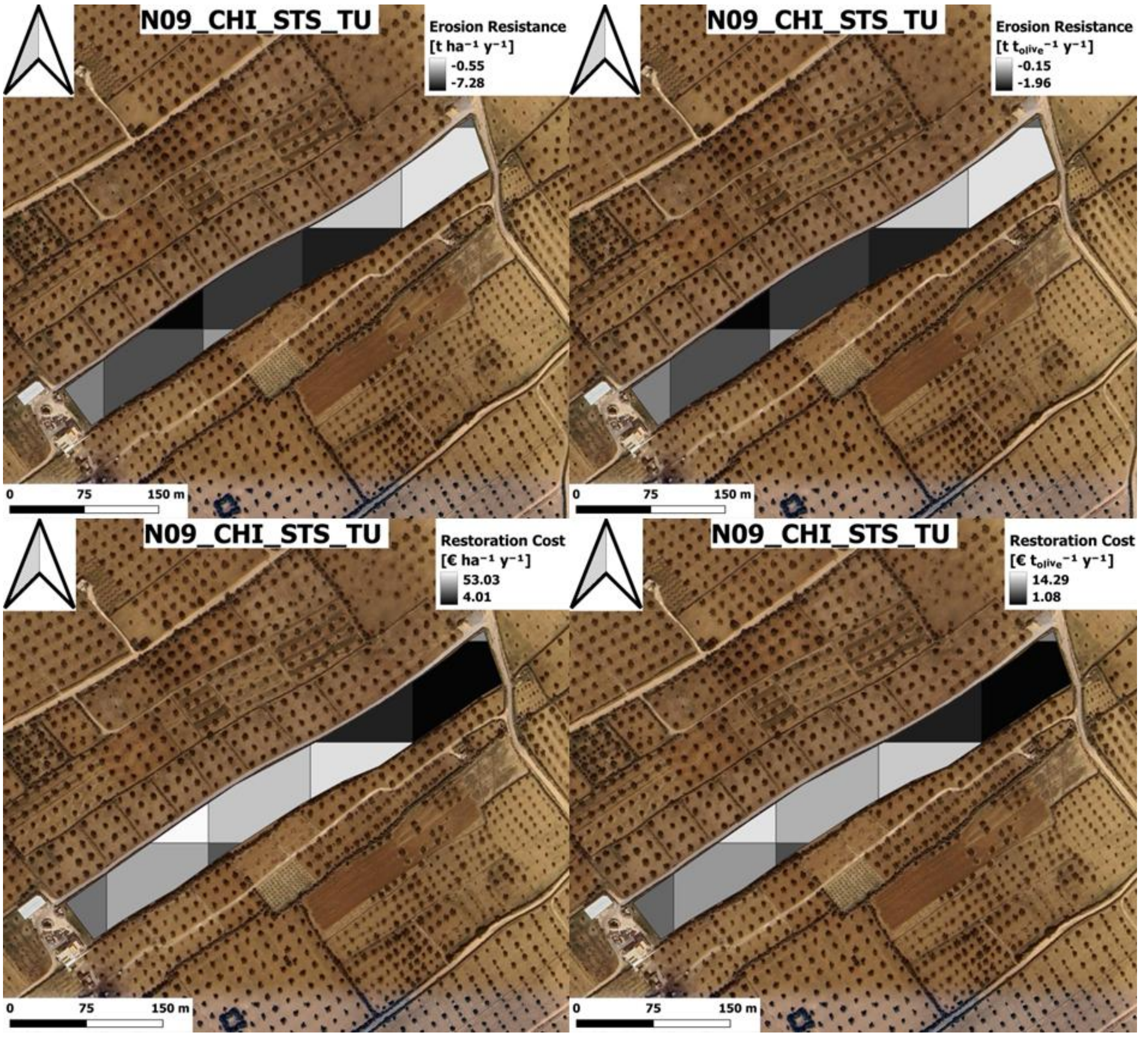


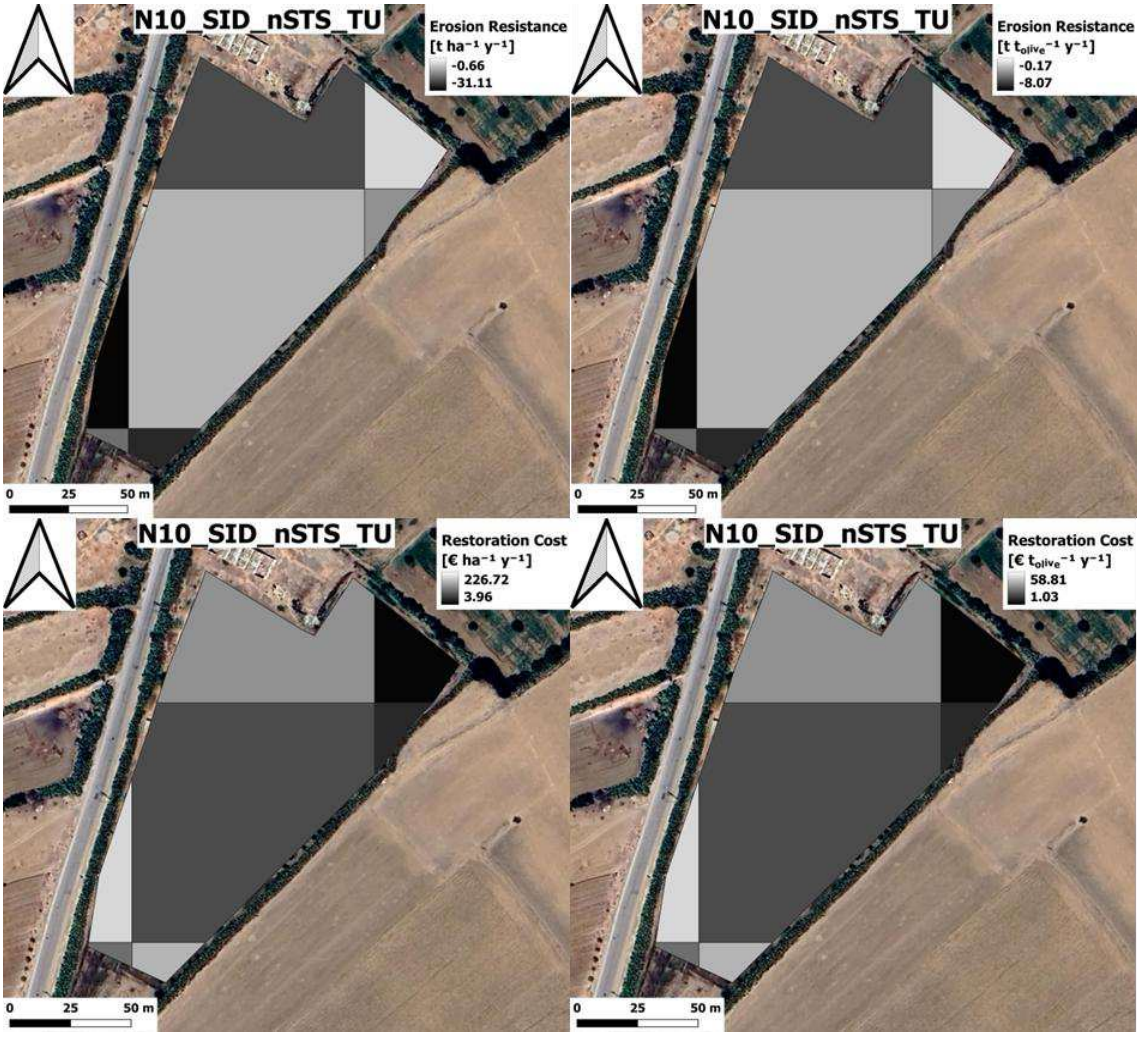
N05_ELF_STS_TU
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.43
-7.08
0
75
150 m
N05_ELF_STS_TU
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.06
-0.91
0
75
150 m
N05_ELF_STS_TU
Restoration Cost
[€ ha⁻¹ y⁻¹]
45.86
2.82
0
75
150 m
N05_ELF_STS_TU
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
5.88
0.36
0
75
150 m

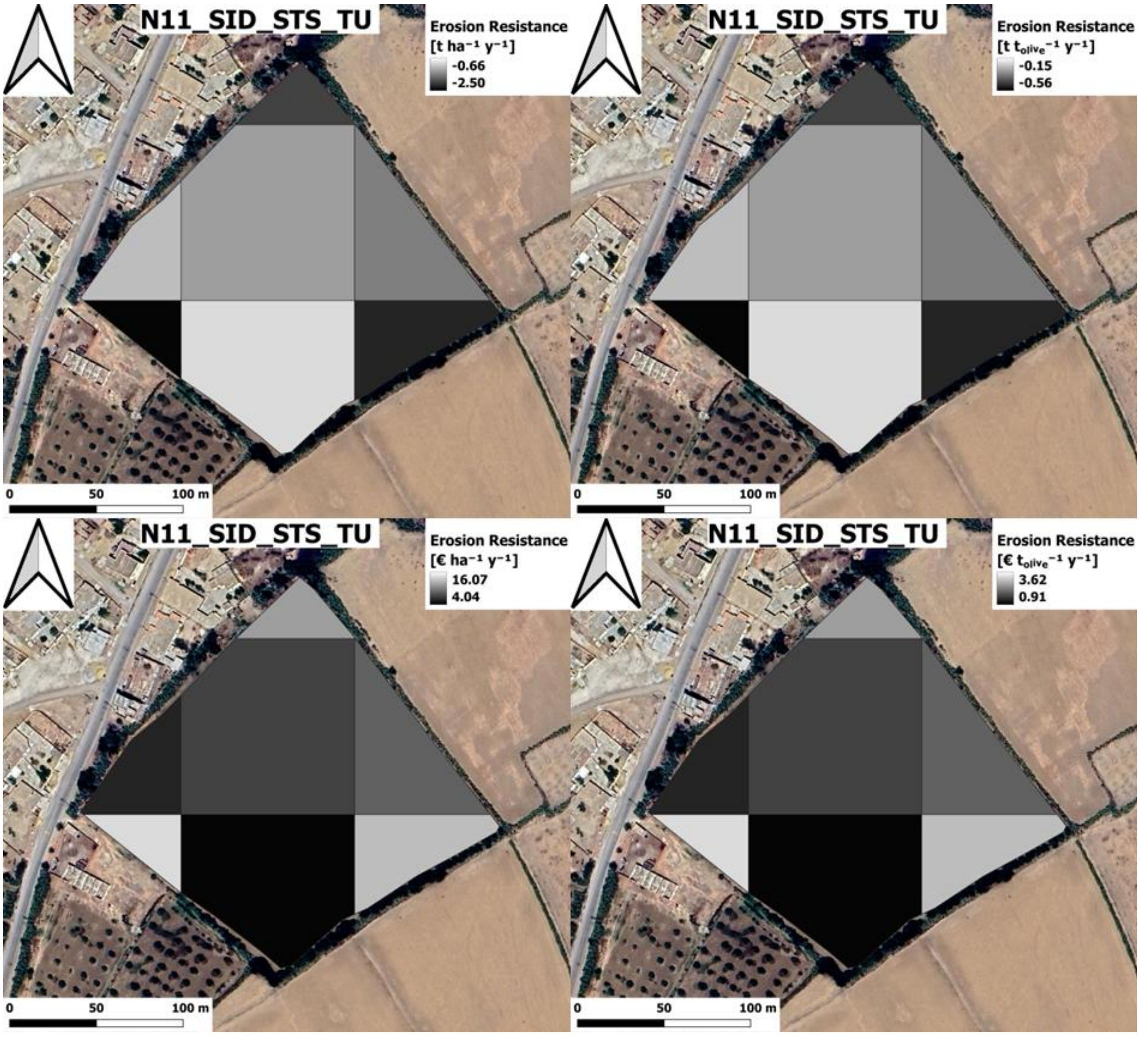

N06_SID_nSTS_TU
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.29
-1.77
0 75 150 m
N06_SID_nSTS_TU
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.12
-0.72
0 75 150 m
N06_SID_nSTS_TU
Restoration Cost
[€ ha⁻¹ y⁻¹]
10.67
1.72
0 75 150 m
N06_SID_nSTS_TU
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
2.94
0.47
0 75 150 m


N07_MEN_nSTS_TU
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.38
-2.15
0 50 100 m
N07_MEN_nSTS_TU
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.15
-0.88
0 50 100 m
N07_MEN_nSTS_TU
Restoration Cost
[€ ha⁻¹ y⁻¹]
12.96
2.28
0 50 100 m
N07_MEN_nSTS_TU
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
5.30
0.93
0 50 100 m


N08_CHI_nSTS_TU
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.52
-7.84
0 75 150 m
N08_CHI_nSTS_TU
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.22
-3.24
0 75 150 m
N08_CHI_nSTS_TU
Restoration Cost
[€ ha⁻¹ y⁻¹]
57.12
3.82
0 75 150 m
N08_CHI_nSTS_TU
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
23.60
1.58
0 75 150 m

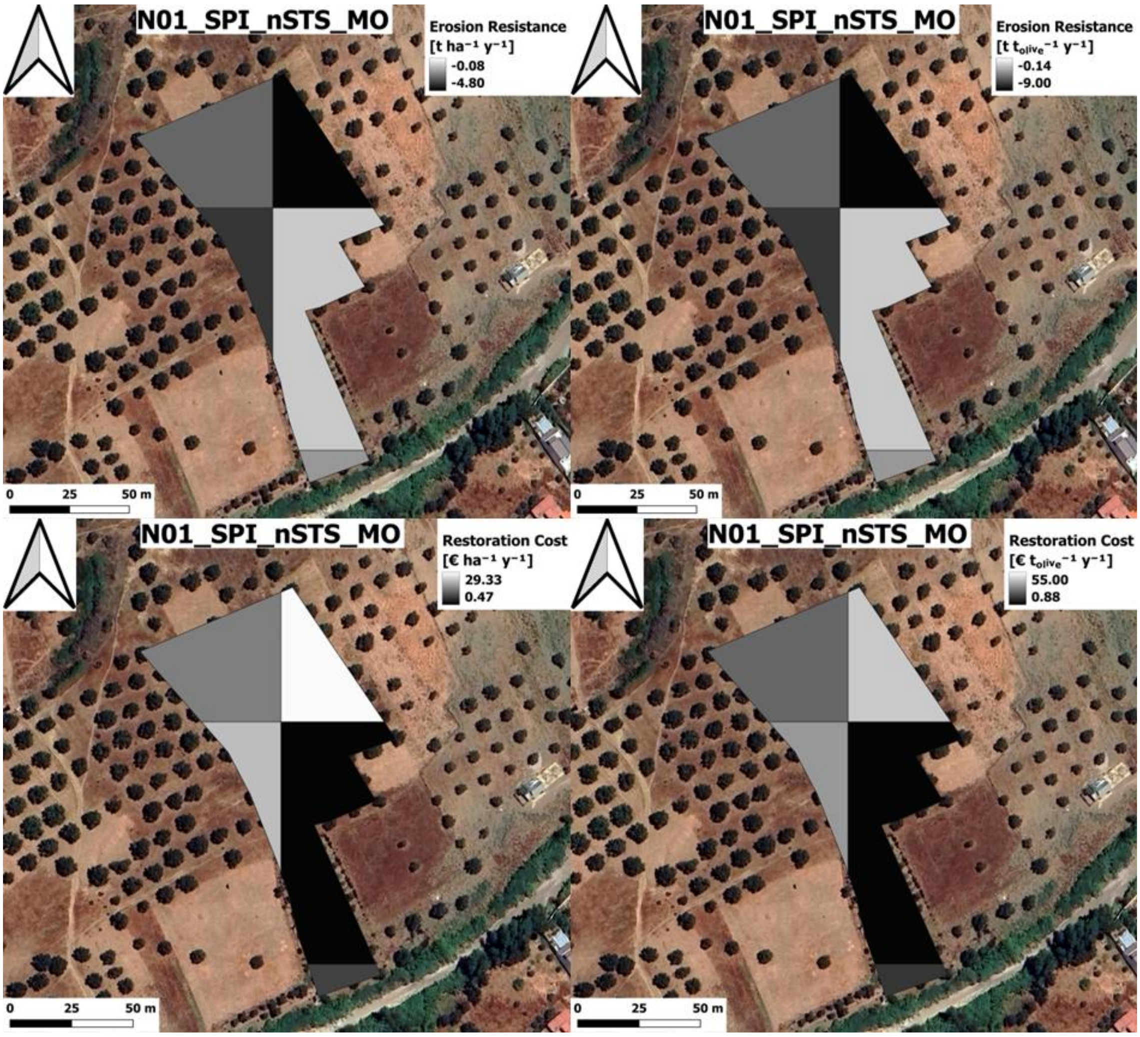

N09_CHI_STS_TU
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.55
-7.28
0
75
150 m
N09_CHI_STS_TU
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.15
-1.96
0
75
150 m
N09_CHI_STS_TU
Restoration Cost
[€ ha⁻¹ y⁻¹]
53.03
4.01
0
75
150 m
N09_CHI_STS_TU
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
14.29
1.08
0
75
150 m

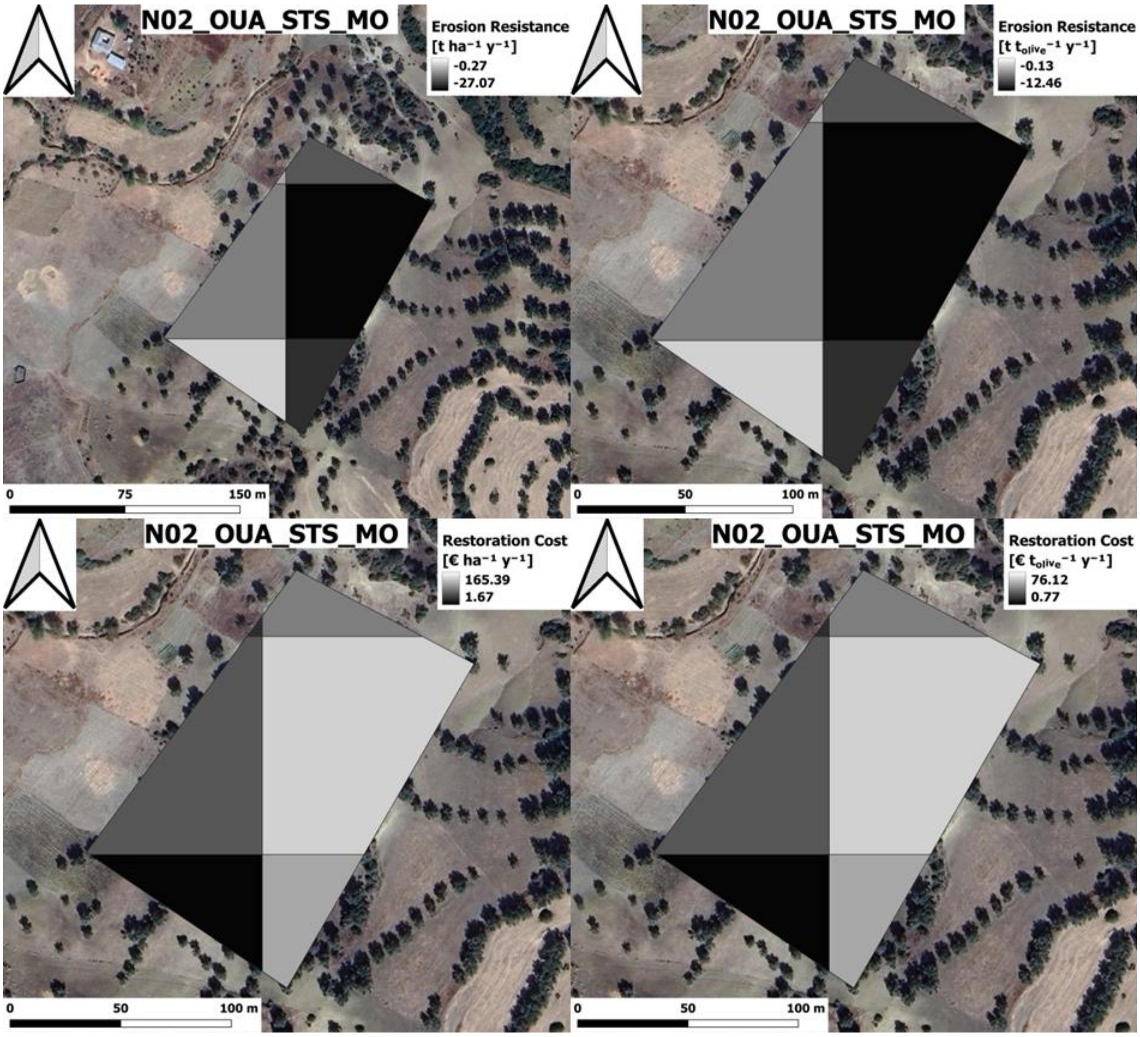

N10_SID_nSTS_TU
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.66
-31.11
0 25 50 m
N10_SID_nSTS_TU
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.17
-8.07
0 25 50 m
N10_SID_nSTS_TU
Restoration Cost
[€ ha⁻¹ y⁻¹]
226.72
3.96
0 25 50 m
N10_SID_nSTS_TU
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
58.81
1.03
0 25 50 m

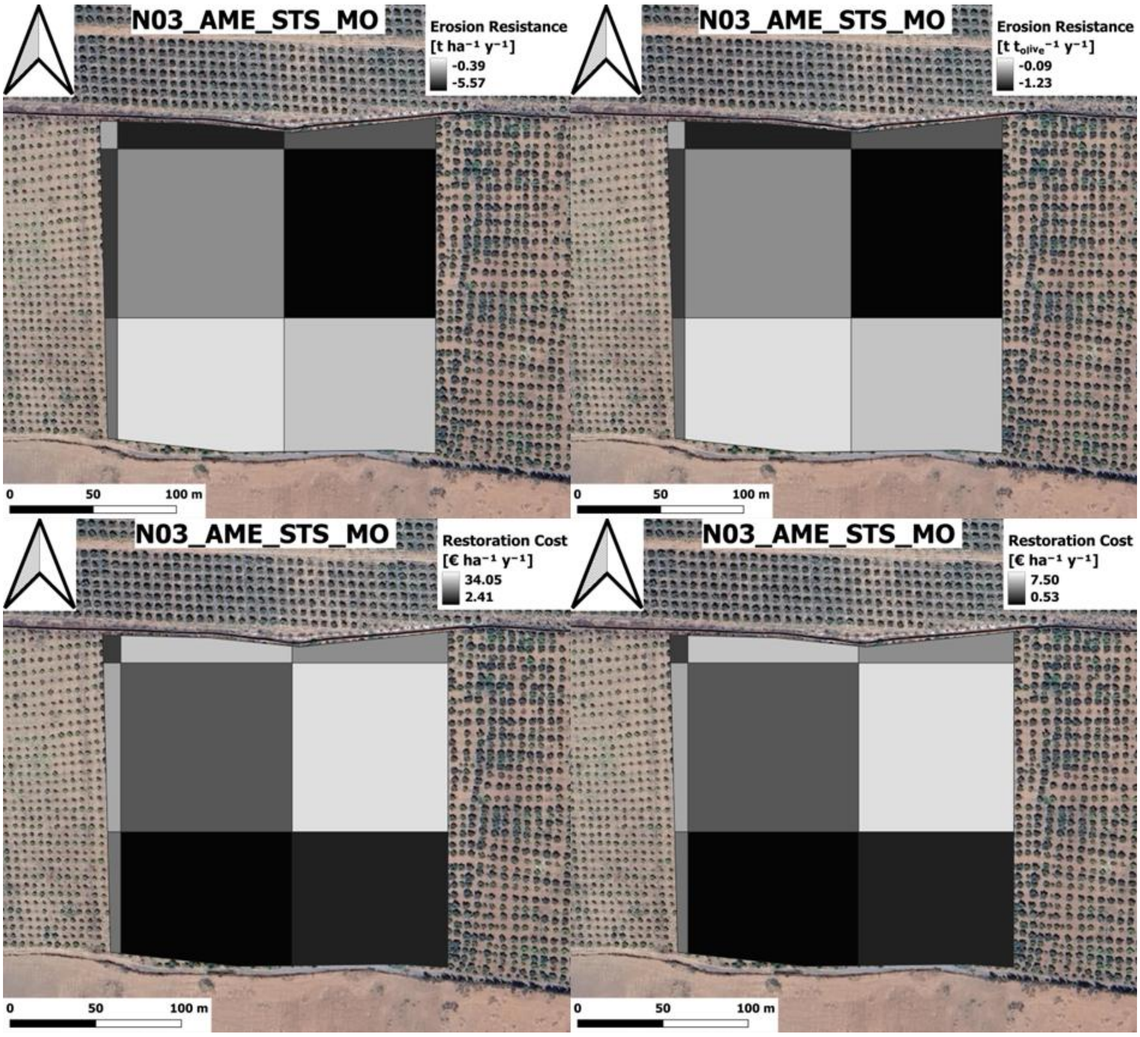

N11_SID_STS_TU
Erosion Resistance
[t ha^{-1} y^{-1}]
-0.66
-2.50
0
50
100 m
N11_SID_STS_TU
Erosion Resistance
[t t_{olive}^{-1} y^{-1}]
-0.15
-0.56
0
50
100 m
N11_SID_STS_TU
Erosion Resistance
[€ ha^{-1} y^{-1}]
16.07
4.04
0
50
100 m
N11_SID_STS_TU
Erosion Resistance
[€ t_{olive}^{-1} y^{-1}]
3.62
0.91
0
50
100 m

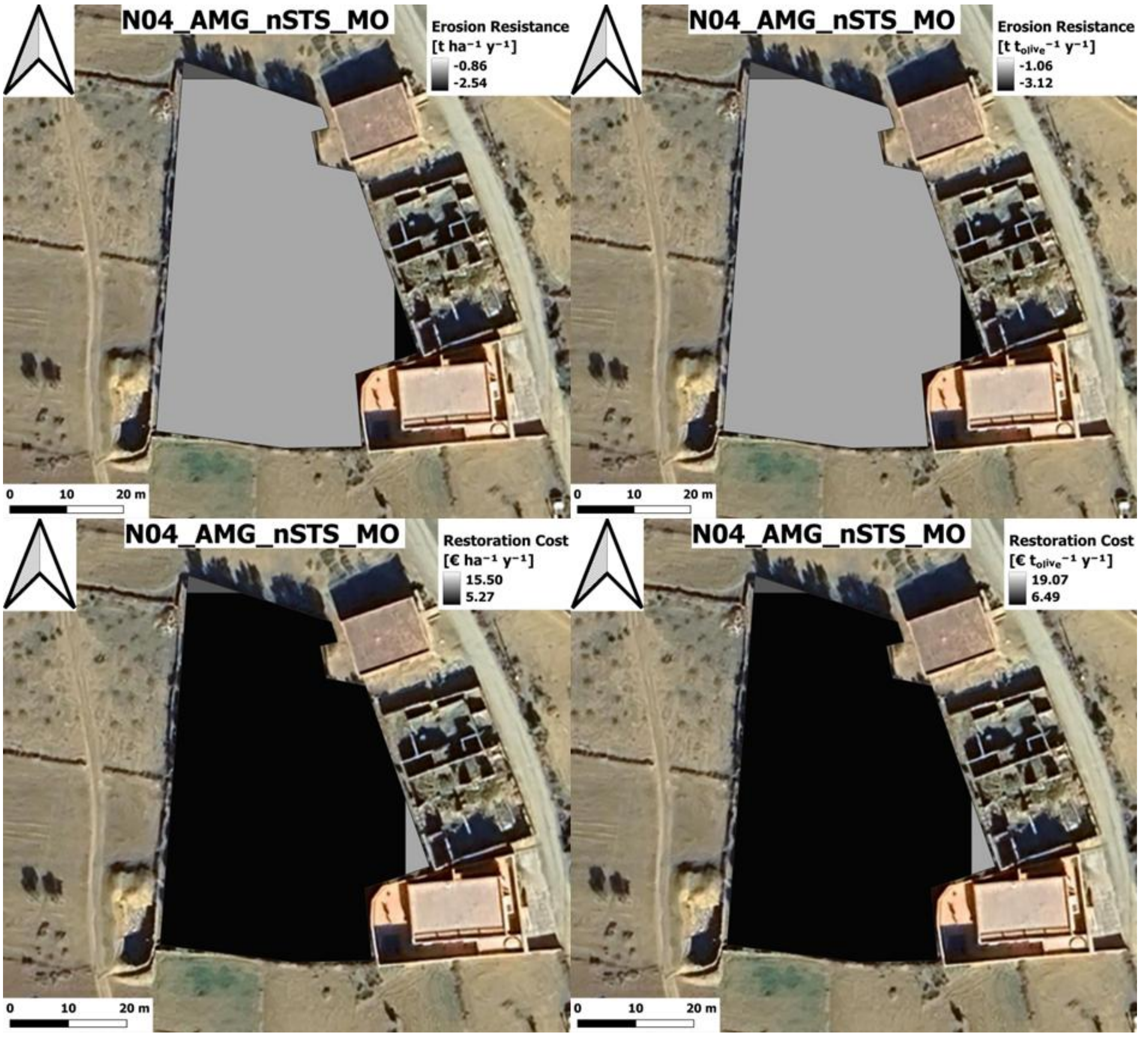
Morocco

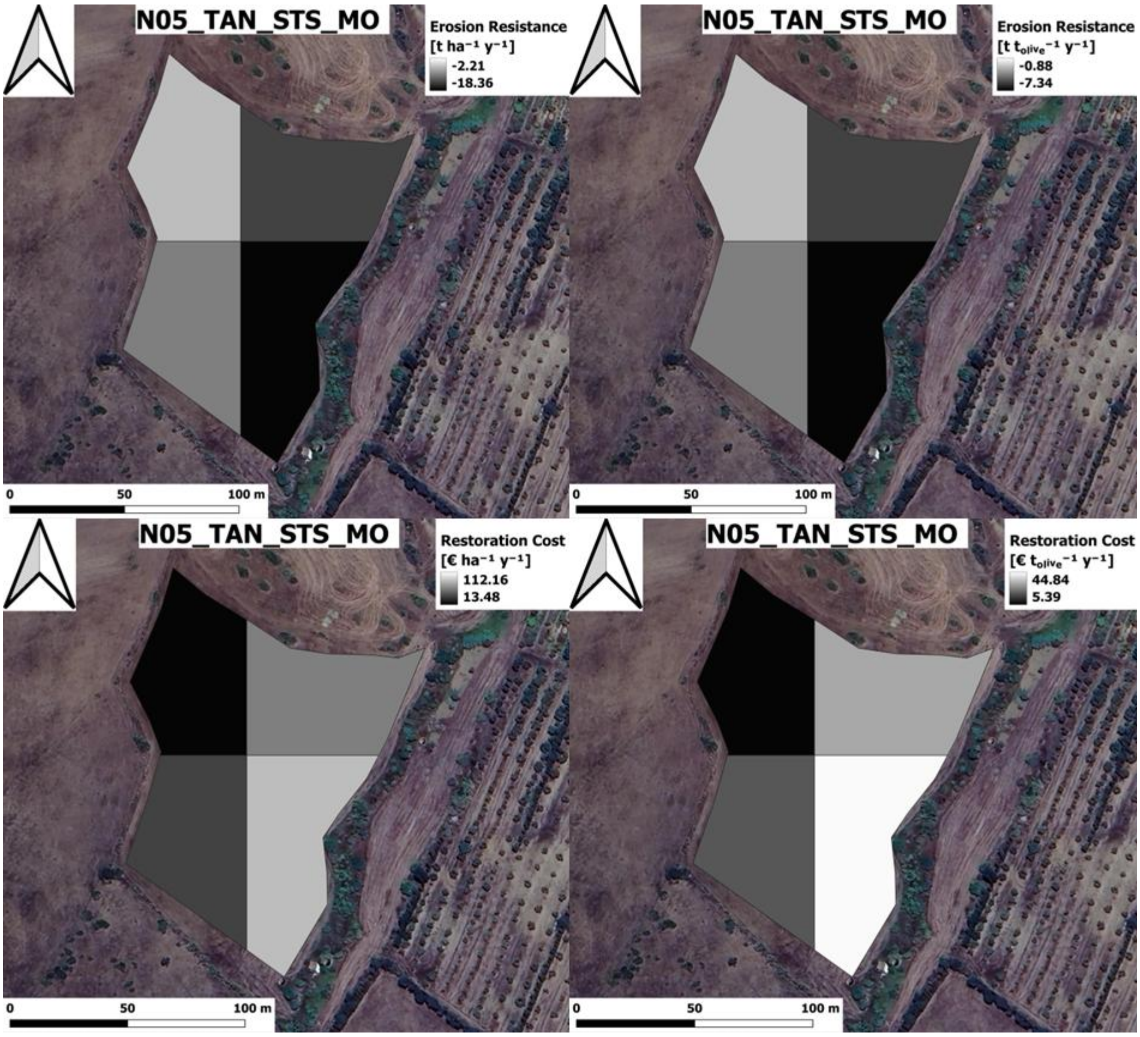
In this section are collected Erosion Resistance maps for Italy. The Coordinate Reference System (CSR) is WGS-84 UTM 30N for all olive farms except for N03_AME_STS_MO and N04_AMG_nSTS_MO which the CSR is WGS-84 UTM 29N.

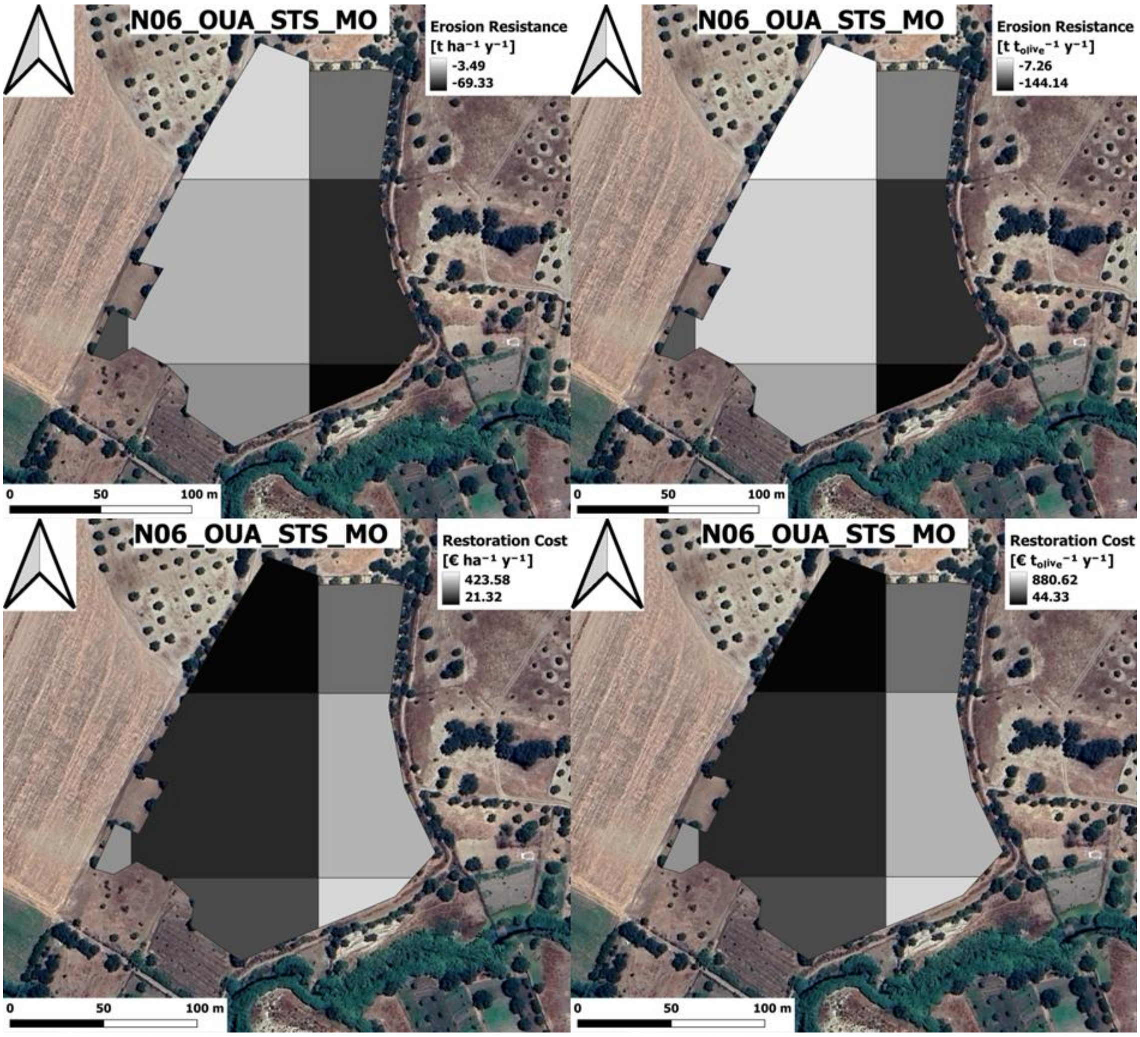


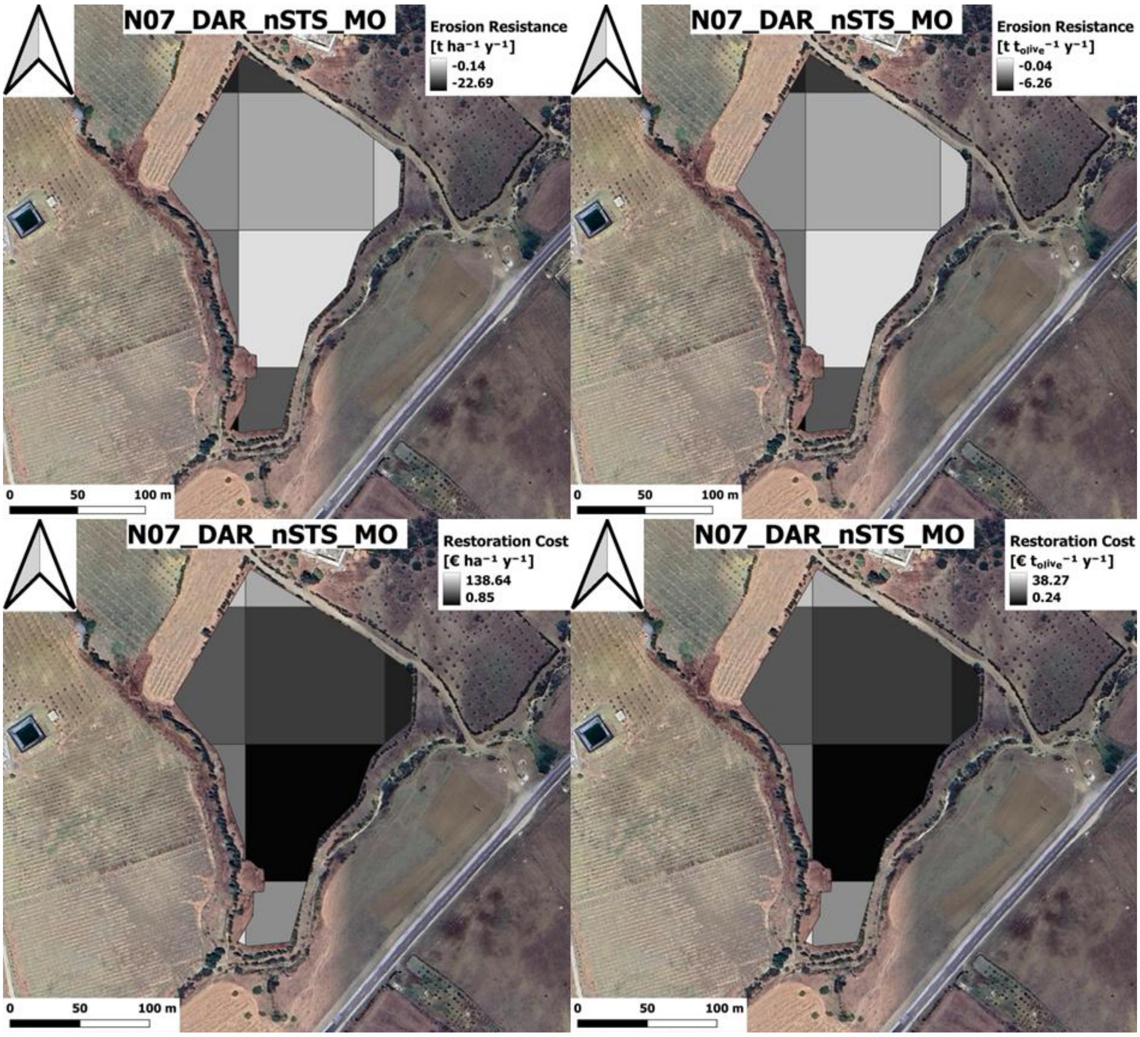

N02_OUA_STS_MO
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.27
-27.07
0 75 150 m
N02_OUA_STS_MO
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.13
-12.46
0 50 100 m
N02_OUA_STS_MO
Restoration Cost
[€ ha⁻¹ y⁻¹]
165.39
1.67
0 50 100 m
N02_OUA_STS_MO
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
76.12
0.77
0 50 100 m


N03_AME_STS_MO
Erosion Resistance
[t ha^{-1} y^{-1}]
-0.39
-5.57
0
50
100 m
N03_AME_STS_MO
Erosion Resistance
[t t_{olive}^{-1} y^{-1}]
-0.09
-1.23
0
50
100 m
N03_AME_STS_MO
Restoration Cost
[€ ha^{-1} y^{-1}]
34.05
2.41
0
50
100 m
N03_AME_STS_MO
Restoration Cost
[€ ha^{-1} y^{-1}]
7.50
0.53
0
50
100 m


N04_AMG_nSTS_MO
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.86
-2.54
0 10 20 m
N04_AMG_nSTS_MO
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-1.06
-3.12
0 10 20 m
N04_AMG_nSTS_MO
Restoration Cost
[€ ha⁻¹ y⁻¹]
15.50
5.27
0 10 20 m
N04_AMG_nSTS_MO
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
19.07
6.49
0 10 20 m


N05_TAN_STS_MO
Erosion Resistance
[t ha⁻¹ y⁻¹]
-2.21
-18.36
0
50
100 m
N05_TAN_STS_MO
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.88
-7.34
0
50
100 m
N05_TAN_STS_MO
Restoration Cost
[€ ha⁻¹ y⁻¹]
112.16
13.48
0
50
100 m
N05_TAN_STS_MO
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
44.84
5.39
0
50
100 m


N06_OUA_STS_MO
Erosion Resistance
[t ha⁻¹ y⁻¹]
-3.49
-69.33
0 50 100 m
N06_OUA_STS_MO
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-7.26
-144.14
0 50 100 m
N06_OUA_STS_MO
Restoration Cost
[€ ha⁻¹ y⁻¹]
423.58
21.32
0 50 100 m
N06_OUA_STS_MO
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
880.62
44.33
0 50 100 m


N07_DAR_nSTS_MO
Erosion Resistance
[t ha⁻¹ y⁻¹]
-0.14
-22.69
0 50 100 m
N07_DAR_nSTS_MO
Erosion Resistance
[t t_olive⁻¹ y⁻¹]
-0.04
-6.26
0 50 100 m
N07_DAR_nSTS_MO
Restoration Cost
[€ ha⁻¹ y⁻¹]
138.64
0.85
0 50 100 m
N07_DAR_nSTS_MO
Restoration Cost
[€ t_olive⁻¹ y⁻¹]
38.27
0.24
0 50 100 m