SELINA
SCIENCE FOR EVIDENCE-BASED AND SUSTAINABLE DECISIONS ABOUT NATURAL CAPITAL

D5.5 Final report: specifying and testing how externalities and disservices can be included in ecosystem accounts

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1 Preface

The importance of biodiversity, natural capital and healthy ecosystems and the services they supply has increasingly been acknowledged in diverse policy initiatives (e.g., the EU nature restoration and amending Regulation from 2024, EU Biodiversity Strategies 2020 and 2030, Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES), UN's Natural Capital and Ecosystem Services Accounting (SEEA EA), Intergovernmental Panel on Climate Change (IPCC) and Convention on Biological Diversity (CBD)).

The EU Horizon Research and Innovation Action "Science for Evidence-based and sustainabLe decIsions about NAtural capital" (SELINA) aims to provide robust information and guidance that can be harnessed by different stakeholder groups to support transformative change in the EU, to halt biodiversity decline, to support ecosystem restoration and to secure the sustainable supply and use of essential Ecosystem Services (ES) in the EU by 2030.

SELINA builds upon the Mapping and Assessment of Ecosystems and their Services (MAES) initiative that has provided the conceptual, methodological, data and knowledge base for comprehensive assessments on different spatial scales, including the EU-wide assessment (Maes et al. 2020) and assessments in EU member states. Knowledge and data for different ecosystem types are increasingly available.

The overall objective of Work Package (WP) 5 "Ecosystem Accounting" is to integrate insights from ecosystem conditions and ecosystem services into the UN System of Environmental Economic Accounting Ecosystem Accounting (SEEA EA) framework. It addresses key challenges such as incorporating externalities, improving the accounts' spatial and temporal resolution, and exploring how different valuation methods influence ecosystem services and asset values.

This document is a Deliverable for WP5 of the SELINA Project, involving a study on how ecosystem disservices and externalities can be aligned with ecosystem accounting following the UN System for Environmental Economic Accounting.

2 Summary

Negative externalities and ecosystem disservices are important for ecosystem and environmental management but these two concepts are not integrated in the System of Environmental Economic Accounting (SEEA) Ecosystem Accounting (EA) framework. The objective of SELINA Task 5.1, therefore, is to examine whether and how negative externalities and/or ecosystem disservices can be connected to the SEEA EA methodology. Deliverable D5.5 presents and discusses the definitions of negative externalities and ecosystem disservices, overlaps and differences between the two concepts and how they are connected to ecosystem accounting. Based on this conceptual and methodological background, we propose that four categories of ecosystem disservices and negative externalities can be integrated in the SEEA EA: (1) ecosystem disservices that are not the direct consequence of current human use or activity (i.e., they are not externalities), (2) ecosystem disservices that occur, or are enhanced, as a consequence of human use or activity (i.e., a negative externality of the activity), (3) reductions in ecosystem service supply as a consequence of current human use or activity (i.e., a negative externality of the activity; not an ecosystem disservice), and (4) negative externalities resulting from specific human activities that involve a decrease in ecosystem condition.

We outline potential approaches for including ecosystem disservices and negative externalities in ecosystem accounts, making a distinction between ecosystem disservices that can be measured as the direct inverse of an ecosystem service and those that cannot. We also identify and propose solutions for challenges that may be encountered, such as how to record negative externalities arising from ecosystem type conversions, the recording of intermediate ecosystem disservices and double-counting issues, and monetary valuation approaches.

Our proposals have been tested in three SELINA test sites. Detailed reports of the scope of work, methods and results obtained by these test sites are presented in the Annexes of this report, one per test site. The main report synthesises the results of these pilots in the test sites, and the lessons drawn related to the integration of ecosystem disservices and negative externalities in ecosystem accounts. Those findings will inform the content of the SELINA Compendium of Guidance, and will feed the Deliverable D5.4, which will articulate key findings of WP5 tasks to provide recommendations for the implementation of SEEA EA in the EU and globally.

3 List of abbreviations

AGB	Above Ground Biomass
BCEF	Biomass Conversion and Expansion Factor
BGB	Below Ground Biomass
CBD	Convention on Biological Diversity
C	Carbon
CLC	Corine Land Cover
CO_2	Carbon dioxide
CICES	Common International Classification of Ecosystem Services
CVM	Contingent Valuation Method
DOM	Dead Organic Matter
EA	Ecosystem Accounting
EDS	Ecosystem Disservices
EFFIS	European Forest Fire Information System
EO	Earth Observation
ES	Ecosystem Services
ET	Ecosystem Type
ESVD	Ecosystem Services Valuation Database
EU	European Union
GEE	Google Earth Engine
IFN	National Forest Inventory
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
LMGL	Lowest Mean Groundwater Level
LULC	Land Use Land Cover
MA	Millennium Ecosystem Assessment
MAES	Mapping and Assessment of Ecosystems and their Services
MMF	Morgan Morgan Finney
NCA	Natural Capital Accounting
N	Nitrogen
NESCS	National Ecosystem Services Classification System
NPV	Net Present Value
NL	The Netherlands
NUTS	Nomenclature of Territorial Units for Statistics
NCP	Nature's Contribution to People
OECD	Organisation for Economic Cooperation and Development
SCC	Social Cost of Carbon
SEEA	System of Environmental Economic Accounting
SEEA EA	System of Environmental Economic Accounting Ecosystem Accounting
SEEA CF	System of Environmental Economic Accounting Central Framework
SNA	System of National Accounts
TCM	Travel Costs Method
TEEB	The Economics of Ecosystems and Biodiversity
UNFCCC	United Nations Framework Convention on Climate Change

4 Introduction

Externalities and ecosystem disservices (EDS) are important concepts for ecosystem and environmental management. Externalities include positive and negative side effects of economic activities, and may have major environmental implications e.g. in the form of emissions of pollution to air, water or soil. Whereas there is increasing attention for Ecosystem Services (ES) in environmental management, the negative contributions of ecosystems to human well-being (e.g. transmission of vector-borne diseases, loss of biodiversity due to invasive species, damage to crops and infrastructure by pests, emission of greenhouse gases, injury or mortality by dangerous species), also called EDS, have not received a comparable level of attention in the ES literature (Shackleton et al., 2016).

Negative externalities and EDS, are not (yet) identified by and integrated in the System of Environmental Economic Accounting (SEEA) Ecosystem Accounting (EA) framework, even though they are important for environmental management. Since accounting principally deals with services with positive (economic) value, it has proven complex to include negative externalities and EDS in the SEEA EA framework as discrete accounting elements. Also, SEEA EA accounts, building upon and aligned with the System of National Accounts (SNA), in principle include only positive numbers reflecting services and assets. The SNA also assumes that all economic goods and services can only have a positive value – there is no entry in the SNA for products with a negative value. Therefore, currently, the effects of negative externalities and EDS may be implicitly captured by ecosystem accounts, for instance in terms of change of ecosystem condition or flows of ES, but are not explicitly captured in specific accounting records.

At the same time, the SEEA Central Framework (SEEA CF) offers a methodology to account for discharges and emissions, and the SEEA CF is used globally to report emissions of, for instance, greenhouse gases and air pollutants such as particulate matter. The SEEA water account, part of the SEEA CF, allows monitoring and reporting pollutants to waterways. Nevertheless, given that SEEA EA aims to be a 'one-stop' comprehensive decision-support system for ecosystem management, it is of interest to explore whether and how negative externalities and/or EDS can be included in SEEA EA. Even though SEEA EA has now become a statistical standard (Edens *et al.*, 2022), the concept and application of EA is still evolving and further revisions of the SEEA EA, or future extensions of the way it is applied, are likely.

The objective of this SELINA D5.5 report, therefore, is to examine whether and how negative externalities and/or EDS can be connected to the SEEA EA methodology. We build upon an earlier paper prepared for the UN statistics Division by Markandya *et al.* (2019), extending this earlier work with further analyses, examples and proposals. Sections 5 and 6 of the report present the conceptual and methodological background, and make several proposals for including the concepts in SEEA EA. Section 7 presents test sites that tested these proposals. Sections 8 and 9 present findings from these tests and conclusion on the integration of negative externalities and EDS in SEEA EA.

This Deliverable D5.5 report builds upon and extends conceptual works presented in the Deliverable D5.1 report, integrating lessons learnt from the test sites pilots. Towards the end of SELINA, this report will be worked into a scientific publication with all contributors to Task 5.1 as co-authors.

5 Concepts and definitions

5.1 Definition of an externality

Externalities are unintended impacts of actions by producers, consumers or communities on other stakeholders in the society. The OECD defines externalities as “situations when the effect of production or consumption of goods and services imposes costs or benefits on others which are not reflected in the prices charged for the goods and services being provided” (Khemani and Shapiro, 1993). Externalities, in the economics literature, therefore require some agent (individual, household, enterprise or community body) to be responsible for the action that has an impact on the well-being of others (Markandya *et al.*, 2019). This means that natural phenomena that generate positive (or negative) effects on welfare without any human involvement are not externalities. In practice, though, anthropogenic and natural causes can be hard to disentangle in many cases, e.g. in the case of forest fires that may be ignited by human actions or lightning, and spread faster as a consequence of climate change and human-induced changes in landscape flammability. Usually there is also a spatial dimension to externalities: the source area of the externalities and the impacted area can overlap or be spatially disjointed and linked by connectivity processes, for instance when water pollution resulting from an agricultural or industrial activity affects downstream water users. Analogous challenges often occur in the assessment of ESs, between areas where ecosystems provide ESs and areas where humans benefit from these ESs (Dworczyk and Burkhard, 2021). There may also be a temporal dimension, i.e., a time lag between the externality being generated and its impact on stakeholders. For example, alien species introduced deliberately to provide a benefit (e.g. water hyacinths used for pond/lake aesthetics), may become invasive over time and affect other economic activities, as externalities (e.g., river blockage for boats; suffocation of waterways).

Externalities can be both positive and negative. They are negative when they reduce the well-being of a third party. Positive externalities have an unintended positive impact on the well-being of others. For example, buyers of organic vegetables may do so for health reasons, but their actions can lead to a positive externality because they reduce pesticide use and loss of biodiversity. Acknowledging that there are both positive and negative externalities, this report is exclusively focused on negative externalities and how they can be included in ecosystem accounts, given that this is often of particular importance to ecosystem management (e.g., in dealing with emissions and discharges from ecosystems). Furthermore, importantly, positive externalities from activities that affect ecosystems can usually be captured in ecosystem accounts as the enhancement of ES supply. For instance, carbon sequestration in forests could be considered a positive externality of forest management aimed at timber production.

The definition of externalities is an anthropocentric one and it is worth noting that there are alternative value systems (Pascual *et al.*, 2023). Social scientists, for instance, make a distinction between the social value of a transaction and the market value, noting that the former may be much higher than the latter. Values in psychology relate to emotions and principles and goals, which guide human behaviour. In environmental sciences, as well as in philosophy, values relating to the living environment are seen as endowing the latter with certain inalienable legal rights, which means that the living environment has value in and of itself, separate and independent from the benefits humans may derive from it for their own purposes (also referred to as intrinsic values) (e.g., Pearce and Turner, 1990). Ecological value refers to the "perceived importance of an ecosystem, which is underpinned by the biotic and/ or abiotic components and processes that characterise that ecosystem" (Barton *et al.*, 2019). The use of the term externality as elaborated in this report does not seek to deny these other perspectives but notes that its use as a tool of policy is mainly framed in economic terms and it is this definition that forms the basis of the discussion in this report.

5.2 Definition of ecosystem disservices

The literature on EDS has been growing since the 2000s. However, a widely accepted definition, conceptual framework and typology of EDS remain elusive (Campagne *et al.*, 2018). One difficulty is that there is a lack of consensus on what constitutes the distinction between EDS and externalities, and whether the concept should be restricted only to negative impacts, which result directly from ecosystem structures, functions and processes (as with ES), or whether it should also include those impacts which may be mostly or entirely precipitated by human activity. The issue is compounded by the fact that discussion of EDS in the literature has largely come from the perspectives of ecological and ES sciences, despite the broad social and economic implications of EDS.

For the purposes of this report, we define EDS as "the ecosystem-generated functions, processes and attributes that result in perceived or actual negative impacts on human well-being", after Shackleton et al. (2016). Examples often presented in the literature include transmissions of vector-borne diseases, loss of biodiversity due to invasive species, damage to crops by wildlife, damage to buildings by termites, emission of greenhouse gases, human morbidity due to parasites, and human injury or mortality by dangerous species. Recognizing that these examples are not always accepted as EDS and raise various scientific and policy issues, a health-focused commentary on EDS, externalities and ecosystem accounting is provided in Annex 1.

Building upon the ES conceptual framework, the IPBES came up with an adapted framework: Nature's Contributions to People (NCP). The IPBES reports highlight that contributions can be both positive and negative. Negative contributions are those perceived as detrimental or harmful by different (groups of) stakeholders or by the same stakeholders but in different socioeconomic, temporal or spatial contexts (Lliso et al 2022). In the IPBES framework, the terminology 'negative NCP' is used as a synonym of 'detriment' or 'detrimental contribution'. Those negative contributions should be deliberately defined, accounted for and valued to better identify social–ecological trade-offs (Lliso *et al.*, 2022).

While EDS have been part of the ES theoretical framework (Braat, 2018), the consideration of negative contributions of nature has been claimed as being more explicit in the definition of NCP (Díaz *et al.*, 2018). Yet, this claim might not be entirely justified and the negative contributions in the IPBES framework are still loosely defined. Kadykalo *et al.* (2019) argued that theoretically the frameworks seem similar in terms of recognizing negative effects on human well-being. The way IPBES defines negative NCP does not differ from EDS, reinforcing the idea that EDS and negative NCP are very much aligned. However, the NCP framework explicitly recognizes the fact that generally NCP are not inherently positive or negative and makes clear that the contributions can be defined and valued as negative, neutral or positive. The value of a NCP can be (understood as) either positive, neutral or negative, depending who perceives the NCP, when and where (Lliso *et al.*, 2022).

EDS are distinct from negative externalities in that there is no requirement for an EDS to be caused by human action. However, the occurrence of EDS is often related to a current or past human activity (see also commentary in Annex 1), and it is therefore difficult to disentangle the underlying natural and human causes. For example, wildlife trampling of crops may occur after forests have been converted to croplands and the wildlife is faced with reduced habitat and food sources; or the decline of endemic species is due to the presence of invasive species that were intentionally or unintentionally introduced by humans. EDS may also be caused by a lack of risk awareness or risk acceptance on the part of humans, for instance by settling in flood-prone areas or volcanically active areas (benefitting from fertile soils but with risk of volcano eruption). This challenge has a corollary in the assessment of ES and the separate quantification of the role of ecosystem inputs and human inputs in the production of ecosystem goods and services. The disentanglement of underlying natural and human causes of EDS, particularly those with historical origins, is likely to be intractable and we therefore propose that EDS should be measured without attempting to attribute responsibility. Similarly, the identification of EDS that are of purely natural origin is rarely likely to be feasible, since all EDS involve people as recipients. Moreover, the distinction between purely natural and human influenced EDS is not necessarily useful for decision-making. From the perspective of current ecosystem management, the measurement of the quantity and value of an EDS is arguably more relevant than the (historical) responsibility for its occurrence.

Conceptually, and from an accounting perspective, we propose to consider EDS as distinct from negative externalities. The key point of distinction is the role of human agency:

1. An EDS is a negative contribution from an ecosystem to human well-being, irrespective of the role of human agency in the underlying processes.
2. A negative externality is an unintended negative consequence of human action on the well-being of a third party. In the context of EA, externalities can take three forms (see also Figure 1):
    a. An increase in the provision of an ecosystem disservice
    b. A reduction in the provision of an ecosystem service
    c. A decrease in ecosystem condition

Note that negative externalities may also directly affect people, i.e. not by means of affecting an ecosystem and the services it provides. In particular, negative externalities may involve exposure of people to harmful chemicals, e.g. in ambient air. These externalities are relevant in the context of the SEEA CF (that records emissions to air) and the SEEA EA (that records ecosystem condition) – but they would not lead necessarily to a reduction of an ecosystem service. To the contrary, discharges to water would generally lead to an impact on an ecosystem service, e.g. fish provisioning or providing opportunities for bathing.

The commentary in Annex 1 illustrates the complexity of distinguishing between negative externalities and EDS with the example of the emergence of Hendra virus disease, which though initially viewed as a problem caused by opportunistic fruit bats, is now more fully understood as an externality from agricultural practice and habitat loss. This highlights how categorising a phenomenon as an EDS and/or an externality may be subject to change over time as the intricacies of disease ecology and human impacts on ecosystem functioning are more clearly understood, and how attempts to understand, assess, account for and respond to EDS benefit from multi-sector, transdisciplinary approaches.

5.3 Including ecosystem disservices and negative externalities in ecosystem accounts

The purpose of including EDS and negative externalities in ecosystem accounts is to provide information to support environmental management. This purpose underpins the consideration of which EDS and negative externalities are potentially relevant. We propose that four categories of EDS and negative externalities can be integrated in the SEEA EA (see Fig. 1):

1. **EDS that involve negative effects from ecosystems to people that are not directly related to human interventions in an ecosystem**. (i.e., without making a distinction between natural and human causation). The quantification of these EDS is potentially useful to inform, monitor and appraise environmental management aimed at mitigating such effects. Examples include the loss of biodiversity due to invasive species; spread of diseases by vector species; human injury/mortality due to snakes, dogs, sharks etc.; human morbidity due to parasites.

2. **Externalities that are increases in the flow of EDS attributable to specific human activities.** I.e., these EDS are a type of externalities that result from activities that may either take place in, or at a distance away from the ecosystem. The quantification of the ecosystem impacts of such externalities is potentially useful to inform mitigation of, or compensation for, negative impacts. Examples include peatland drainage for agricultural use that increases the emissions of carbon (the EDS is the emission of carbon due to fire and/or oxidation from peatland; the underlying human activity is the lowering of groundwater levels to facilitate agriculture); conversions of forest to agricultural land that reduce wildlife habitats and eventually lead to crop damage (the disservice is the wildlife damage to crops; the human activity is the forest conversion leading to a reduction in habitat).

3. **Externalities that are reductions in the supply of ES attributable to specific human activities** (i.e., a negative externality of the activity; not a disservice). The quantification of such externalities is potentially useful to inform mitigation of, or compensation for, negative impacts. Examples include reduction in forest recreation due to logging; loss of biodiversity due to intensive tourism; reduction in carbon sequestration due to logging; loss of ES due to human-induced forest fires. ES flows recorded in ecosystem accounts already account implicitly for the effects of externalities, i.e. those ES flows are final flows, net of reductions due to externalities.

However, in the current SEEA EA, these reductions are neither explicitly recorded nor attributed to human activities.

4. **Externalities resulting from specific human activities that involve a decrease in ecosystem condition.** Such externalities impact human well-being through pathways that are not primarily related to ecological characteristics and processes in ecosystems (i.e. ES and EDS), but to changes in physico-chemical characteristics of ecosystems. For instance, the drainage of peatlands leads to soil subsidence and an alteration of soils carrying capacity, with resulting damages to infrastructures (see the Netherlands test site Annex report).

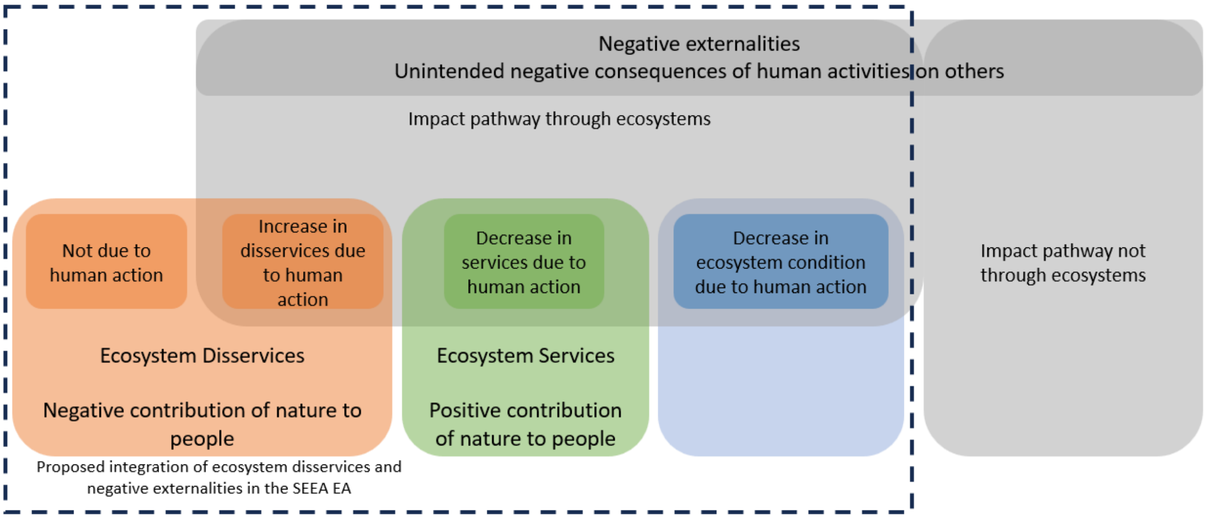

Figure 1: Integration of four categories of EDS and negative externalities in the SEEA EA.

Note that negative externalities may impact human well-being through pathways that are not primarily related to ecosystems, for instance air pollution impacts on human health. In principle, such negative externalities fall outside the scope of SEEA EA and are best linked to the SEEA CF accounts (but air pollution data may be used in SEEA EA as input data for modelling the air filtration regulating ES).
Finally, emissions from machinery used in managing ecosystems are not considered relevant for inclusion in SEEA EA, since this machinery is considered part of the economy in national accounting and those emissions are already captured by the SEEA CF.

6 Approaches and challenges to including ecosystem disservices and negative externalities in ecosystem accounts

In the economics literature, building upon Coase (1960), there are several pathways described how to reduce negative externalities. For instance, actors affected by externalities may negotiate and pay for actions to reduce the externality. This works best when there is a well-understood connection between one or a limited number of polluters, and one or a limited number of actors incurring the externality. In practice, there may be many barriers to leaving it to stakeholders to deal with externalities – for example, in cases when there is incomplete understanding of the externality, the costs of mitigation are prohibitively high leading to competition effects of producers, or there is a lack of trust between actors to negotiate in good faith. Where externalities are considered an excessive burden on society, governments tend to step in with regulations. Hence, there is strong rationale to include negative externalities in ecosystem accounts.

In this section, we outline potential approaches for including EDS and negative externalities in ecosystem accounts. For the three first categories identified in the previous section, we distinguish cases when EDS and negative externalities are measurable with the same metrics as ES (6.1) from cases when it is not the case (6.2). The fourth category is discussed apart, as it does not involve a connection to ES or EDS flows (6.3). We also identify and propose solutions for challenges that may be encountered (6.4).

6.1 Ecosystem disservices and negative externalities that are measurable with the same metrics as an ecosystem service

EDS that are measured with the same metrics as ES, as well as negative externalities that involve a reduction in the supply of ES, may logically best be integrated in or connected to the ES account. For instance, peatland drainage leads to CO_2 emissions that reduce the net sequestration of CO_2 (and carbon storage) in all ecosystems at national scale (or may even lead to net negative sequestration, i.e. emissions, in all ecosystems at national scale). In other words, these EDS lead to an opposite effect – that can be measured with the same metrics of the carbon sequestration service in, say, growing forests. In this case, since the metrics with which the service and the disservice are measured are the same, the disservice can be connected or even deducted from the corresponding entry in the ES account. For example, if part of a forest ecosystem on peat is drained leading to CO_2 emissions, and the remainder of the forest is a net absorber of CO_2, then the ES account could, in principle, indicate the net CO_2 sequestration in this ecosystem asset or ecosystem type. Note that this approach does not conform to the SEEA EA, which proposes that only the positive (gross) sequestration is included in the account. However, it is policy-relevant to consider the net as well as the gross sequestration. Indeed, the legislation for SEEA EA in the EU[1] requires measuring the net sequestration of carbon as the final indicator to be reported (and this may lead to negative values for this indicator). Reporting gross and net CO_2 sequestration is also much more aligned with the greenhouse gas reporting principles of the UNFCCC, that require reporting

[1] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=COM:2022:329:FIN

of both emissions from and sequestration in ecosystems. In order to maintain alignment with SEEA EA, and to be transparent to the users of the account on the underlying data and models, it is important to clearly indicate what the ES is, and what comprises the ecosystem disservice (hence, both gross flows as well as the net flow should be reported). We acknowledge that reporting gross and net flows requires more data compared to reporting only net flows (and this was a motivation behind requiring only reporting net flows in the EU legal proposal). However, for the design of a comprehensive methodology which may support further ecosystem accounting efforts in the future, we aim to be as methodologically consistent and policy-relevant as possible. Note that recent developments in satellite data availability and processing have the potential to reduce these concerns regarding data shortages in the future (e.g. Shendryk, 2022 and SELINA D5.2 report on the use of satellite data to enhance EA).

Examples of how EDS and negative externalities with a corresponding ES can be reported are provided in Sections 8.1 (effects of peatland drainage on CO_2 emissions in The Netherlands) and 8.2 (effects of wildfires on forest-related ESs in Northern Portugal).

6.2 Ecosystem disservices and negative externalities that are not measurable with the same metrics as an ecosystem service

Many EDS and negative externalities, however, are not directly connected to an ES and therefore cannot be expressed in the same metrics as an ES. For example, the predation of surfers by sharks (an EDS) does not have a corresponding ES. In this case, a separate accounting table needs to be made. Given that EDS, in the same sense as ES, comprise a flow from the ecosystem to society, the recording of EDS is most logically connected to the ES account. EDS can be recorded both in biophysical and monetary terms. Table 1 presents an example of a biophysical EDS accounting table. In the case of wildlife trampling, ecosystem types represent the ecosystems that provide, e.g., elephant habitat, such as a forest or savannah.

Table 1 : Disservices accounting table – example.

	Unit	**Forest**	**Savannah**	**Wetland 3**	**Total**
Disservice account					
Disservice 1: crop losses due to wildlife trampling	1000 ton of paddy	150	350		500
Disservice 2: avian flu	1000 poultry specimens affected			15000	15000

It is worth noting that some EDS can, indirectly, be linked to a service. For instance, wildlife trampling of crops causes a reduction in crop provisioning ES. Care needs to be taken not to double count a disservice. Hence, crop losses due to wildlife trampling cannot be added (deducted from) to the actual crop production as recorded in the ES account since the account shows the net crop production inclusive of crop losses. A disservice, in this case, functions as an intermediate disservice – a negative effect from one ecosystem asset to another ecosystem asset. If the disservice and the service are both to be reported, a correction needs to be made, showing gross crop production (before wildlife losses), wildlife related crop losses, and net crop production (see example in Table 2).

In the case of Avian flu, recording is simpler since there is no corresponding entry in the ES accounts, and recording following Table 1 suffices (Note, however, that avian flu does function as an intermediate disservice in the production of poultry, and the net production of poultry is recorded in the SNA).

Table 2 : Intermediate disservices accounting table – example.

	Unit	Crop-land	Forest	Sava-nnah	Total
Gross service: Crop production	1000 ton of paddy	5600			5600
Intermediate disservice: crop losses due to wildlife trampling	1000 ton of paddy	-500	150	350	500
Net service: crop losses due to wildlife trampling	1000 ton of paddy	5100			5100

6.3 Negative externalities that involve a decrease in ecosystem condition but are not related to ecosystem disservices or ecosystem services

Negative externalities that involve a decrease in ecosystem condition but are not related to EDS or ES flows cannot be integrated in ES accounts. However, the change in ecosystem condition will be recorded in Ecosystem Condition accounts. Moreover, this category of negative externalities may potentially be integrated in the monetary valuation of ecosystem assets, as illustrated in the case of peatland drainage in the Netherlands (see Annex report on the Dutch test site).

6.4 Challenges in recording ecosystem disservices including negative externalities from ecosystem management in the accounts

6.4.1 Ecosystem type conversions

Negative externalities may also arise because of conversions from one ecosystem type to another, for instance the conversion of a natural forest to e.g. an agricultural field. In such cases, negative externalities include the EDS generated by the newly created agricultural field (e.g. N emissions), but also the loss of ES that the cleared forest was providing (e.g. carbon sequestration, water flow regulation, habitat for biodiversity, protection from wind and mass flows, water filtration, pest and disease regulation (see Annex 1). EDS generated by the newly created agricultural field can be treated as EDS that are the inverse of an ES (e.g. CO_2 emissions) or as EDS not connected to an ES (e.g. N emissions), as proposed in Section 6.1 and Section 6.2, respectively. However, the loss of services that were provided by the forest cannot be attributed directly to the newly created agricultural field. Nevertheless, the SEEA EA accounts will inform the user of the account of these effects: in the year after the land use conversion, the services provided by the forest will have been lost. Hence, ecosystem type conversion does not pose a problem in relation to recording EDS. Note that, where spatial data underpinning the SEEA EA accounts are made available to the user, the specific loss of ES due to ecosystem type conversion can be identified by comparing maps prior to and after the conversion. If there is a policy demand for such information, the loss of services due to ecosystem type conversion can be separately reported.

6.4.2 Recording intermediate ecosystem disservices

As is the case of ES, EDS can be intermediary. For example, the supply of final ES (e.g. crop provisioning) can be negatively affected by ecosystem processes (e.g. pest species finding a habitat in ecosystems nearby agricultural land) and such processes can be regarded as intermediate EDS. In this case, care should be taken to avoid double counting – the final service will already reflect losses due to intermediate EDS. When a disservice is a pollution source in an ecosystem type, another ecosystem type can be a sink of this pollutant where it is broken down. As an example, take N emissions to water. One could record a disservice flow from agricultural fields (N emissions). This would reduce the recreational services provided by downstream lakes, e.g. the number of days a lake is accessible for bathing (effects of algae blooms). The service provided by the lake is lower compared to the situation without the agricultural run-off, i.e. the disservice. If the disservice is included in the account as a final disservice, double counting of the negative effect may occur. Hence, in this case, the disservice needs to be recorded as an intermediate disservice. Hence if the disservice N runoff is included as a negative when accounting for the service provided by the agricultural field, in theory the services of the downstream lake would have to be increased by a similar amount (akin to the treatment of intermediate ES in SEEA EA). In practice, this is not likely to be feasible in most cases, since it would involve considerable effort and result in potentially unrealistically high, theoretical ES supply in the lake.

Hence, it is proposed to only include intermediate EDS in accounts in cases for which there is a specific policy need to bring out the interactions between ecosystems in the ecosystem accounts. Of course, not all EDS are intermediate, i.e. all EDS that directly affect people (e.g. by causing negative health effects) are final EDS. Greenhouse gas emissions can also be considered a final EDS, since their effect occurs in the long term and much of the consequences of current emissions are not fully reflected in current ES supply.

We would like to note that monetary valuation of intermediate EDS may often not be practically feasible, given that it involves analysing complex ecological processes that are not directly related to a benefit for people. Furthermore, monetary value of EDS cannot be simply added to (or deducted from) the value of services since this would lead to double counting. Hence, we postulate that the monetary valuation of intermediate EDS is not generally relevant for accounting purposes.

Importantly, flows of pollutants may lead to both final and intermediate EDS, and these may be difficult to disentangle. For instance, N emissions to air and water have both an effect on ecosystems and ES supply and an effect on human health (nitrate in drinking water, Particulate Matter precursor). Only recording the N flows that are final EDS may not be feasible in most cases since it is difficult to separate final and intermediate parts of the N flows. In this case, we would record all N emissions in the physical account (they are also very relevant for policy) and, in monetary accounts, only include the value of final EDS (human health cost) and exclude the value of intermediate EDS to avoid double counting.

6.4.3 Monetary valuation

We suggest that the following list of principles is adhered to when performing monetary valuation for EDS, reduction of ES and externalities in the context of accounting:

- Valuation of EDS and negative externalities should conform to the general guidance on monetary valuation in the SEEA EA (2021). This means the measurement of exchange values and not welfare values or other value concepts.
- Conceptually, prices for EDS and negative externalities could be framed in terms of markets to avoid negative impacts or reduction of output from services priced in (simulated) markets. This includes implicit prices that are revealed through transactions in related markets (e.g. hedonic pricing of reductions in air pollution or flood risk in residential and agricultural property markets).
- It may also be possible to obtain proxy prices for EDS and negative externalities using prices for the inverse and equivalent positive impact. For example, the value of a reduction in recreational activity due to degradation of a coral reef could be measured using information on the price of a recreational visit (e.g., a dive fee). This approach potentially ignores the implications of loss aversion and associated asymmetry of values for gains and losses of ES (although this is perhaps less relevant for exchange values than it is for welfare values).
- For EDS that are the direct inverse of an ES (e.g. carbon emissions and carbon sequestration), the same valuation methods should be applied as for the ecosystem service (e.g. damage costs measured by carbon credit prices).
- Value transfer methods are applicable to generate spatially variable value estimates at large geographic scales across multiple ecosystem service providing units. The potential for using value transfers for EDS may be limited, however, due to limited primary valuation research on the value of EDS. We note that most major classifications of ES (e.g. CICES, MA, TEEB, NESCS) and databases of valuations (e.g. ESVD) do not include EDS.
- Many EDS and negative externalities impact human health. It has long been established, however, that the national accounts do not place a direct value on health outcomes and instead the focus is placed on measuring the inputs to human health, e.g., outputs related to doctors and hospitals (cf. SEEA EA 2021, p225, 12.26 (UN et al., 2021)). The value of health impacts represents an important area of analysis that is broader than the ecosystem accounts. Regarding the economic valuation of health impacts resulting from EDS and negative externalities, there is an extensive and well-developed literature base on the valuation of (both positive and negative changes in) health endpoints than could be drawn on. To a large extent, the methods used to value health endpoints (e.g. loss of productivity, cost of treatment) are consistent with the general guidance on monetary valuation in the SEEA EA (UN et al., 2021).

7 Pilot accounts based on the test sites

This section presents a summary of the pilot accounts that have been compiled in three SELINA test sites to test the integration of EDS and NE in SEEA EA. Detailed reports of works carried out in each test site are presented in Annexes 2, 3 and 4.

7.1 Externalities from peatland management in the Netherlands

7.1.1. Background and scope

Around 8% of the Netherlands is covered by peatlands, of which most are located in the Provinces of Friesland, Noord-Holland, Zuid-Holland, Utrecht and Overijssel (Fig.2). The current use of peatlands results from a long history of peat extraction for energy (burning of dried peat) and the construction of polders, crop cultivation and eventually dairy farming. Linked to these uses, peatland drainage has been leading to oxidation and soil subsidence, with groundwater levels coming close to the field surface. Consequently nowadays, these peatlands can only be used as grasslands (van den Born et al., 2016).
Thus, the large majority (>90%) of Dutch peatlands are drained and used as meadows for dairy farming. This management of peatlands to optimize biomass provision has several negative consequences on other ES and biodiversity. The oxidation of peat leads to CO_2 emissions. In total, each year, peatlands emit some 6 to 7 million tons CO_2 (Lof et al., 2017; van der Net et al., 2023). The intensification of grassland management, through drainage and fertilisation, has negative effects on meadow birds (Kleijn et al., 2010). Water quality requirements of the Water Framework Directive are in general not met in peat areas, due to nitrogen and phosphorous losses resulting in eutrophication (Deltares, 2021). Drained peatlands have a reduced capacity to regulate water flows and are more vulnerable to floods and droughts (Kok & Angelova, 2020; van den Born et al., 2016). Last but not least, soil subsidence generates damages to infrastructures and buildings in rural areas (Kok & Angelova, 2020; van den Born et al., 2016).

In this study, we assessed the effect of peatland drainage on the provision of two ES, grass provision and meadow birds habitat provision, and two negative externalities, CO_2 emissions and damages to buildings. CO_2 emissions are negative externalities that are increases in the flow of EDS attributable to dairy farming on peatlands (see Section 5.3 of SELINA Deliverable D5.1). Conceptually, CO_2 emissions are the inverse of the CO_2 sequestration ES and can be integrated in the carbon sequestration supply and use table of ES accounts (see Section 6.1 of D5.1). Damages to buildings that result from soil subsidence are negative externalities that result from a decrease in ecosystem condition.

We did not include the effects of peatlands drainage on water quality and regulation of water flows due to a lack of suitable data to assess these effects in a spatially explicit manner with a national coverage. Moreover, we did not include emissions from energy use (pumps used for drainage) and from livestock. These emissions are part of the economy and already captured by the SEEA CF, hence not in the scope of SEEA EA in principle.

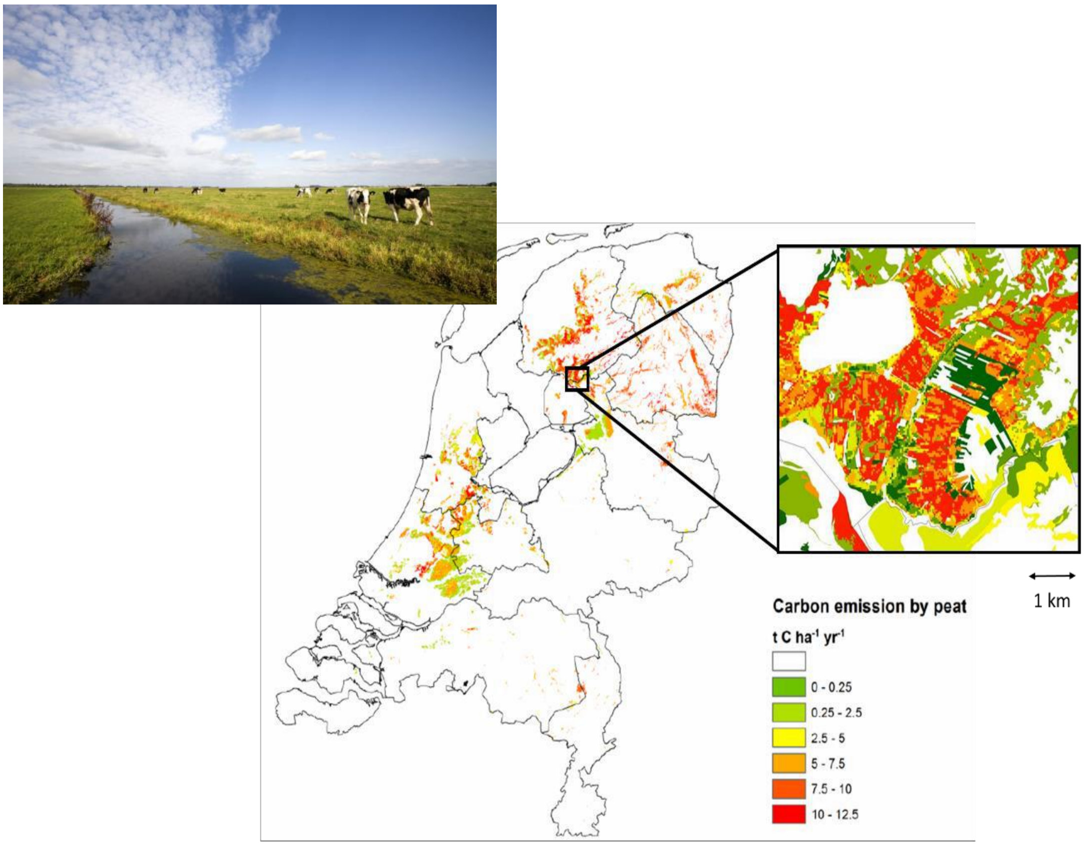

Figure 2 : Netherlands test site. Most Dutch peatlands are managed as grassland for dairy farming.

7.1.2. Methodology

Grass provision from peatlands was estimated using data from the NL SEEA EA on grazed biomass provisioning ES (biophysical and monetary accounts). With regards to meadow birds habitat provision, we used as indicator the Black-tailed Godwit (*Limosa limosa*) breeding density, predicted by a species distribution model (Sovon Vogelonderzoek Nederland, 2022), and a cost-based approach for non-use values for monetary valuation. For C emissions and sequestration, we used maps of C emissions from peat(y) soils and C sequestration in above- and below-ground biomass from the SEEA EA Carbon Account and followed the carbon monetary valuation approach of the SEEA EA accounts. For building damages, we directly used available monetary values, from a model that estimates building damage costs due to soil subsidence at municipal scale. We refer the reader to the NL test site report in the D5.5 Appendix for a detailed description of the methods.

7.1.3. Pilot accounts

Our results show how negative externalities associated to milk production on peat meadows can be included in SEEA EA through extended ES/EDS supply tables. Furthermore, we demonstrate how flows of ES and Negative externalities/EDS can be compared and integrated in ecosystem asset valuation (see report in Annex). In the case of dairy production on peat meadows, the net present value of ES, including grass provision, is much lower than the net present value of negative externalities/EDS, so that the ecosystem value of peat meadows is negative when taking into account dairy production negative externalities.

Table 3: Negative externalities of peatland drainage in the Netherlands. Physical SEEA EA ES supply table including disservices.

Ecosystem service	**Unit**	**Grassland:** Sown pastures and fields (on peat)
Grass provisioning	Ktons grass/year	1,779
Global climate regulation		
C sequestration (gross)	Ktons C/year	37.5
C emission (disservice)	Ktons C/year	1,062.5
Net C sequestration	Ktons C/year	- 1,025

Table 4: Negative externalities of peatland drainage in the Netherlands. Monetary SEEA EA ES supply table including disservices.

Ecosystem service	**Unit (1000 euro)**	**Grassland:** Sown pastures and fields (on peat)
Grass provisioning		125,090
Global climate regulation		
C sequestration (gross)		7,868.5
C emission (disservice)		222,067.5
Net C sequestration		- 214,199

Note that the damage to buildings is not included in the extended ES accounts, even though it was included in our case study. Damage to buildings are considered a negative externality that result from a change in ecosystem condition (decrease of soils' carrying capacity due to soil subsidence), but without any associated ES or EDS flow. Thus, integration in an extended ES supply table is not suitable. However, this negative externality was quantified in monetary terms and taken into account in ecosystem asset valuation (see above and Annex report on the Netherlands test site).

Note that peat meadows drainage may also lead to offsite peat oxidation and C emissions in neighbouring semi-natural peatlands (see Section 4.3 in the report in Annex). Following the above approach of a combined ES and EDS supply table, these offsite C emissions are attributed as EDS to the semi-natural peatlands ET. However, these offsite C emissions are negative externalities of the peat meadows management for milk production.

7.2 Negative externalities from forest fires in Portugal

7.2.1. Background and scope

Forests and seminatural areas cover around 59% of the Northern Portugal - NUTS-II EU administrative region (Figure 3) and provide a wide range of provisioning (e.g., timber, firewood, wild mushrooms), regulating (e.g., carbon storage, soil retention, water regulation), and cultural (e.g., nature-based tourism) ES (Ribeiro et al., 2011). Nevertheless, the high fire activity in Northern Portugal poses a threat to the supply of these services. The 2017 fire season was the most devastating nationwide and among the worst in Northern Portugal, resulting in ca. 90 thousand hectares of burned land (INE, 2024).

The impacts of forest fires in Portugal can be seen as negative externalities driven by poor practices and socioeconomic changes in forestry and agricultural sectors, such as 1) neglected forest management (e.g., lack of fuel management); 2) intensified forest management (e.g., monoculture plantations), 3) inappropriate afforestation (e.g., highly flammable species in former croplands/pasturelands); 4) farmland abandonment (Mateus and Fernandes, 2014).

Forest fires can reduce pre-existing ES by promoting soil erosion, reducing biomass supply, altering carbon stocks, or increasing losses in nature-based recreation potential (Sil et al., 2019). Social costs of forest fires in Portugal average ca. €370 million/year, of which ca. 70% are related to ecological damages (Mateus and Fernandes, 2014; Mendes et al., 2021). In this context, quantifying and valuing the negative impacts of forest fires, particularly on ES supply, is crucial to developing effective policies for fire risk management, natural resource management, and incorporating natural capital into decision-making. This test site aims to demonstrate how forest fires negatively impacted a set of key ES in Northern Portugal (**Fehler! Verweisquelle konnte nicht gefunden werden.**) and how these negative externalities can be integrated into ecosystem accounts.

Table 5: ES and targeted negative externalities (reduced ES) derived from forest fires in the Northern Portugal test site.

Ecosystem Service			Negative externalities due to forest fires
Provisioning	Fibres and other materials from cultivated plants	Timber provision: the volume of woody biomass in forests (i.e., potentially harvestable).	Reduction of timber provision: due to damages in the volume of woody biomass in areas burned at high/very high fire severity.
		Cork provision: the volume of cork in cork oak forests (i.e., potentially harvestable).	Reduction of cork provision: due to damages in cork oak forests in areas burned at high/very high fire severity.
Regulating & Maintenance	Regulation of chemical composition of atmosphere and ocean	Carbon storage: the amount of carbon stored by above- and belowground biomass and dead organic matter in forests and shrublands.	Reduction of carbon storage: due to losses in carbon stocks in burned areas according to different levels of fire severity.
	Control of erosion rates	Soil retention: the amount of soil retained by forests and shrublands.	Reduction of soil retention: due to vegetation losses in burned areas according to different levels of fire severity.

Cultural	Physical and experiential interactions with the natural environment	Nature-based tourism: recreation potential in forested landscapes.	Reduction of nature-based tourism: loss in recreation potential according to different levels of fire severity.

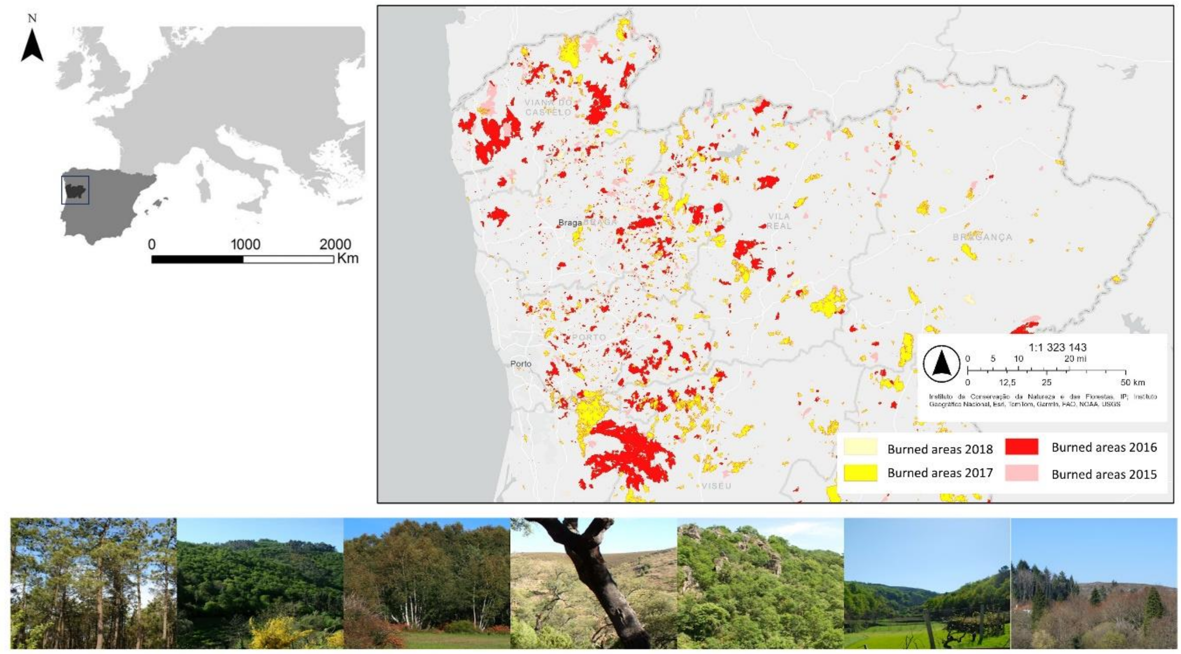

Figure 3 : Northern Portugal test site. Left panel: Northern Portugal (NUTS-II EU) test site. Right panel: area burned in Northern Portugal between 2015 and 2018. Bottom panel: forested landscapes in Northern Portugal.

7.2.2. Methodology

Negative externalities, i.e., the reduction of pre-existing ecosystem service supply due to forest fires, were evaluated in both biophysical and monetary units by developing an integrated modelling framework coupling spatial explicit data on land cover and forest fires with public statistics, modelling tools, and relevant scientific literature. The application of the modelling framework focused on forest and shrubland areas impacted by forest fires that occurred in 2017 in Northern Portugal. First, ecosystem service supply was estimated in the pre-fire environment, i.e., within the area corresponding to fire perimeters prior to fire occurrence. Then, the potential impacts of forest fires on ES were analysed based on the extent and severity of the burned area to estimate the ecosystem service supply in the post-fire environment, i.e., within the area corresponding to fire perimeters after fire occurrence. Finally, the reduction in ecosystem service supply due to forest fires was estimated as the difference between ES estimates in the pre-fire and post-fire environment (see the PT test site report in Appendix for a detailed description of the methods).

7.2.3. Results

The analytical framework developed at the Northern Portugal test site proved to be comprehensive and straightforward for assessing negative externalities resulting from the effects of wildfires on ES, in both biophysical and monetary terms, thus allowing integration into ecosystem accounting frameworks (Table 6 and Table 7; see the test site report in Annex 3 for full results). Overall, our analysis indicated that 65% of timber production was lost due to forest fires, with monetary losses ranging from 21.3 to 69.8 million Euros. Regarding cork production, losses caused by forest fires corresponded to 52% of cork, with monetary losses ranging from 8.4 to 51.3 million Euros. Carbon stored in forest and shrubland areas was reduced by 17% due to forest fires, equating to social costs ranging from 5.0 to 65.9 million Euros and monetary losses in carbon credits ranging from 41.5 to 64.8 million Euros. Regarding the impact of forest fires on soil retention by forests and shrublands, there was a reduction of 9%, with soil replacement costs estimated at 1.8 million Euros, and soil erosion mitigation measures ranging from 124.4 to 653.6 million Euros. Recreation opportunity areas were reduced by 94% due to forest fires, with monetary losses estimated at 2.82 million Euros.

Table 6: Negative externalities of forest fires in Northern Portugal. Biophysical SEEA EA ES supply table including negative externalities.

Ecosystem service	**Unit**	**Forest**	**Shrubland**
Timber provision (gross)	m^3	2,669,691.91	N.A.
Loss of timber provision (due to forest fires)	m^3	1,732,488.10	N.A.
Timber provision (net)	m^3	937,203.81	N.A.
Cork provision (gross)	kg	168,580.6	N.A.
Loss of cork provision (due to forest fires)	kg	88,032.2	N.A.
Cork provision (net)	kg	80,548.5	N.A.
Carbon stored (gross)	Mg	1,320,847.2	545,507.8
Loss of carbon stored (due to forest fires)	Mg	182,442.5	126,220.2
Carbon stored (net)	Mg	1,138,404.7	419,287.6
Soil mass retained (gross)	Mg	2,549,174.7	1,199,611.6
Loss of soil mass retained (due to forest fires)	Mg	226,580.6	106,626.2
Soil retained (net)	Mg	2,322,594.1	1,092,985.5
Recreation opportunity (gross)	ha	17,753.5	23,265.5
Loss of recreation opportunity (due to forest fires)	ha	16,203.8	22,371.6
Recreation opportunity (net)	ha	1,549.7	893.9

N.A.: Not applicable. Note that estimates shown for timber provision are based on intensive land-use plantations (maritime pine forests and eucalyptus forests) and deciduous temperate forests (oak forests). Estimates shown for cork provision are based on temperate pyric sclerophyll forests and woodlands (cork oak forests). Estimates shown for carbon storage, soil retention, and recreational opportunity include temperate deciduous forests (e.g., oak forests, oak woodlands), temperate pyritic sclerophyll forests (e.g., holm oak forests, holm oak forests), temperate pyritic wet forests (e.g., invasive species), intensive land-use plantations (e.g., maritime pine forests, stone pine forests, eucalyptus forests), and seasonally dry temperate heath and shrublands.

Table 7: Negative externalities of forest fires in Northern Portugal. Monetary SEEA EA ES supply table including negative externalities.

Ecosystem service	Unit	Forest	Shrubland
Timber provision (gross)	Mln €	66.0	N.A.
Loss of timber provision (due to forest fires)	Mln €	42.7	N.A.
Timber provision (net)	Mln €	23.3	N.A.
Cork provision (gross)	Mln €	61.7	N.A.
Loss of cork provision (due to forest fires)	Mln €	32.2	N.A.
Cork provision (net)	Mln €	29.5	N.A.
Carbon stored (gross)	Mln €	21.3	8.8
Loss of carbon stored (due to forest fires)	Mln €	2.9	2.0
Carbon stored (net)	Mln €	18.4	6.8
Soil mass retained (gross)	Mln €	13.6	6.4
Loss of soil mass retained (due to forest fires)	Mln €	1.2	0.6
Soil retained (net)	Mln €	12.4	5.8
Recreation opportunity (gross)	Mln €	1.38	1.57
Loss of Recreation opportunity (due to forest fires)	Mln €	1.29	1.54
Recreation opportunity (net)	Mln €	0.09	0.03

N.A.: Not applicable. Note that monetary values shown for the timber and cork provision are averages. Monetary values shown for carbon storage are based on estimates for the Social Cost of Carbon in Europe reported by Nordhaus (2017). Monetary values shown for soil retention are based on replacement cost reported by Marta-Pedroso et al. (2007).

7.3 Ecosystem disservices and negative externalities resulting from wildfires and intensive agriculture development in Peloponnese, Greece

7.3.1. Background and scope

Greece is one of the most visited tourist destinations in the EU, especially in the summer months, with the vast majority of tourism and related infrastructure concentrated along the country's coastline where nature is the main attraction. Simultaneously, Greece is one of the most biodiverse countries in the EU, with 27.3 % of its terrestrial and 19.6 % of its marine area, respectively, included in the Natura 2000 protected areas Network. The objective of the SELINA test site (Figure 4), is to map and assess how the EDS (natural forest fires) and externalities of wildfires (fires triggered or began by human action) and intensive agriculture are affecting nature-based recreation/tourism and how these EDS and negative externalities can be quantified and integrated in the SEEA EA framework. More precisely, we identified and assessed these EDS and negative externalities using remote sensing methods, based on EO data, with special focus in areas belonging to National parks and Natura 2000 protected areas. Moreover, specific objectives deal with the development of relevant, standardised indicators for NCA. The impact of wildfires, including the megafires of 2007, and subsequent fire events that occurred in the same areas during the last twenty years, were evaluated combined with the potential of ecosystem recovery period.

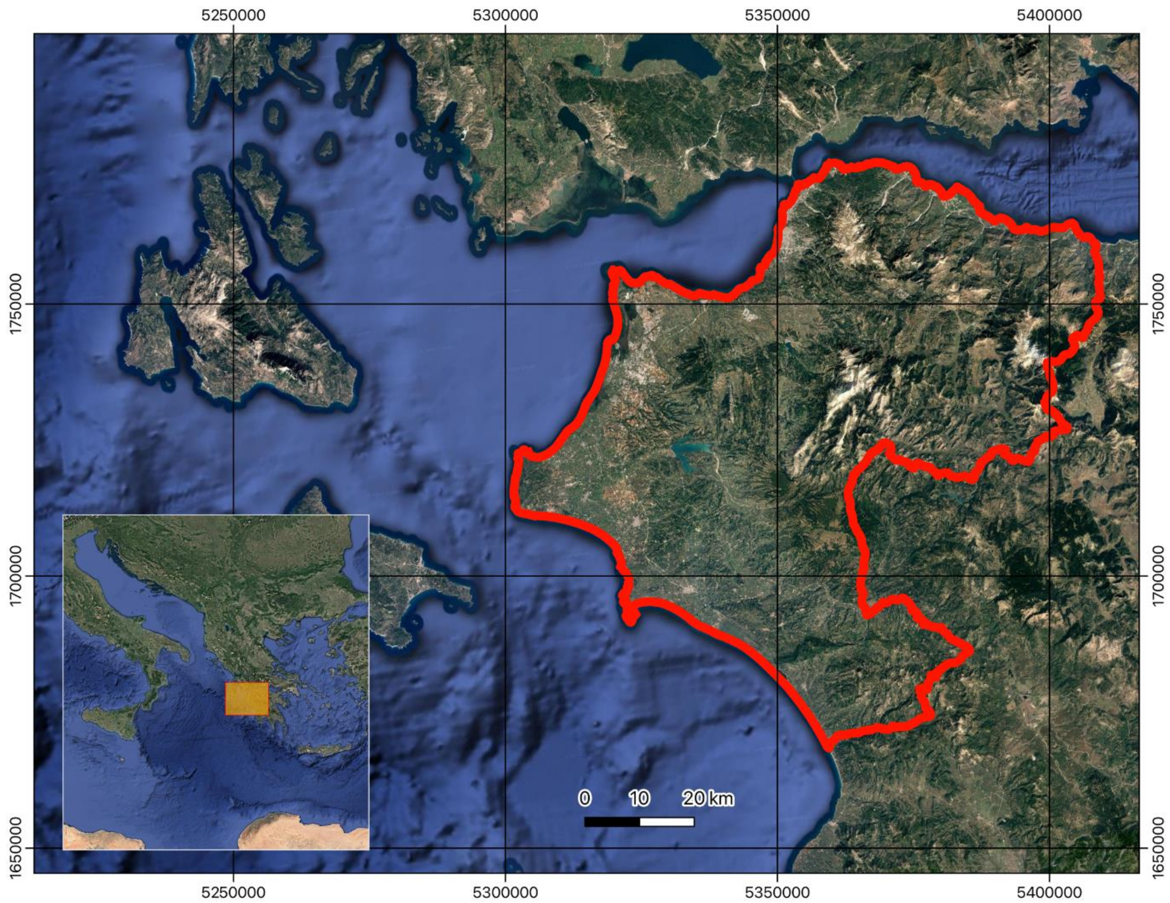

Figure 4. Test Site of Peloponnese, located at the western part of Greece. The red line includes an updated (more extensive) area in Peloponnese, in order to better capture the targeted externalities.

In **Fehler! Verweisquelle konnte nicht gefunden werden.Fehler! Verweisquelle konnte nicht gefunden werden.**, a characteristic part of the study area is presented depicting how wildfires and agricultural activities and infrastructure impact ecosystems, landscapes and land uses, from 1945 to present day.

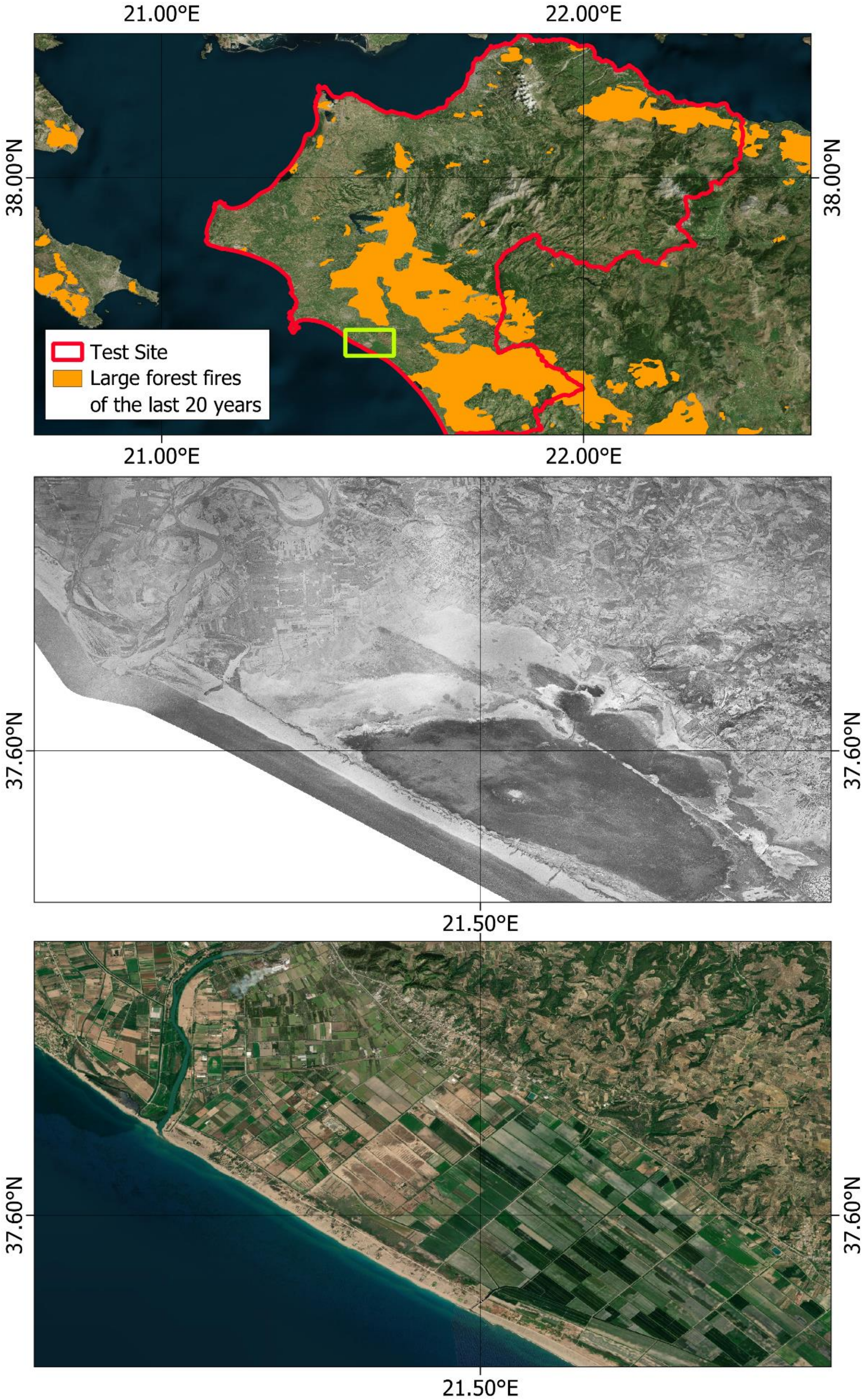

Figure 5: Peloponnese test site. Upper panel: study area and forest fires of the last 20 years. Middle and lower panel: Characteristic example of agricultural activities affecting ecosystems and their services. In the lower panel (current status), the severe modification of the river route and its riparian ecosystems, compared to the image in the middle panel (year 1945), is depicted, alongside the loss of the river deltaic system. Former wetlands and riparian areas are now covered by intensive agricultural activities (lower panel).

7.3.2. Methodology

In the Test Site of Peloponnese, we collected all available data from open-source platforms, regional, and local agencies in order to identify the externalities present in the region and select the appropriate data to conduct an accounting process, based on one or more externalities. Forest fires have been included in their spatial and temporal aspect, and intensive agriculture as an existing externality that decreases nature recreation potential. The analysis and mapping procedure was made in the Google Earth Engine (GEE) platform, using the relevant datasets (GEE assets) prepared by the UPATRAS team. To assess the forest fire impact on the Test Site's ecosystems, we used the available EFFIS layer, including all registered forest fires occurred in the area since 2007, as a vector file, that includes the date of each fire event (per fire polygon). This allows to quantify the fire area, as well as the ecosystem types burnt and the time of the fire incident, in order to calculate the potential restoration response time of the different ecosystem types. We have considered that only non-intensive and traditional agricultural practice contributes to recreation potential (e.g. for agrotourism), and only those categories are integrated in the model. Intensive agriculture classes are considered as externalities.

Monetary valuation of nature recreation was based on the Methodology for estimating the value of forest land in Greece (Albania et al. 2018), officially adopted by the Greek State (see Greek Test Site Annex), according to the formula:

$Vr = Area_r * MARr$,

Where,

Vr is the annual recreational value in €,

$Area_r$ is the area of the forest in which recreation is exercised in ha, and

MARr is the annual recreational value per hectare in €/ha.

7.3.3. Results

The results of this study reveal that by using open-source datasets and platforms, we can integrate externalities such as forest fires in the SEEA EA framework (see Table 8). We have developed an index (R-Index) that corresponds to the recreation in nature potential, that integrates in the 0.00-1.00 scale all parameters related to nature recreation attractiveness and potential of a site including the parameter of externalities such as forest fires. This index can be further explored to include more externalities that may be relevant to assess this cultural ecosystem service (e.g. recreation area loss due to severe flooding, coastal erosion etc.).

Table 8: Negative externalities related to forest fires and intensive agriculture in Peloponnese, Greece. Biophysical (recreation potential index – R-Index) and monetary SEEA EA ES supply table including negative externalities (ΔR-Index).

Year / Ecosystem types (level 2)	Area (ha)	R-Index	MARr	Δ-R-Index	Recreation value	Δ-Recreation value
2021						
Broad-leaved forest	9425.42	0.953	94.3	0.002	846,623.55 €	-
Coniferous forest	37159.42	0.936	94.3	-0.001	3,280,050.75 €	-
Mixed forest	14517.94	0.836	94.3	-0.09	1,144,455.93 €	-
Transitional woodland-shrub	82530.45	0.738	75.4	-0.01	4,592,394.80 €	-
2022						
Broad-leaved forest	9425.42	0.955	94.3	0.00	848,690.84 €	2,067.30 €
Coniferous forest	37159.42	0.935	94.3	0.00	3,275,859.09 €	-4,191.66 €
Mixed forest	14517.94	0.837	94.3	0.002	1,146,550.52 €	2,094.60 €
Transitional woodland-shrub	82530.45	0.749	75.4	0.011	4,660,241.19 €	67,846.40 €
2023						
Broad-leaved forest	9425.42	0.956	94.3	0.002	850,113.76 €	1,422.92 €
Coniferous forest	37159.42	0.936	94.3	0.001	3,280,583.58 €	4,724.49 €
Mixed forest	14517.94	0.844	94.3	0.01	1,155,909.73 €	9,359.21 €
Transitional woodland-shrub	82530.45	0.750	75.4	0.00	4,664,089.40 €	3,848.20 €

8 Conclusion

In this Deliverable D5.5 report, we reviewed the concepts of externalities and EDS within the conceptual framing of ecosystem accounting, and proposed the integration of four categories of EDS and negative externalities to the SEEA EA: (1) EDS that are the negative effects from ecosystems to human well-being, not directly related to human activities; (2) Negative externalities that are an increase in the flow of EDS attributable to specific human activities; (3) Negative externalities that are reductions in the supply of ES attributable to specific human activities, and (4) Negative externalities that involve a decrease in ecosystem condition attributable to specific human activities

Furthermore, we presented methodological approaches to integrate flows of negative externalities and EDS in ecosystem accounts. We distinguished EDS that are measured with the same metrics as ESs (e.g. CO_2 emissions) from EDS that are not directly connected to an ES (e.g. predation of surfers by sharks). While the former can be recorded alongside ESs flows in ESs accounting tables, the later require a distinct EDS accounting table. We also addressed: (1) The treatment of negative externalities and EDS arising from ecosystem type conversions: (2) How to record intermediate EDS, i.e. a negative effect from one ecosystem asset to another ecosystem asset, and avoid double-counting, and (3) the monetary valuation of negative externalities and EDS in the context of SEEA EA.

The methodological robustness and real world applicability of approaches presented in this report were tested in a number of SELINA test sites and Demonstration Projects (DPs). These tests showed that EDS and externalities are strongly linked, yet they are highly dependent upon context. Different ecosystems in different countries will have very different EDS that could be connected to the accounts. Furthermore, accounting for EDS and negative externalities typically requires ecological / spatial modelling in order to understand how environmental change is affecting ES supply, or leading to the generation of EDS. It may be a challenge to find the data and capacity to apply accounting for EDS and negative externalities at national scales – and in any case further piloting is needed to test how to account for EDS and negative externalities in different contexts compared to the ones we have included in the SELINA project and in this report. The work will also benefit from including the outcomes of the DP in La Reunion, once they become available.

Hence, it is too early to propose rolling out accounting for EDS and negative externalities in the ecosystem accounting community. More discussions on the potential set-up of the approach to account for EDS and negative externalities and more cases studies in different socio-economic contexts are needed to further develop a broadly acceptable approach to account for EDS and negative externalities. In this context, an important consideration is that EDS and negative externalities as connecting elements between the SEEA Central Framework (that records negative externalities, but does not connect negative externalities to ecosystem types or location) and the spatially explicit SEEA Ecosystem Accounts. Nevertheless, there is a rationale for further work in this field: better monitoring and accounting for EDS and negative externalities can support environmental management by filling important data gaps that are not currently covered in the accounts.

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Annex 1: Health-related perspectives on ecosystem disservices and negative externalities

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1. Introduction

In presenting definitions of negative externalities and EDS, this report has recognised that as yet there is no general scientific consensus on a definition, typology or conceptual framework for addressing or assessing EDS. A related difficulty is that arguments and perspectives on EDS tend to come from a relatively narrow scientific and cultural perspective, typically from “ecosystem services” science and adjacent disciplines. For example, discussions of EDS frequently focus on the negative impacts which nature, or interaction with nature, can have on human health, and yet perspectives coming directly from epidemiology or other health sciences or from public health planning or policy are rarely included in these discussions. This means that crucial understanding of the mechanisms by which such health threats arise and the degree to which they may be deemed risks at all is overlooked.

Some authors have suggested that various ecosystem functions, without human influence, can pose threats to human health (e.g. Lytimaki and Sipila, 2009; Dunn, 2010), without giving appropriate consideration to the often highly complex interactions between people and ecosystems which might not only determine the actual existence of such threats, but also the spatial and temporal occurrences and flows of related EDS. This issue has a correlate in the sphere of health policy, where assumptions of a simplistic habitat-pathogen-disease paradigm have sometimes led to poorly informed interventions involving destruction of natural habitats without fully considering the negative consequences of such actions on ES which support human well-being. This underscores the need for transdisciplinary approaches to the identification, assessment and management of EDS to ensure that trade-offs and externalities are appropriately considered.

Here, we discuss the issue of EDS from a human health perspective in order to highlight some of the gaps in current conceptual frameworks and discussions on EDS in the literature, whilst also arguing that accounting for EDS and negative externalities - regardless of how they are framed - as part of a comprehensive system of environmental economic accounts is important in order to facilitate mainstreaming of biodiversity and ES into the health sector and to more completely inform decision making on conservation and sustainable use of the natural environment.

2. The importance of linking biodiversity, ecosystem services and health

There are several reasons why the health sector represents a key area for mainstreaming biodiversity, ES and natural capital concepts. Firstly, the varied and intricate relationships between nature and health are increasingly well understood and have been explored in detail in scientific literature (see for example WHO-CBD 2015), and have become of increasing concern to governments and citizens throughout the EU and worldwide. Recent experience of the Covid-19 pandemic has provided clear examples of the connections between the health of the environment, human health and the health of other species (IPBES, 2020). Second, investments in health and healthcare account for a significant amount of public expenditure at regional, national and local levels, with an EU average investment equivalent to approximately 11% of GDP and 20% of gross national budgets, as well as being a key area of household expenditure for individuals and families. Furthermore, the health sector directly or indirectly encompasses a broad diversity of scientific disciplines, policy sectors and areas of economic activity; as such, the health sector may be seen as an important instrument of economic policy (Jagrič et al., 2020). Third, health is recognised as a significant component of well-being, influencing individual and societal metrics on quality of life, lived experience, personal development and social interaction (e.g. Ruggeri et al., 2020) and therefore should factor explicitly in assessments of ES/nature’s contribution to people.

While the significance of biodiversity and ecosystems to health is well established, it is important to note also that the health sector itself can, through various policies, programmes and practices, have negative impacts on biodiversity, ecosystems and ES (e.g. Boxall and Kretsch, 2015). Where these impacts threaten the sustainability of ES which contribute positively to health or other elements of well-being, or potentially increase negative externalities, it is important that they be recognised, understood, and appropriately accounted for in the development, assessment and review of related policies.

Various barriers to mainstreaming in the health sector have been discussed in the literature (e.g. Kretsch, 2016; Campbell-Lendrum, 2005). Historically, one difficulty has been a perception within the health sector, or in government agencies in various countries, that biodiversity and ecosystems are a source of significant health threats and therefore options for the management of those risks should include ecosystem degradation or culling of wild species. Examples include policies on the destruction of wetlands in many countries as a means of combating malaria (Keiser et al., 2005) and calls for the widespread culling of wild birds and removal of their preferred habitats as a means of preventing the spread of avian influenza (Cromie et al, 2011; Cook and Karesh, 2012). Although the term EDS is not necessarily used in such contexts, these concerns clearly correlate with the idea that biodiversity and ecosystems can pose inherent threats to human health which should be accounted for when developing strategies for management of the natural environment. If such strategies in turn pose a threat to the sustainability of ES associated with the targeted biodiversity and ecosystems, then the ultimate impacts on human health and well-being may be negative – in addition to the fact that these actions may often be counter-productive (Miguel et al., 2020).

Although the relationships between elements of biodiversity or certain ecosystems and health are in some cases clearly identified, the exact mechanisms and pathways through which ecosystems influence health outcomes are sometimes poorly understood, or are highly case specific, depending upon, for example, climate, geography, and cultural perspectives and behaviours (IPBES, 2020; Clark et al., 2014). In order to give an accurate economic account of the relationships between ecosystems and health, and therefore to produce accounts that can better inform decision making, it is important that these pathways (linking ecosystem structure and function and health-related ES, EDS, benefits, costs or values) are carefully examined. This is particularly important for complex systems where a full understanding of which aspects of biodiversity or ecosystem functioning can or should be managed to address health issues requires careful consideration of the interaction between ecosystem health and the health of animals, plants and humans, as well as of the past, present or future role of human activity and behaviour in driving those health issues (Ostfeld and Keesing, 2017).

This requires a nuanced approach to EDS and a conceptual framework for EDS accounting that can facilitate a better assessment of health risks, pathways, and responses. Such a framework has not yet been established for either ES or EDS related to health, and though it is beyond the scope of this short report, it will be addressed in other tasks within SELINA. For the purposes of this Deliverable report, some examples of the complexities involved are provided in the following section.

3. Health threats from nature: ecosystem, disservices, reduced services, or negative externalities? Or does it matter?

The notion of EDS, as with ES, involves some degree of human agency identifiable at one or more stages of the ES (or EDS) cascade – e.g. anthropogenic impacts on ecosystem structures or functions, specific behaviours resulting in contact with biodiversity, societal or personal perceptions on benefits or disbenefits, or specific views on impacts and values. Whether perceived health risks associated with ecosystems are classified as EDS or externalities, or as a result of reductions in ES stocks or flows, or as primarily driven by ecological processes or by human influence on those processes, is of material relevance to the development of interventions intended to effectively alleviate those risks. Whilst this classification is not necessarily of relevance to the process of accounting – i.e., knowledge of the root causes of an emerging disease outbreak does not necessarily inform an assessment of the immediate human cost of that outbreak - the outputs of accounting efforts can help to prioritise and promote more detailed investigations into the scale and determinants of negative human-nature interactions, and provide an economic argument for enhanced transdisciplinary approaches to ecosystem assessment and management.

Recent research into emerging infectious diseases have highlighted some of the difficulties involved in attributing health risks to ecosystem structures, processes or functions. The case of Hendra virus disease in Australia, which causes acute respiratory infection in humans and horses, is a useful example. Although virological studies have since indicated that Hendra virus has circulated in Australian flying foxes (fruit bats) since before the arrival of Europeans, it only came to public health attention when it caused fatal disease outbreaks in horses and the humans who interacted with them in 1994 in eastern Australia, with bats being identified as the reservoir by 2000 (Halpin et al., 2000). Outbreaks were not recorded between 1996 and 2002, however from 2003 bat behaviour and the number of outbreaks changed rapidly. The proximate causes of the outbreaks relate to increased occurrence of bats in urban and agricultural lands; however, the primary root cause was a change in the roosting and feeding behaviour of fruit bats in response to habitat loss (by 1996, the bats' primary forest habitat had been cleared by over 70% of its pre-colonial extent) exacerbated by climate change associated with El Nino events which impact on the flowering and fruiting periods of the bats' preferred food trees. Bats which had historically been resident in their home forest habitats adapted to food shortages by becoming nomadic, moving closer to human settlements and farms to avail of other sources of food, placing them in contact with horses, which passed infections on to humans and pets. A comprehensive picture of how the emergence of Hendra was the result of largely anthropogenic changes in bat ecology was not ascertained until 2023 (Eby et al., 2023).

Similar patterns have been identified in the emergence of other infectious diseases over the past several decades, with the risk of several disease outbreaks (in humans, livestock or wildlife) relating to changes in the ecology of pathogens or vectors driven by human influence on landscapes and biodiversity. In the case of Hendra, prior to recent insights into the social-ecological dynamics involved the disease was largely framed as an issue of human-livestock-wildlife conflict, with bats being increasingly regarded by the public and health authorities as pests – a source of EDS. The current understanding, based on a more integrated scientific approach, frames the issue largely as an externality from agricultural expansion and deforestation, with significant implications for future disease management, outbreak prediction, and habitat management. In many cases, the precise drivers, proximate causes and root causes of disease emergence are not identified until many years after the disease is first reported.

The existence of pathogens and parasites, and by extension any potential they may have to cause disease in humans or in other species of economic importance, should not alone be sufficient to class an ecosystem or species as being a source of EDS, since the positive and often essential role of pests and pathogens in ecosystem functioning must also be taken into account. Indeed, from an ecosystem management perspective, since they shape host population dynamics, alter interspecific competition, influence energy flow and appear to be important in the maintenance of biodiversity, conserving populations of pathogens and other species potentially harmful to humans may be essential to the sustainability of ES as well as the reduction of other EDS (Fischhoff et al, 2020; Hatcher et al., 2012; Delaux and Schornack, 2021) – again dependent upon context. Whilst it may arguably be in the best interests of human well-being to eradicate certain pathogens – smallpox and the malaria parasite being cases in point – such valuations taken in the absence of a comprehensive account of likely ES trade-offs and potential externalities will be incomplete at best, leading to uncertainties around the sustainability of related interventions.
Similar issues arise across several other proposed classes of EDS presented in the literature. For example: risks of attack by wild animals may be an EDS or an externality due to human encroachment on habitats or loss of predators' food resources (IUCN, 2023); the risks of antibiotic resistance, proposed as an EDS by some authors, are largely driven by pollution and human over-use of antimicrobial compounds (Boxall and Kretsch, 2015); and the increasing occurrence of harmful algal blooms in many aquatic and marine ecosystems is associated with pollution and anthropogenic climate change (Gilbert, 2020).

The key take-away here is that while economic accounting of negative aspects of biodiversity and ecosystems is essential in order to build a comprehensive and balanced picture of ES/nature's contributions to people, the utility of such assessments can be severely limited by narrowly-defined or silo-based determinations of EDS. From a public health perspective, identifying a particular ecosystem-related health threat as being a disservice or an externality may be less important in the face of urgent health risks than unpacking the complex drivers, pathways, responses and trade-offs involved. Nevertheless, ensuring that the results of ecosystem economic accounting speak substantively to the data needs of the health sector whilst avoiding confusion and also providing appropriate direction for policy and practical responses – for biodiversity conservation, ecosystem service sustainability, and health - requires that the conceptual frameworks used for EDS and externalities are based on careful consideration of how such risks are framed and communicated.

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D5.5 Annex 2: Negative externalities from peatland management in the Netherlands

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1. Preface

This report is an Annex of SELINA Deliverable D5.5 Specifying and testing how externalities and EDS can be included in ecosystem accounts. It presents in detail the analyses that were carried out in the test site "Externalities from peatland management in the Netherlands" for the purpose of Task 5.1. The content of this report is developed based on the study carried out by Ugne Grikinyte (2024) during her Master Thesis. We refer the reader to Grikinyte (2024) for details about the methodology and results.

2. Introduction

2.1. Site description

Around 8% of the Netherlands is covered by peatlands, of which most are located in the Provinces of Friesland, Noord-Holland, Zuid-Holland, Utrecht and Overijssel (Fig.1). The current use of peatlands results from a long history of peat extraction for energy (burning of dried peat) and the construction of polders, crop cultivation and eventually dairy farming. Linked to these uses, peatlands drainage has been leading to oxidation and soil subsidence, with groundwater levels coming close to the field surface. Consequently nowadays, these peatlands can only be used as grasslands (van den Born et al., 2016).

Thus, the large majority (>90%) of Dutch peatlands are drained and used as meadows for dairy farming (Fig. 2). This management of peatlands to optimize biomass provision has several negative consequences on other ES and biodiversity. The oxidation of peat leads to CO_2 emissions. In total, each year, these emit some 6 to 7 million tons CO_2 (Lof et al., 2017; van der Net et al., 2023). The intensification of grassland management, through drainage and fertilisation, has negative effects for instance on meadow birds (Kleijn et al., 2010). Water quality requirements of the Water Framework Directive are in general not met in peat areas, due to nitrogen and phosphorous losses resulting in eutrophication (Deltares, 2021). Drained peatlands have a reduced capacity to regulate water flows and are more vulnerable to floods and droughts (Kok & Angelova, 2020; van den Born et al., 2016). Last but not least, soil subsidence generates damages to infrastructures and buildings in rural areas (Kok & Angelova, 2020; van den Born et al., 2016).

2.2. Negative externalities and ecosystem disservices

In this study, we assessed the effect of peatland drainage on the supply of two ES, grass provision and meadow birds habitat provision, and two negative externalities, CO_2 emissions and damages to buildings. CO_2 emissions are negative externalities that are increases in the flow of EDS attributable to dairy farming on peatlands (see Section 5.3 in D5.5). Conceptually, CO_2 emissions are the inverse of the CO_2 sequestration ES and can be integrated in the carbon sequestration supply and use table of ES accounts (see Section 6.1 in D5.5). Damages to buildings that result from soil subsidence are negative externalities that result from a decrease in ecosystem condition (see Section 5.3 in D5.5).

We did not include the effects of peatlands drainage on water quality and regulation of water flows due to a lack of suitable data to assess these effects in a spatially explicit manner with a national coverage. Moreover, we did not include emissions from energy use (pumps used for drainage) and from livestock. These emissions are part of the economy and already captured by the SEEA CF, hence not in the scope of SEEA EA in principle.

3. Data and methods

We compared the supply of the selected ESs and negative externalities/EDS in peat meadows with varying drainage levels and in semi-natural (not drained) peatlands.

The NL SEEA EA extent accounts Ecosystem Type (ET) raster map with a resolution of 10 meter was used as a base map. All other input data maps (see Tables 1, 2, 3, 4 and 5) have been resampled to the same resolution of 10 m to perform the processing steps described below.

The first step of the analysis was to create a map of peat meadows and semi-natural peatlands, by combining the ET map in the NL SEEA EA extent accounts with a soil type map. Areas of pasture ET and areas of bogs/fens ET located on peat(y) soils were selected to generate a map of peat meadows and semi-natural peatlands (see Fig. 1).

Next, we used a map of the lowest mean groundwater level (LMGL) (https://www.klimaateffectatlas.nl) as proxy of drainage depth and combined this MLGL map with the map of peat meadows to estimate their drainage level. We classified peat meadows in 5 drainage level classes using quintiles: Highest (> 1.16 m. below the surface), High (0.99 – 1.16 m.), Middle (0.86 – 0.99 m.), Low (0.72 – 0.86 m.) and Lowest (< 0.72 m.) (see Fig. 2).

Table 1: Data inputs used for mapping peat meadows and drainage classes.

Data inputs	Format	Units	Resolution (m)	Date	Source
Ecosystem types map	Raster	Categorical map	10x10	2020	NL SEEA EA (Statistics Netherlands, 2021)
Soil types and subtypes map	Vector	Categorical map	n/a	2014	WUR (de Vries et al., 2014)
lowest mean groundwater level (LMGL) map	Raster	Meters below surface	250x250	2019	Klimaateffectatlas.nl

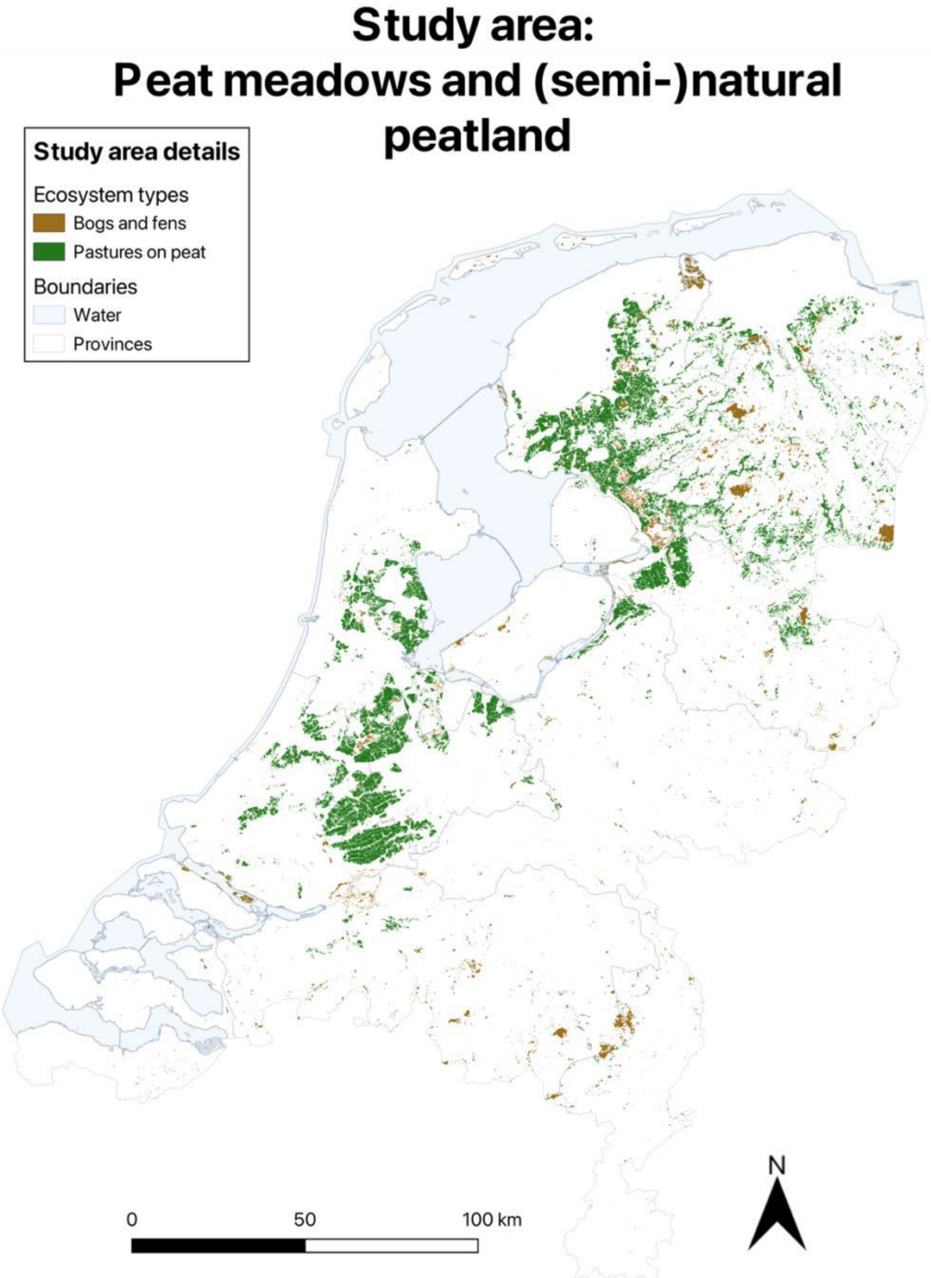


Fig. 1: map of peat meadows and semi-natural peatlands in the Netherlands. Source: Grikinyte (2024).

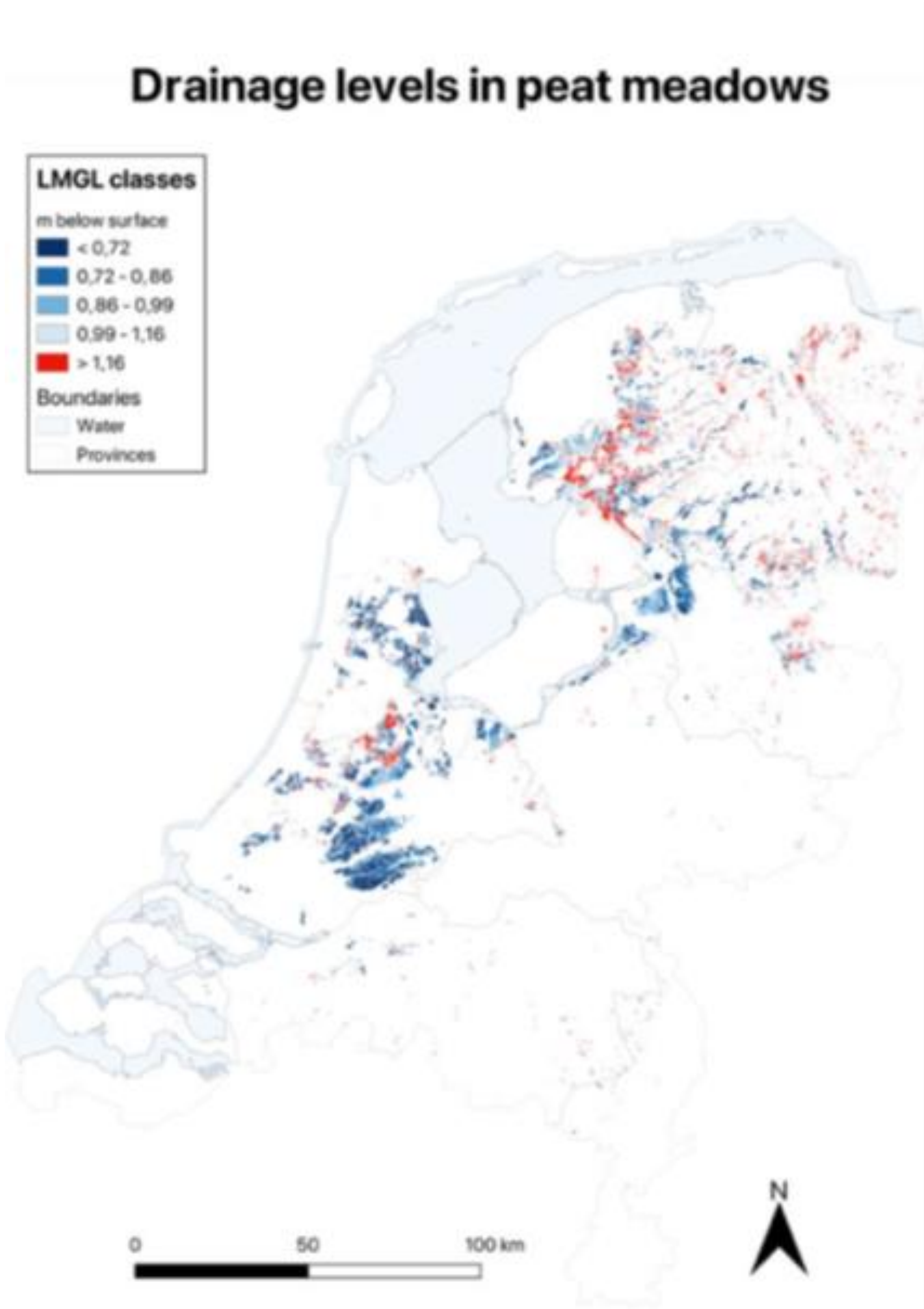


Fig. 2: Map of peat meadows classified per drainage level. Source: Grikinyte (2024).

We calculated and compared the mean monetary value of ES and EDS for the five peat meadows drainage classes. Moreover, we compared C emissions/sequestration biophysical and monetary results between drained peat meadows and semi-natural peatlands[2].

[2] Comparison of peat meadows with semi-natural peatlands was not relevant for the other ES and EDS: there is no grass provision on semi-natural peatlands; semi-natural peatlands do not provide habitat to meadow birds (but do provide habitat to other species); there is no drainage in semi-natural peatlands, thus no building damage costs attributable to semi-natural peatlands.

3.1. Grass provision

3.1.1 Biophysical quantification

Grass provision from peatlands was estimated using data from the NL SEEA EA on grazed biomass provisioning ES (Table 2). This ES includes the provision of grass (silage and meadow grass), hay and fodder maize and is expressed in yield of fodder and grass (ton per ha per year). The ES is estimated by combining spatial data on crop types with statistics on harvested yields (silage grass, hay and maize yields are specified per province, grazed meadow grass yields are specified for 5 large regions).
The grazed biomass provisioning ES estimated in 2020 (most recent year available) was combined with the map of peat meadows to estimate grass provision by peat meadows in tons of grass/ha/year.

3.1.2 Monetary valuation

We used the grazed biomass ES monetary value from the NL SEEA EA, based on lease prices from agricultural land (Table2). The NL SEEA EA provides a monetary ES supply table with the total value of grazed biomass supply (in million euros/year) per province. This total value aggregates values of grazed biomass supplied by peat meadows, other types of grasslands and maize fields. To estimate the value supplied by peat meadows per province, we multiplied the total value provided in the NL SEEA EA by the ratio of peat meadow areas over the total area of grasslands and maize fields.

Table 2: Data inputs used for assessing grazed biomass provision ES.

Data inputs	Format	Units	Resolution (m)	Date	Source
Grazed biomass provision ES biophysical map	Raster	Tons/ha/year	10	2020	NL SEEA EA (Statistics Netherlands & Wageningen University and Research, 2024)
Grazed biomass provision ES monetary	Excel table	Million euros/Province/Year	n/a	2020	

3.2. Meadow birds habitat provision

We took the Black-tailed Godwit (*Limosa limosa*) breeding density as indicator of meadow birds habitat provision. Peat meadows are an important breeding habitat for Black-tailed Godwits, and the Netherlands is a critical habitat for this species. The bird is also considered an indicator for the suitability of meadows for meadow birds. The population of Godwit has declined by more than 70 per cent since the 1970s (Sovon Vogelonderzoek Nederland, 2024), and this decline is related to the intensification of grasslands management, which in turn is related to high drainage levels. In general, low water levels and monoculture grass species lead to few insects and other feed for the young birds. Hence, the density of Black-tailed Godwits is a good indicator of overall biodiversity in peat meadows, including plant species richness, abundance of insects and worms.

3.2.1 Biophysical quantification

We used the Black-tailed Godwit breeding density predicted by a species distribution model (Sovon Vogelonderzoek Nederland, 2022) to quantify Black-tailed Godwit breeding density in peat meadows (Table 3).

3.2.1 Monetary valuation

We applied a cost-based approach for non-use values (NCAVES & MAIA, 2022), to allocate a value in Euros per Black-tailed Godwit breeding pair. Based on the budget allocated to improve peat meadow birds habitat in the Dutch CAP National Strategic Plan, we calculated a cost per hectare of improving meadow birds habitat. Next, using data from the national action plan to support Black-tailed Godwits (Aanvalsplan Grutto, 2020), we estimated the area (in hectares) of improved habitat required to increase the Black-tailed Godwits breeding pairs density by one. The value of one Black-tailed Godwit breeding pair corresponds to this later area multiplied by the cost per hectare of improving meadow birds habitat.

Table 3: Data inputs used for assessing meadow birds habitat provision ES.

Data inputs	**Format**	**Units**	**Resolution (m)**	**Date**	**Source**
Modeled Black-tailed Godwit breeding densities in the Netherlands.	Raster	Number of breeding pairs/pixel	250	2021	Sovon Vogelonderzoek Nederland (2022)

3.3. CO_2 emissions

3.3.1. Biophysical quantification

We used maps of CO_2 emissions from peat(y) soils and C sequestration in above and below ground biomass from the SEEA EA Carbon Account (Table 4). The two maps were combined with the map of peat meadows and semi-natural grasslands to estimate net C emissions (C emissions minus C sequestration) for each drainage class and the semi-natural peatlands.

3.3.2. Monetary valuation

In line with the NL SEEA EA (Statistics Netherlands & Wageningen University and Research, 2024), we used the efficient price of CO_2, which is the price at which the necessary cumulative reduction in CO_2 emissions to achieve internationally agreed targets is achieved at the lowest costs (PBL Netherlands Environmental Assessment Agency, 2016).

Table 4: Data inputs used for assessing C emissions and sequestration

Data inputs	Format	Units	Resolution (m)	Date	Source
C emissions from peat oxidation	Raster	Tons/ha	10	2013	NL SEEA EA (Lof et al., 2017)
C sequestration in above and below ground biomass	Raster	Tons/ha	10	2013	NL SEEA EA (Lof et al., 2017)

3.4. Building damages

For building damages, we directly used available monetary values (Table 5) without prior biophysical quantification. We used the outputs of a model that estimates building damage costs due to soil subsidence at municipal scale (https://klimaatschadeschatter.nl/). The modelled total building damage costs per municipality include damages in rural but also urban areas. However, we expect that soil subsidence in peat meadows only have an impact on building in rural areas. To take this into account, we multiplied the total damage costs by the ratio of rural built-up areas over the total built-up areas in each municipality. Next, we distributed the damage costs to peat meadow areas based on their drainage level, using the LMGL as proxy. For each municipality, we scaled the drainage level of pixels from 0 to 1. The damage costs attributed to each pixel was calculated as the total municipal damage costs multiplied by the ratio of the pixel drainage level over the sum of all pixels drainage levels in the municipality.

Table 5: Data inputs used for assessing building damage costs.

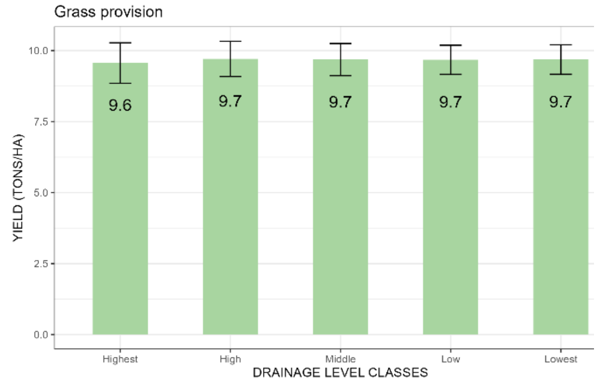
Data inputs	Format	Units	Resolution (m)	Date	Source
Cumulative buildings damage monetary data per municipality.	Excel table	Euros/ Municipa-lity	n/a	2018 - 2050	Climate Damage Atlas (Klimaatschadeschatter.nl)
lowest mean groundwater level (LMGL) map	Raster	Meters below surface	250x250	2019	Klimaateffectatlas.nl

4. Results

4.1. Grass provision

The biophysical value of grass provision did not show any relation to the drainage level class (Fig.3). This is due to the resolution of the yield input data. Data on harvested yields were available at provincial scale (silage, hay and maize) or for 5 large regions (grazed biomass), thus did not capture fine scale spatial variations linked to differences in drainage practices.

Thus, differences in the NPV of grass provision (Fig. 4) were due to differences in the monetary valuation of the service.



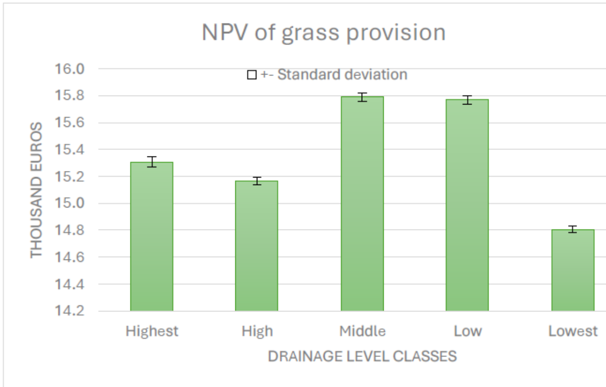


Fig. 3: grass provision per peat meadows drainage class in tons/ha.

Fig. 4: Net Present Value (NPV) of grass provision per peat meadows drainage class, in thousand euros per ha, 2020-2050. Source: Grikinyte (2024).

4.2. Meadow birds habitat provision

As expected, the breeding density of Black-tailed godwits increased with rising groundwater levels (Fig. 7). A high groundwater level in grasslands is favourable for meadow birds, because the vegetation growth is delayed in spring, leading to a relatively open vegetation beneficial for foraging by chicks. Moreover, wet top soil layers are more accessible for foraging and contain a higher abundance of worms than dry soils.

As we used the monetary value of one Black-tailed godwits breeding pair to value the meadow birds habitat provision, this ES NPV follows the same pattern as the Black-tailed godwits breeding density.

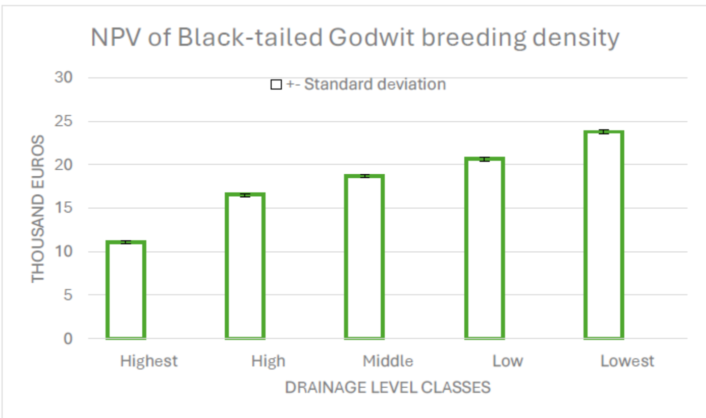


Fig. 7: Net Present Value (NPV) of meadow birds habitat provision per peat meadows drainage class, in thousand euros per ha, 2020-2050. Source: Grikinyte (2024).

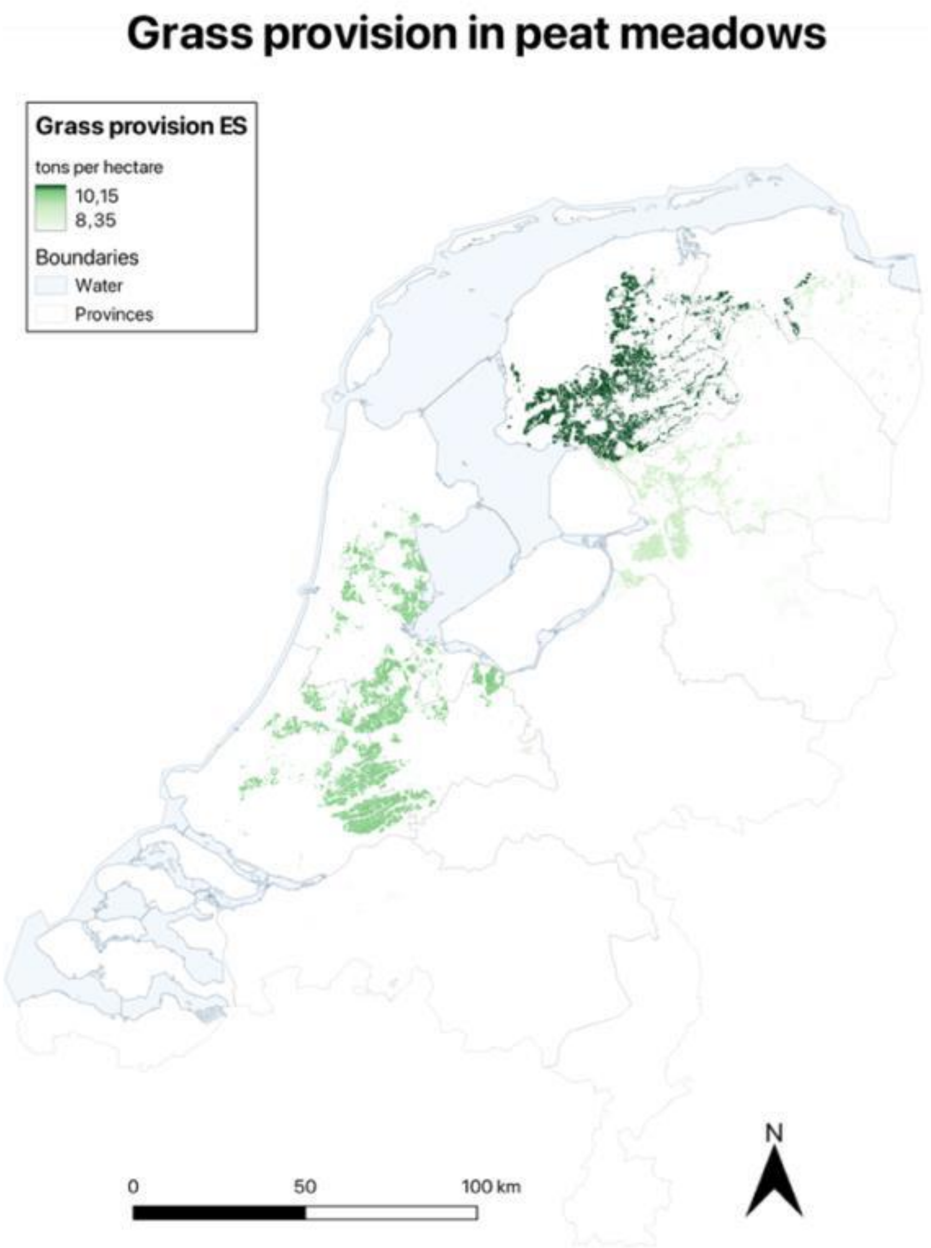


Fig. 5: map of grass provision in peat meadows in tons per ha, 2020. Source: Grikinyte (2024).

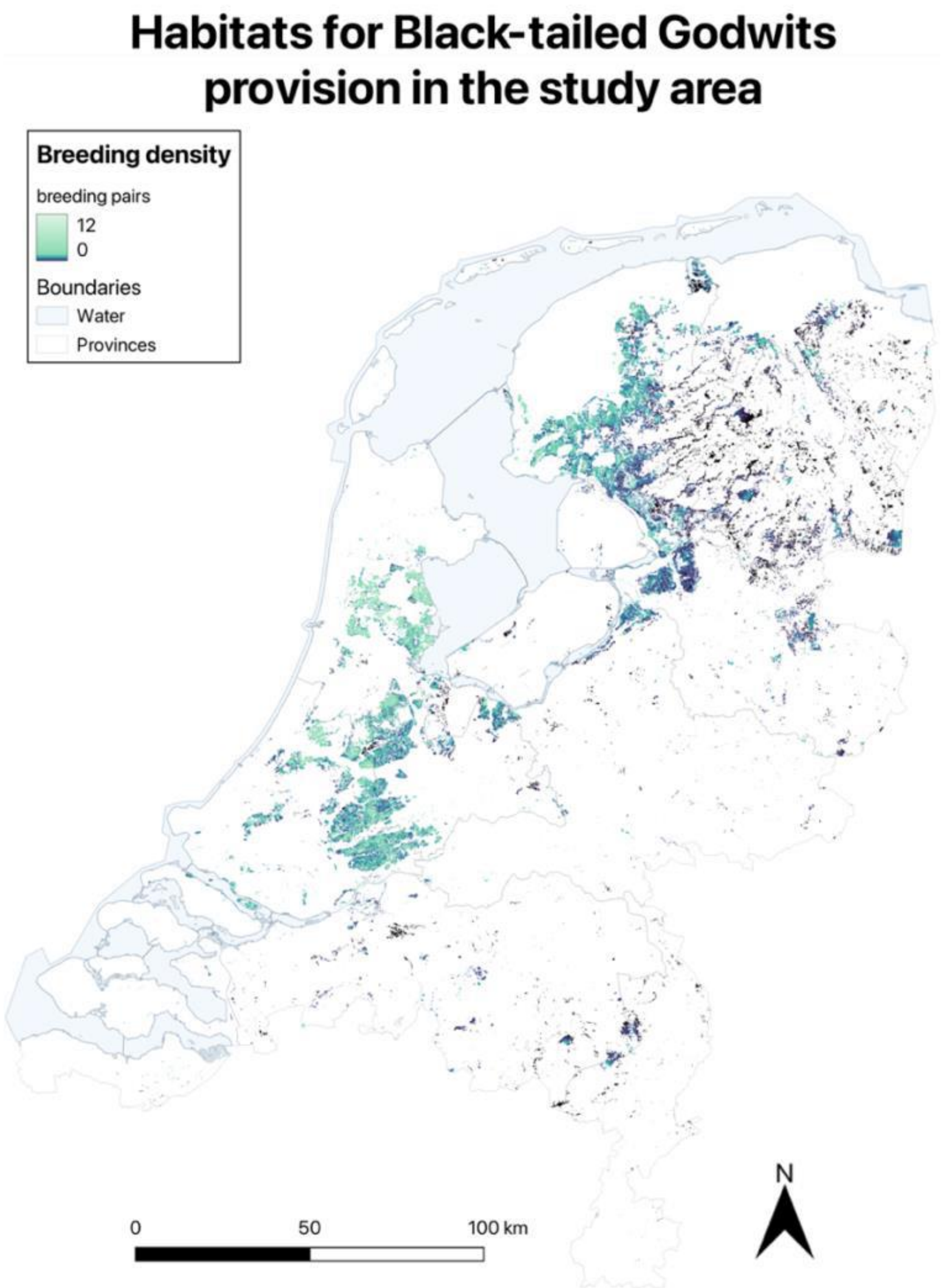
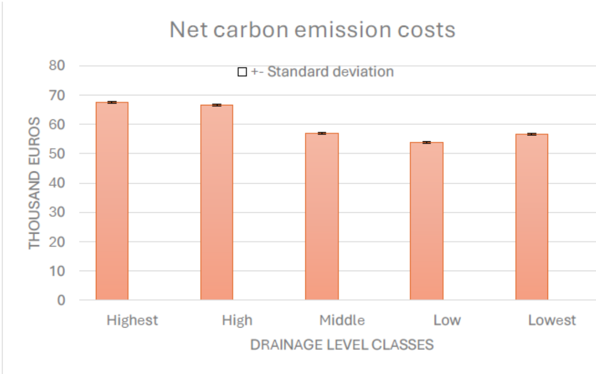


Fig. 6: map of Black-tailed Godwits breeding parts density (breeding pairs per 250m pixel), 2021. Source: SOVON in Grikinyte (2024).

4.3. Carbon emissions

As expected, net C emissions increased with drainage level (Fig.10). However, Net C emissions in undrained semi-natural peatlands were similar to net C emissions from peat meadows - lower than net C emissions from peat meadows in the highest drainage class, but slightly higher than net C emissions from peat meadows in the middle and lowest drainage classes (Fig. 11). In the Netherlands, preserving natural wetlands is challenging due to the drainage and soil subsidence of adjacent agricultural land (Verhagen et al., 2009). Semi natural peatland areas, which are situated higher up than the surrounding drained peatlands, tend to lose water (van den Born et al., 2016) and oxidate, which leads to C emissions.



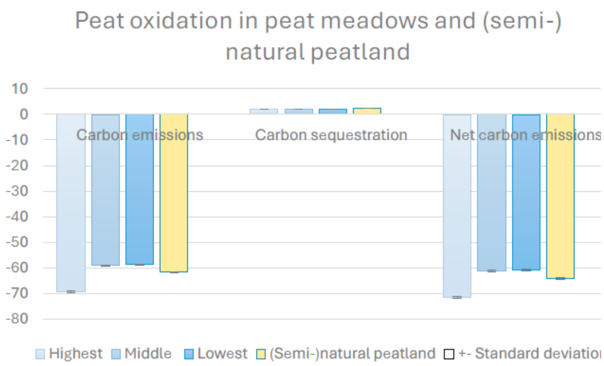


Fig. 10: Net Present Value (NPV) of net carbon emissions per peat meadows drainage class, in thousand euros per ha, 2020-2050. Source: Grikinyte (2024).

Fig. 11: Carbon emissions and sequestration in peat meadows and semi-natural peatlands. Source: Grikinyte (2024).

4.4. Buildings damages

The building damage costs model provides lower and upper estimates, which correspond to different assumptions regarding repair costs of damaged buildings. We present lower and upper estimates of total building damage costs over 2020-2050 in Figure 12, but the lower estimate was used in the results presented in Section 5.

The relationship between drainage levels in peat meadows and building damage costs was unclear and inconsistent. The findings contradict literature suggesting that lower groundwater levels and higher subsidence increase building damage. This inconsistency likely results from explaining factors that could not be taken into account when spatially allocating municipal damage costs to peat meadows: building foundation types, soil types and urban vs rural repair costs.

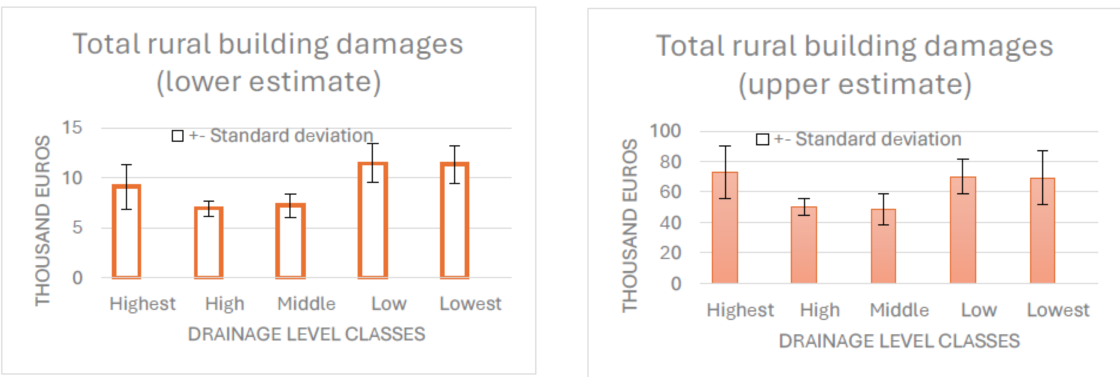


Fig. 12: NPV of rural buildings damage costs in thousand euros per ha, by drainage level class, 2020-2050. Source: Grikinyte (2024).

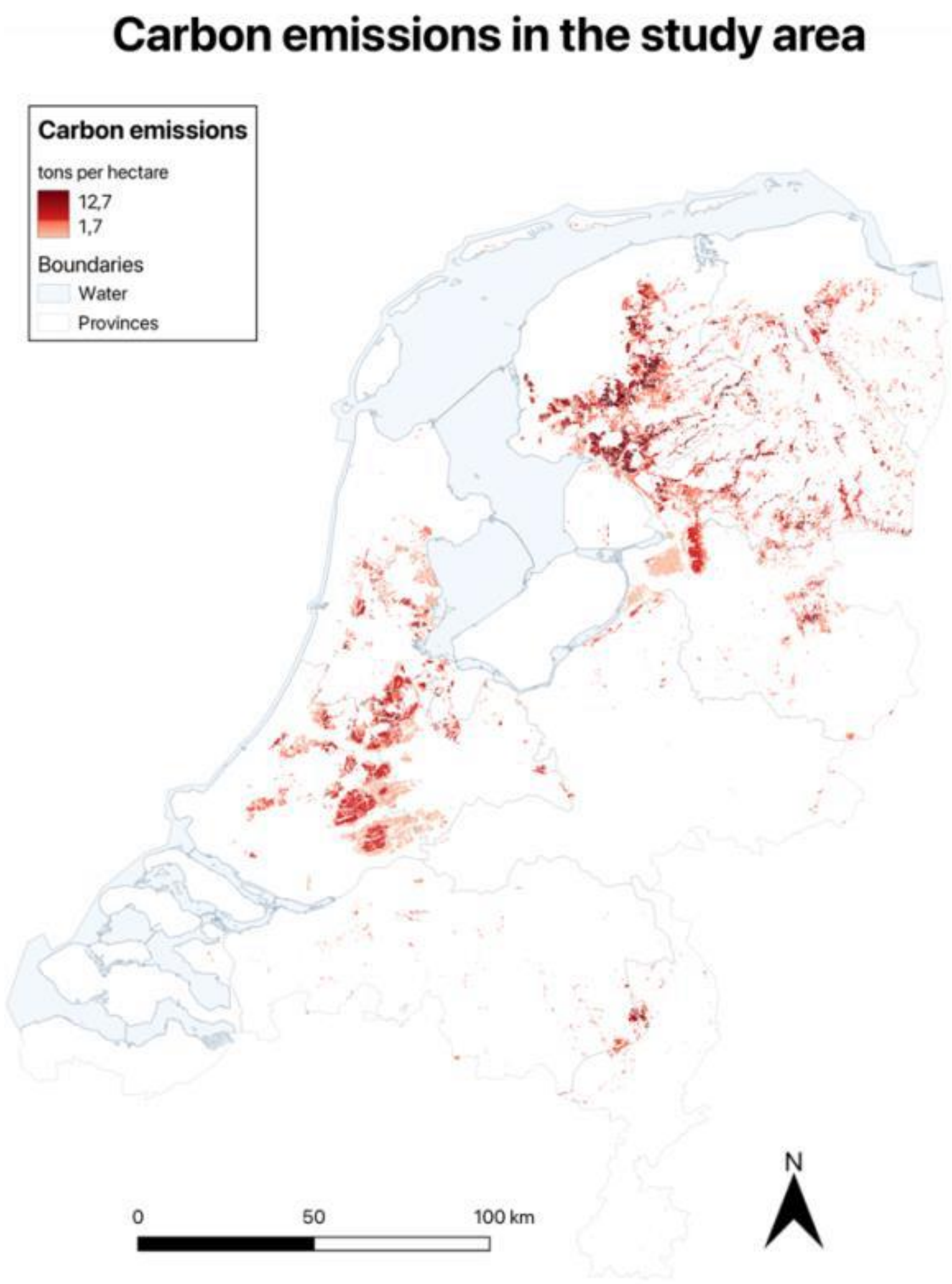


Fig. 8: map of C emissions, in tons C per ha, 2013. Source: Grikinyte (2024).

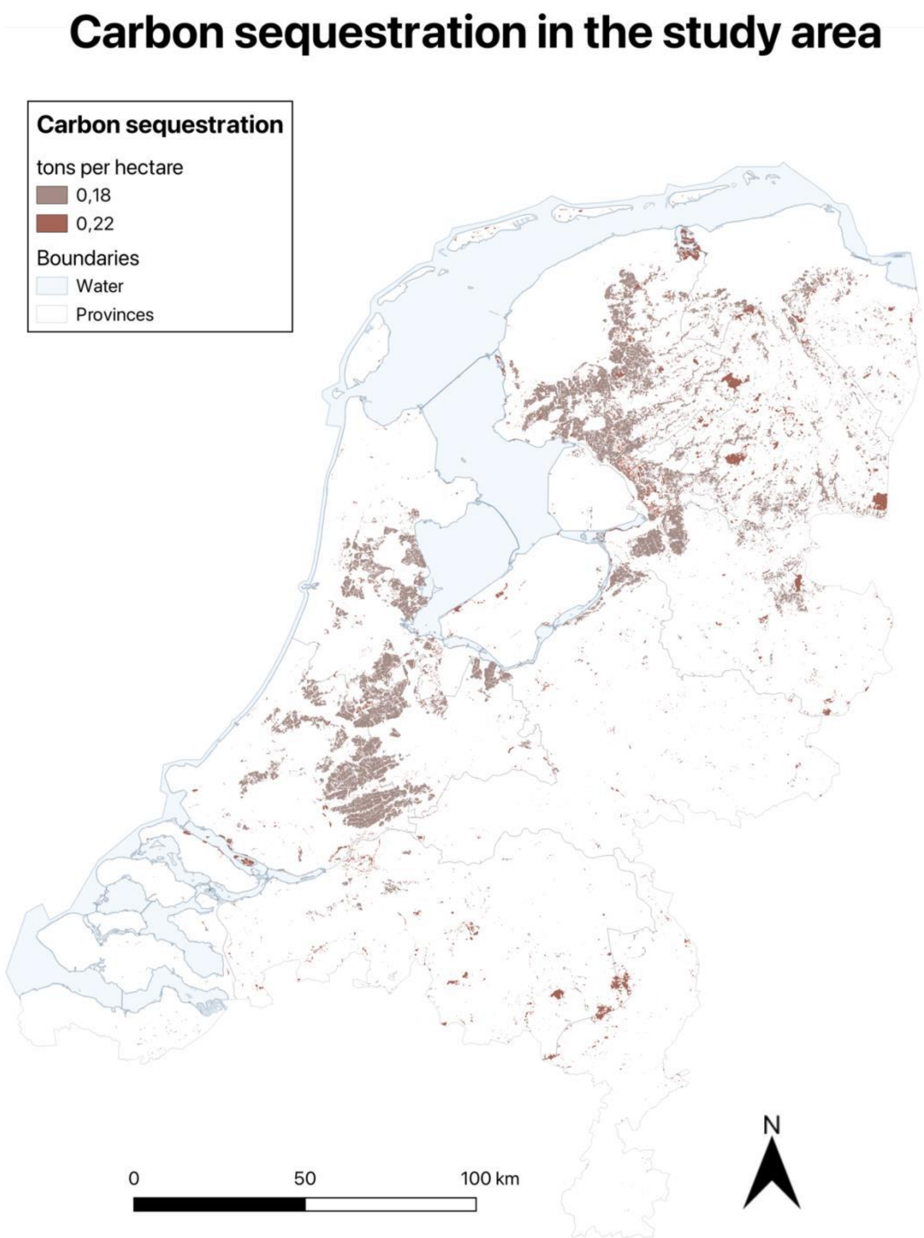


Fig. 9: map of C sequestration, in tons C per ha, 2013. Source: Grikinyte (2024).

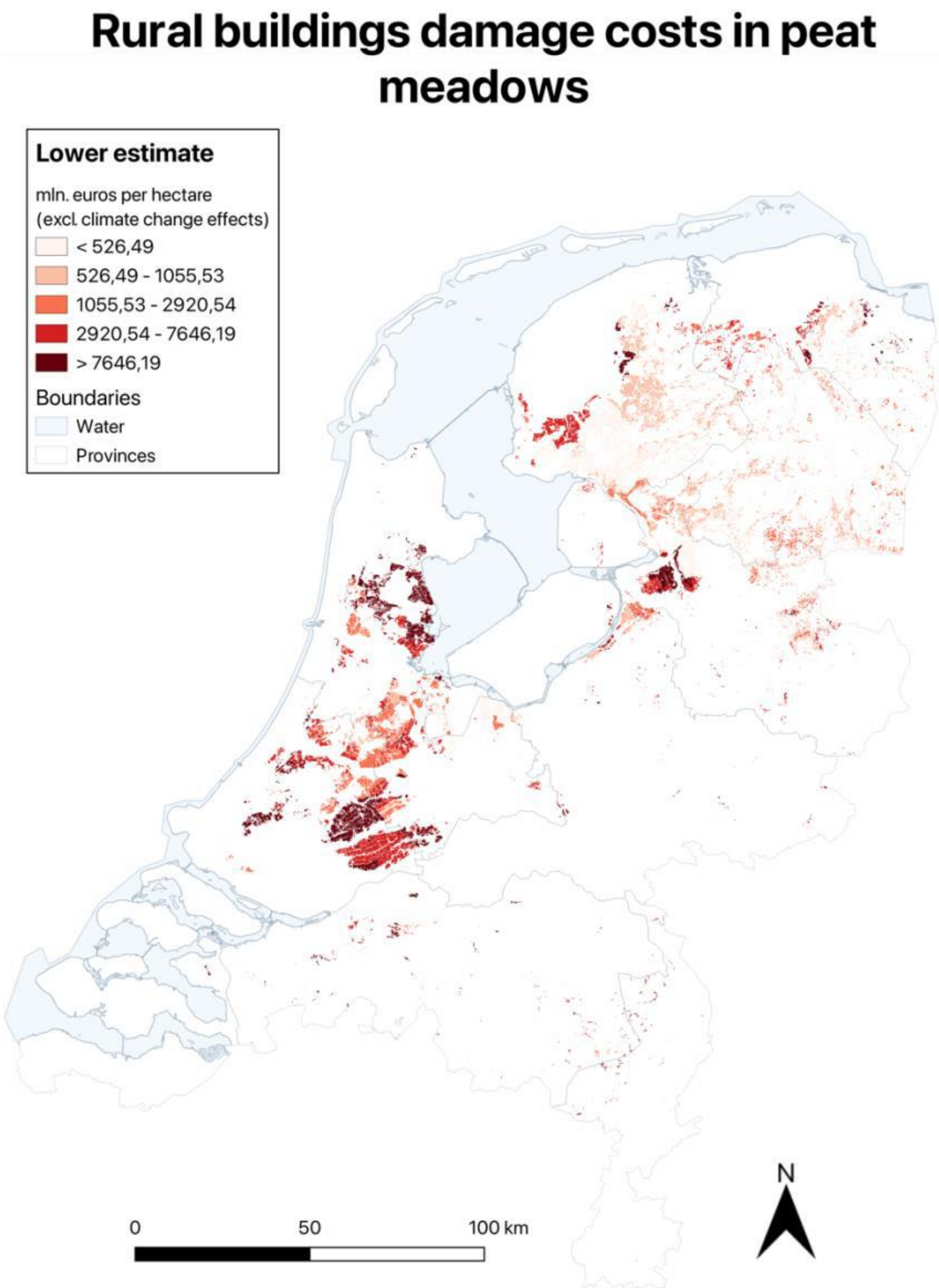


Fig. 13: map of rural building damage costs lower estimate, in million euros per ha, 2018-2050. Source: Grikinyte (2024).

5. Integration in ecosystem accounts

5.1. Carbon emissions

Following the approach proposed in SELINA Deliverable D5.5 (Section 6.1), C emissions are presented in an extended global climate regulation ES supply table (Table 6). In standard ecosystem accounts, only the gross ES, C sequestration in above and below ground biomass, is included in the ES supply table. Integrating the EDS of C emissions requires extending the standard ES supply table with the EDS, C emissions, and the Net ES (C sequestration minus C emissions). In the case of peat meadows in the Netherlands, the net global climate regulation ES is negative, due to drainage and peat oxidation.

Table 6: Integration of the C emissions EDS in a SEEA EA ES supply table.

Ecosystem type	**Unit**	**Peat meadow**					
Drainage level	**Groundwater level in m. below the surface**	> 1.16	0.99 – 1.16	0.86 – 0.99	0.72 – 0.86	< 0.72	All
Gross Ecosystem Service							
C sequestration in above and below ground biomass	ton C/ha/year	0.18	0.18	0.18	0.18	0.18	0.18
	Kton C/year	8	7	7	8	7	37.5
Ecosystem Disservice							
C emissions from peat oxidation	ton C/ha/year	5.9	5.8	4.8	4.4	4.6	5.46
	Kton C/year	246	226	195	205	190	1,062.5
Net Ecosystem Service							
Net C sequestration	ton C/ha/year	5.7	5.6	4.6	4.3	4.4	4.9
	Kton C/year	239	219	187	197	183	1,025

Note that peat meadows drainage may also lead to offsite peat oxidation and C emissions in neighbouring semi-natural peatlands (see 4.3). Following the above approach of a combined ES and EDS supply table, these offsite C emissions are attributed as EDS to the semi-natural peatlands ET. However, these offsite C emissions are negative externalities of the peat meadows management for milk production.

5.2. Building damages

Damages to buildings that result from soil subsidence are negative externalities that result from a decrease in ecosystem condition (see Section 5.3 in D5.5). Thus, damages to buildings cannot be integrated as increased EDS or reduced ES flows in extended SEEA EA ES/EDS supply tables. However, damages to buildings can be taken into account in the monetary valuation of ecosystem assets, as shown in Section 5.3 below.

5.3. Integration of ES and EDS/NE in Asset valuation

We estimated how the peat meadows ecosystem asset value varies with different levels of drainage. In the SEEA EA, the value of an ecosystem asset is the sum of discounted future flows of ES provided by the asset (Section 8.2.3 in United Nations et al. (2021)). In our study, we extended this definition to take into account EDS and negative externalities. Thus, the ecosystem asset value was calculated as the sum of discounted future flows of ES minus the sum of discounted future flows of EDS (CO_2 emissions) and negative externalities (building damages) provided by the asset.

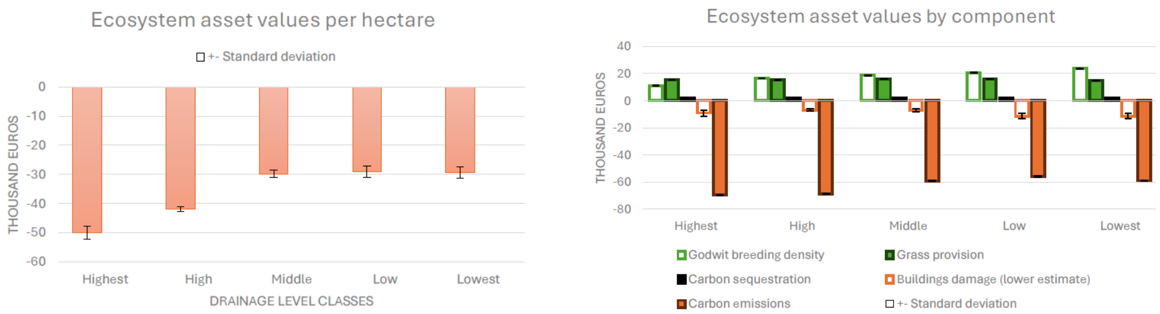


Fig. 14: Peat meadows ecosystem asset value per ha in each drainage class, total (left) and disaggregated per ES/EDS/Negative externality (right).

Figure 14 shows that the peat meadows asset value is negative for all drainage classes, and that peat meadows in the highest drainage class have the lowest ecosystem asset value. Those results are largely driven by the EDS of carbon emissions. Those results were obtained with the lower estimate of building damage costs. With upper estimates of building damage costs, the asset value would become even more negative.

6. Discussion

Our results show how negative externalities associated to milk production on peat meadows can be included in SEEA EA through extended ES/EDS supply tables. Furthermore, we demonstrate how flows of ES,EDS and negative externalities can be compared and integrated in ecosystem asset valuation. In the case of dairy production on peat meadows, the NPV of ES, including grass provision, is much lower than the NPV of EDS and negative externalities, so that the ecosystem value of peat meadows is negative when taking into account dairy production negative externalities.

It should be noted that building damages do not actually take place in the peat meadows ecosystem assets, but in neighbouring built-up ecosystem assets. In this study, we spatially allocated the building damages in built-up areas (estimated at municipality scale) to the peat meadows located in the municipality (see Section 4.4). Therefore, building damages were assigned to the peat meadows. Using the same line of reasoning, C emissions that take place in semi-natural peatlands neighbouring drained peat meadows (see 4.3) could also have been assigned to drained peat meadows in Table 6. However, this would mix onsite C emissions in peat meadows with offsite C emissions generated by drainage. For clarity and consistency, it appears preferable to assign the supply of EDS flows to the ecosystem asset where they take place. Following this principle, building damage should be assigned to the rural built-up ecosystem types. Thus, when assessing negative externalities of dairy production, EDS flows from peat meadows but also from semi-natural peatlands and rural built-up areas indirectly affected by drainage will have to be included. Those negative externalities can then be combined with the economic output, ES use (grass provision) and production (meadow birds habitat provision) by the dairy sector for further analysis, such as estimating a shadow price for milk produced in peat meadows areas.

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Annex 3: Negative externalities from forest fires in Northern Portugal

Lead beneficiary: CIBIO/BIOPOLIS
Author/s: Ângelo Sil, A. Sofia Vaz, Cláudia Carvalho-Santos, Silvana Pais, Adrián Regos, Bruno Marcos, João Gonçalves, João P. Honrado

1. Preface

This report is an Annex of Deliverable D5.5 Specifying and testing how externalities and EDS can be included in ecosystem accounts. It presents in detail the analyses that were carried out in the test site 17 – Northern Portugal for the purpose of Task 5.1.

2. Introduction

2.1. Site description

The test site is the Northern Portugal - NUTS-II EU administrative region, covering ca. 21.515 km^2 (Fig. 1). This region has ca. 3.6 million inhabitants, of which 90% live in coastal areas. The employed population (43% of the total inhabitants) primarily works in the tertiary (64%) and secondary (34%) economic sectors, while only 2% are currently employed in the primary sector (INE, 2021). Northern Portugal is predominantly mountainous, with elevation ranging from 0 to 1545 m. It comprises two biogeographical regions, the Atlantic (west) and the Mediterranean (centre and east) (EEA, 2016). In the western coastal areas, the climate is characterised by temperate summers and mild winters, whereas the eastern inland region experiences hot summers and cold winters (Beck et al., 2018). Mean annual precipitation varies from ca. 540 to 1500 mm, while mean annual temperature ranges from ca. 10 to 15 ºC (IPMA, 2011).

The main Land Use and Land Cover (LULC) types at the test site are forests (37%), agriculture (29%), and shrublands (22%) (DGT, 2018). Protected areas account for 25% of the total surface area (ICNF, 2023). Forests and seminatural areas in the region provide a wide range of provisioning (e.g., timber, firewood, wild mushrooms), regulating (e.g., carbon storage, soil retention, water regulation), and cultural (e.g., nature-based tourism) ES (Ribeiro et al., 2011). Fire activity in Northern Portugal is very high, with 11.972 fires/year burning 48.828 ha/year, primarily forests (56%) and shrublands (40%) (INE, 2024). The 2017 fire season was the most devastating nationwide and among the worst in Northern Portugal, resulting in ca. 90 thousand hectares of burned land (INE, 2024). The intensification of fire regimes in Portugal, mainly driven by human-induced changes in landscape flammability and ignition patterns coupled with more severe climate conditions (Fernandes et al., 2014), has resulted in substantial ecological and socioeconomic damage in recent decades (Mateus and Fernandes, 2014; Mendes et al., 2021).

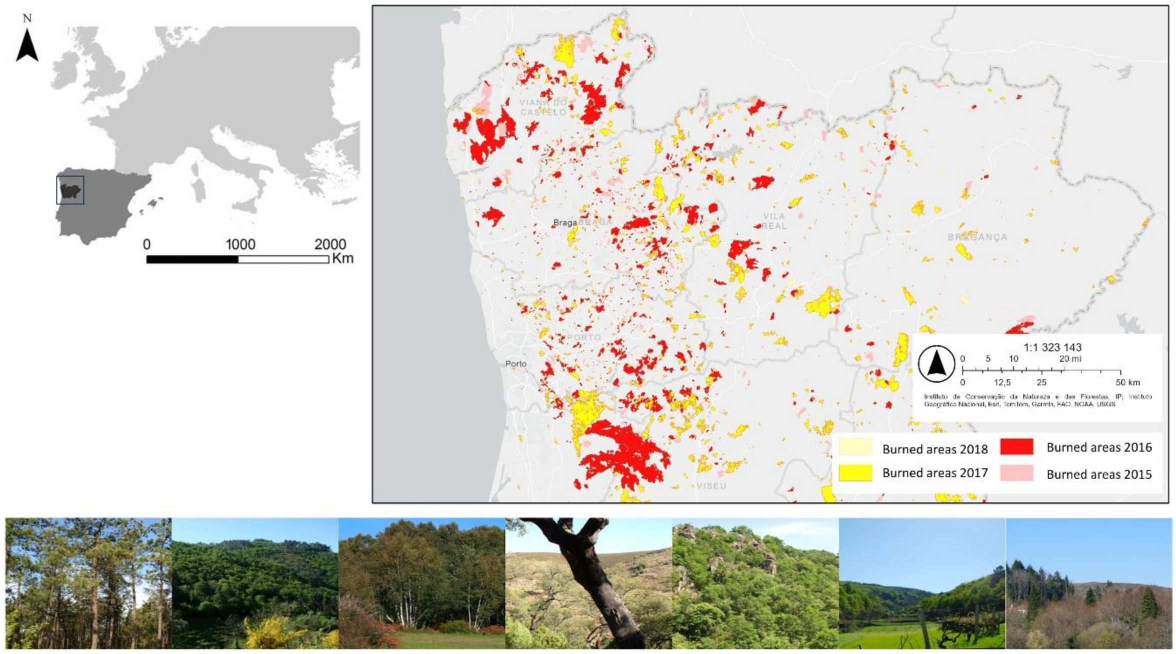

Figure 1: Northern Portugal test site. Left panel: Northern Portugal (NUTS-II EU) test site. Right panel: area burned in Northern Portugal between 2015 and 2018. Bottom panel: forested landscapes in Northern Portugal.

2.2. Negative externalities

The impacts of forest fires in Portugal can be seen as negative externalities driven by poor practices and socioeconomic changes in forestry and agricultural sectors, such as 1) neglected forest management (e.g., lack of fuel management); 2) intensified forest management (e.g., monoculture plantations), 3) inappropriate afforestation (e.g., highly flammable species in former croplands/pasturelands); and 4) farmland abandonment (Mateus and Fernandes, 2014). Forest fires can foster EDS that directly affect people by threatening human safety, health, and infrastructure (Augusto et al., 2020; Ribeiro et al., 2020). Besides, forest fires can negatively impact pre-existing ecosystem service supply by promoting soil erosion, reducing biomass supply, altering carbon stocks, or increasing losses in nature-based recreation potential (Sil et al., 2019). Social costs of forest fires in Portugal average ca. €370 million/year, of which ca. 70% are related to ecological damages (Mateus and Fernandes, 2014; Mendes et al., 2021). In this context, quantifying and valuing the negative impacts of forest fires, particularly on ES, is crucial to developing effective policies for fire risk management, natural resource management, and incorporating natural capital into decision-making. This test site aims to demonstrate how forest fires negatively impacted a set of key ES in Northern Portugal (Table 1) and how these negative externalities can be integrated into ecosystem accounts.

Table 1: Ecosystem services and negative externalities due to forest fires assessed in the Northern Portugal test site.

Ecosystem Service			Negative externalities due to forest fires
Provisioning	Fibres and other materials from cultivated plants	Timber provision: the volume of woody biomass in forests (i.e., potentially harvestable).	Reduction of timber provision: due to damages in the volume of woody biomass in areas burned at high/very high fire severity.
		Cork provision: the volume of cork in cork oak forests (i.e., potentially harvestable).	Reduction of cork provision: due to damages in cork oak forests in areas burned at high/very high fire severity.
Regulating & Maintenance	Regulation of chemical composition of atmosphere and ocean	Carbon storage: the amount of carbon stored by above- and belowground biomass and dead organic matter in forests and shrublands.	Reduction of carbon storage: due to losses in carbon stocks in burned areas according to different levels of fire severity.
	Control of erosion rates	Soil retention: the amount of soil retained by forests and shrublands.	Reduction of soil retention: due to vegetation losses in burned areas according to different levels of fire severity.
Cultural	Physical and experiential interactions with the natural environment	Nature-based tourism: recreation potential in forested landscapes.	Reduction of nature-based tourism: loss in recreation potential according to different levels of fire severity.

3. Data and methods

3.1. General approach

Negative externalities, i.e., the reduction of pre-existing ecosystem service supply due to forest fires, were evaluated in both biophysical and monetary units by developing an integrated modelling framework coupling spatial explicit data on land cover and forest fires with public statistics, modelling tools, and relevant scientific literature (Fig. 2). The application of the modelling framework focused on forest and shrubland areas impacted by forest fires that occurred in 2017 in Northern Portugal. First, ES supply was estimated in the pre-fire environment, i.e., within the area corresponding to fire perimeters prior to fire occurrence.

Then, the impacts of forest fires on ES were analysed based on the extent and severity of the burned area to estimate the ES supply in the post-fire environment, i.e., within the area corresponding to fire perimeters after fire occurrence. Finally, the reduction in ES supply due to forest fires was estimated as the difference between ES estimates in the pre-fire and post-fire environment.

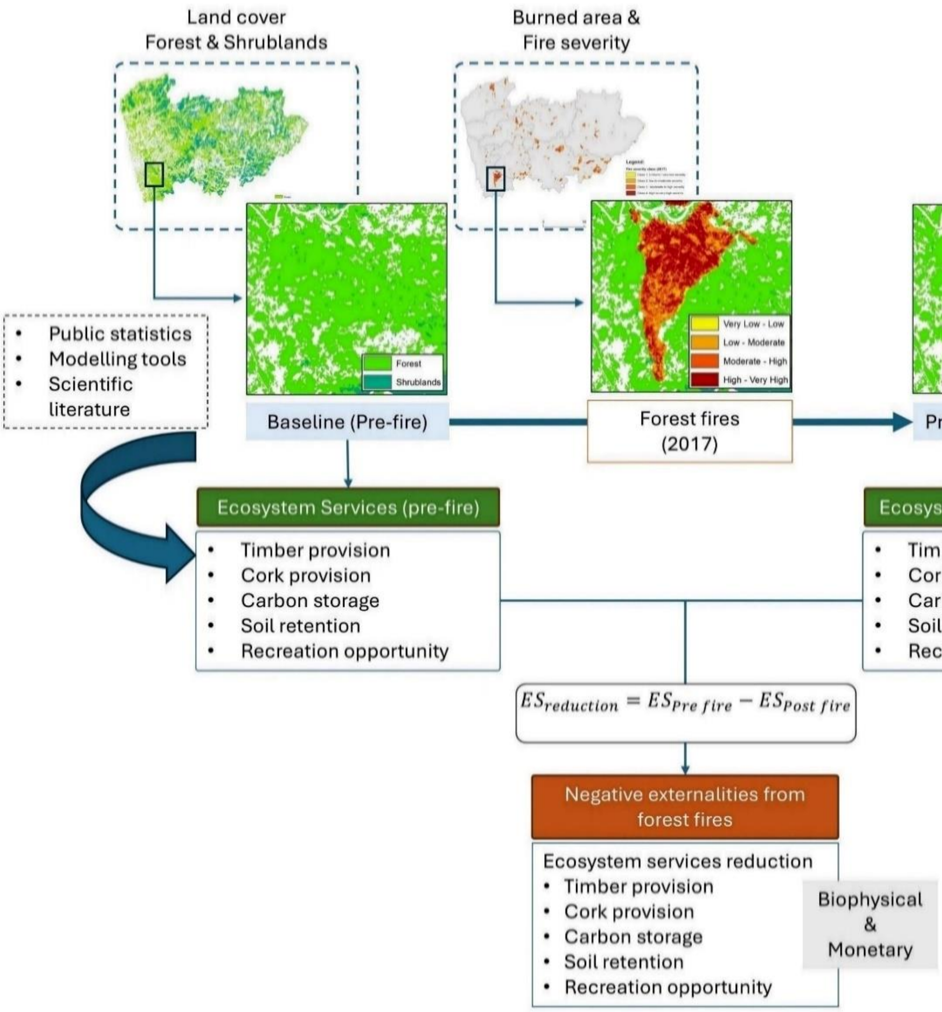

Figure 2: Diagram of the modelling workflow applied to assess negative externalities from forest fires in Northern Portugal.

3.2. Land cover data

The cartographic basis used in the Northern Portugal test site was the Carta de Ocupação e uso do Solo (COS) - the Portuguese land use and land cover spatial database for 2015 (DGT, 2018). COS is a spatial vector database (i.e., based on polygons) that represents homogeneous units of land use and land cover, i.e., any area equal or greater than the minimum cartographic unit defined (1 ha), with a distance between lines equal or greater than 20 m and whose percentage of a certain class of land use and land cover is equal or greater than 75% of the total delimited area. It has a hierarchical structure of five levels of detail and 48 classes at the fifth level. The georeferenced system is ETRS89/PT-TM06, and the cartographic projection is Transverse Mercator (DGT, 2018).

Forest and shrubland land cover classes were used to carry out the various assessments in the Northern Portugal test site. Forest extent is ca. 792 thousand hectares and comprises several broadleaved and coniferous species (Table 2). Shrubland extent corresponds to ca. 479 thousand hectares.

Table 2: Extent of forest land cover in the Northern Portugal test site.

Forest land cover	Area (ha)	% of total forested area
Chestnut	12,990.5	1.6%
Cork oak	39,080.2	4.9%
Eucalyptus	189,259.0	23.8%
Holm oak	8,047.4	1.0%
Invasive species	3,187.7	0.4%
Maritime pine	294,412.6	37.2%
Other broadleaved	96,623.9	12.2%
Other coniferous	22,713.0	2.9%
Other oaks	125,958.5	15.9%
Stone pine	579.5	0.1%

3.3. Burned area and severity data

In 2017, forest fires burned ca. 34.000 hectares of forest and 39.000 hectares of shrublands, representing 4% of the forested area and 6% of the shrubland area in Northern Portugal. High to very-high fire severity areas (Class 4) accounted for 31% of the total burned area, moderate to high severity (Class 3) for 29%, low to moderate severity (Class 2) for 35%, and very-low severity (Class 1) for 5% of the total burned area (Table 3).

The quantification of burn severity followed the SeverusPT approach (Gonçalves et al., 2024a), for which data is available at https://data.severuspt.ipvc.pt/products (visited: 07/02/2025; Gonçalves et al. (2024b)). This approach targets the fire perimeters above 10 hectares from the European Forest Fires Information System (EFFIS) database, which covers, according to this dataset, roughly 99.6% of the burned area in mainland Portugal from 2006 to 2023. A delta-based approach using the Normalized Burn Ratio (NBR) spectral index (Eqn.1) was employed to calculate burn severity indicators by comparing pre- and post-fire images. NBR is based on the normalised ratio between the near-infrared (NIR) and the shortwave infrared (SWIR) bands. Sentinel-2 level-2A (surface reflectance) image time series, with 5-day temporal resolution and 20 meters of spatial resolution, were employed in severity estimates. Clouds and cirrus were removed from each image based on quality bands.

Two median image composites were generated for pre- and post-fire conditions using three months of imagery. To minimise seasonality effects, the post-fire composite was compared to the same period in the previous year (i.e., the homologous pre-fire composite). The delta NBR (dNBR; Eqn. 2) index was then calculated to estimate burn severity.

NBR=(NIR-SWIR)/(NIR+SWIR) (Eqn. 1)

dNBR = NBRpre-fire-NBR post-fire(Eqn.2)

The severity classes for the dNBR indicator were derived from the analysis of value distributions from 2018 to 2023 based on products generated by the SeverusPT project. Class thresholds were determined using quintiles of the distribution (for values greater than zero), producing four categories (ranked in ascending severity order): Class 1 <= 0.01: unburnt / very-low severity; Class 2 [0.01, 0.36[: low to moderate severity; Class 3 [0.36, 0.55[: moderate to high severity; Class 4 >= 0.55: high to very-high (Fig. 3).

The burn severity products were validated through an in-field campaign that employed the geometrically structured Composite Burn Index (GeoCBI; De Santis and Chuvieco (2009)) across 120 survey plots within 28 different fire perimeters burned areas and 120 survey sites. Results suggest the best scores for the Sentinel-2-derived products and dNBR, with an R2 equal to 0.68 (non-linear model), a Spearman correlation of 0.71, and a non-linear correlation equal to 0.73.

Table 3: Extent of burned forest and shrubland areas (total and proportion by severity class) due to forest fires in 2017 in the Northern Portugal test site.

Land cover	**Burned area (ha)**	**Burned area (%) per fire severity class**			
		1	2	3	4
Chestnut	339.9	5%	38%	23%	35%
Cork oak	1735.4	1%	46%	31%	22%
Eucalyptus	10569.8	4%	25%	30%	41%
Holm oak	335.5	1%	49%	35%	16%
Invasive	876.6	1%	34%	25%	41%
Maritime pine	13599.4	5%	32%	28%	35%
Oaks	2790.6	11%	38%	23%	29%
Other Broadleaves	3080.2	6%	46%	26%	22%
Other Conifers	910.9	6%	33%	24%	37%
Shrublands	39443.8	5%	37%	29%	29%
Stone pine	8.9	14%	73%	11%	2%
Total	73691.0	5%	35%	29%	31%

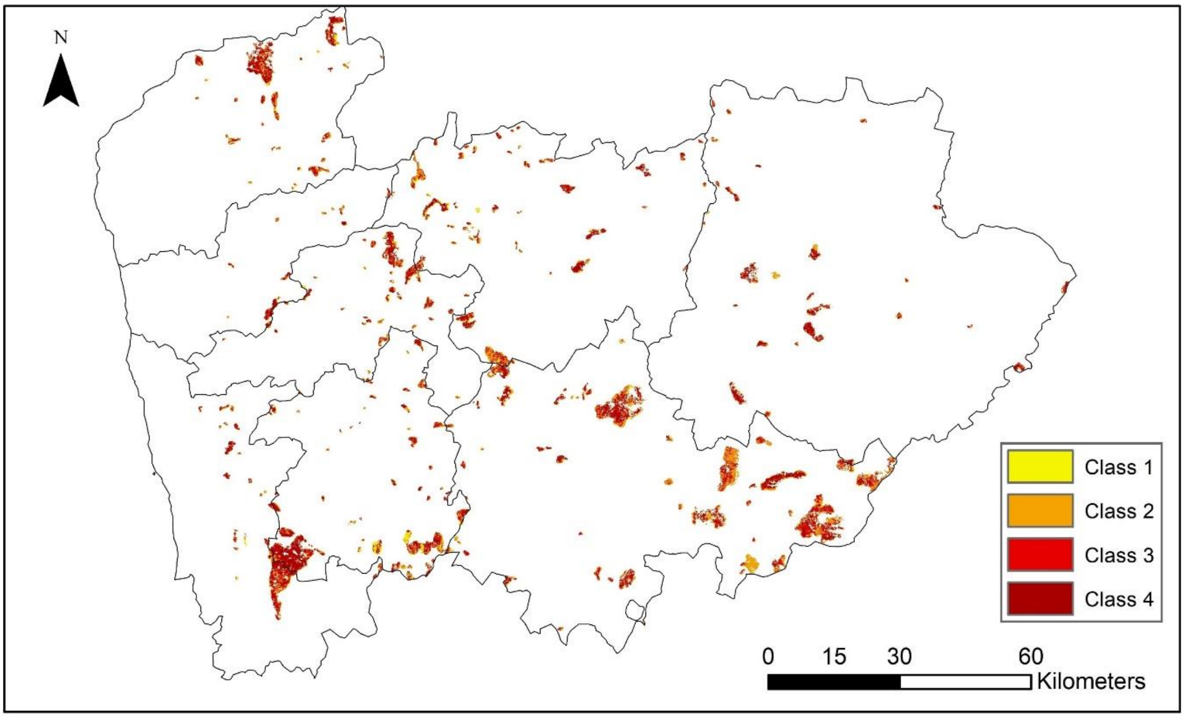

Figure 3: Fire perimeters per burn severity classes for 2017 forest fires in Northern Portugal. Class 1: unburnt/very-low severity; Class 2: low to moderate severity; Class 3: moderate to high severity; Class 4: high to very-high severity.

3.4. Assessment of negative externalities

3.4.1. Timber provision reduction

3.4.1.1. Biophysical quantification

Losses in woody biomass volume (m^3) in areas burned at moderate/high or high/very high fire severity were used as a biophysical indicator of reduced timber provision due to forest fires. Biophysical estimates followed three main steps. In step 1, the volume of potentially harvestable woody biomass in maritime pine, eucalyptus and oak forests (pre-fire) was estimated from the Portuguese National Forest Inventory - IFN6 in 2015 (ICNF, 2015). First, data on growing volume (i.e., living trees) by forest species and NUTS III region in Northern Portugal was collected. Next, woody biomass volume estimates by forest species and NUTS III were spatialised by combining IFN data with land use and cover spatial data (Section 3.2 – Land cover data). Then, woody biomass volume was projected for 2017 by applying annual growth rates (APA, 2018, 2022; ICNF, 2005). In step 2, the impact of forest fires on woody biomass volume was estimated. Thus, spatial data on woody biomass volume obtained in Step 1 was overlapped with spatial data on fire perimeters and respective fire severity classes (Section 3.3 – Burned area and severity data). A damage factor was applied according to fire severity classes, assuming that the potentially harvestable volume of woody biomass would be lost whenever fires burned at moderate/high or high/very high fire severity. In step 3, losses in woody biomass volume due to forest fires were estimated as the difference between the woody biomass volume in the pre- and post-fire environment.

3.4.1.2. Monetary valuation

The monetary losses in woody biomass volume (Euro) in areas burned at moderate/high or high/very high fire severity were used as an economic indicator of the reduction in timber provision due to forest fires. A market price-based valuation method was applied based on public statistics from the Simplified Forest Product Quotation System (SIMeF) in Portugal, which reports sales of standing woody biomass in public areas managed by the Portuguese Institute for Nature Conservation and Forests (ICNF, 2022). Minimum, average, and maximum prices for standing woody biomass (Euro/m3) of maritime pine, eucalyptus and oak forests in Northern Portugal were collected and adjusted for inflation to 2017 euros. Then, prices were multiplied by woody biomass volume (m3) previously estimated for pre-fire and post-fire environments. Finally, monetary losses in woody biomass volume due to forest fires were calculated as the difference between pre- and post-fire values.

3.4.2. Cork provision reduction

3.4.2.1. Biophysical quantification

Cork production losses (kg) in areas burned at moderate/high or high/very high fire severity were used as a biophysical indicator of reduced cork provision due to forest fires. Biophysical estimates followed three main steps. In step 1, cork potentially harvestable (pre-fire) was estimated based on data from the Portuguese National Forest Inventory - IFN6 in 2015 (ICNF 2015). First, data on cork oak productivity (kg/ha/yr) in both pure and mixed cork stands were collected. Then, data were spatialised by combining IFN data with land use and land cover spatial data (Section 3.2 – Land cover data). Projections for 2017 assumed constant cork oak productivity. In step 2, the impact of forest fires on cork oak production was assessed. Spatial data on cork oak production obtained in Step 1 was overlapped with spatial data on fire perimeters and their respective fire severity classes (Section 3.3 - Burned area and severity data). A damage factor was applied according to fire severity classes, assuming that the potentially harvestable cork would be lost whenever fires burned at moderate/high or high/very high fire severity. In step 3, cork losses due to forest fires were estimated as the difference between cork production in the pre- and post-fire environment.

3.4.2.2. Monetary valuation

The monetary losses in cork production (Euro) in areas burned at moderate/high or high/very high severity were used as an economic indicator of the reduction in cork provision due to forest fires. A market price-based valuation method was applied based on public statistics from the Simplified Forest Product Quotation System (SIMeF) in Portugal, which reports sales of non-woody products by private producers (ICNF, 2024). Minimum (95 Euro/kg), average (366 Euro/Kg) and maximum (583 Euro/Kg) cork prices in Northern Portugal were collected. These prices were adjusted for inflation to 2017 euros. Then, prices were multiplied by cork production (kg) previously estimated in pre-fire and post-fire environments. Finally, monetary losses in cork production due to forest fires were calculated as the difference between pre- and post-fire values.

3.4.3. Carbon storage reduction

3.4.3.1. Biophysical quantification

Carbon stock losses (Mg) in areas burned at different severity levels were used as a biophysical indicator of reduced carbon storage capacity due to forest fires. Biophysical estimates followed four main steps. In step 1, biomass and carbon stocks in forests and shrublands (pre-fire) were estimated. First, data on growing volume (i.e., living trees) by forest species and NUTS III region in Northern Portugal was collected from the Portuguese National Forest Inventory - IFN6 in 2015 (ICNF 2015). Next, the growing volume was converted into total biomass by applying biomass conversion and expansion factors (BCEF) for each forest species (APA, 2020). Then, total biomass was stratified into above-ground biomass (AGB), i.e., trunk, branches and leaves, by applying conversion factors (APA, 2018), and below-ground biomass (BGB), i.e., roots, by applying root-shoot ratios (APA, 2020). After that, carbon stocks in each of the biomass components were estimated by applying carbon factors for each forest species (APA, 2020). Additionally, carbon stocks in AGB of the forest understory strata, dead organic matter (DOM) of litter and dead woody biomass, and shrublands (AGB and BGB) were collected from the Portuguese National Forest Inventory - IFN6 (ICNF 2015) and the Portuguese National Inventory Report (APA, 2018; APA, 2022). Finally, biomass and carbon stock estimates were spatialised by combining IFN data with land use and cover spatial data (Section 3.2 – Land cover data). In step 2, biomass and carbon stocks were projected for 2017 by applying annual growth rates obtained from available published data (APA, 2018; APA, 2022; ICNF, 2005).

In step 3, the impact of forest fires on carbon stocks was estimated. First, combustion factors by fire severity classes and carbon pool were collected based on scientific literature and description of fire effects in field plots within the SeverusPT project (Table 4). It was assumed that fine live fuels (i.e., leaves and branches) rather than coarse live fuels (i.e., tree trunks) would contribute to effective losses of trees' AGB carbon stocks since these are the fuels usually consumed by fire (Fernandes and Loureiro, 2013). Then, spatial data on carbon stocks obtained in previous steps were overlapped with spatial data on fire perimeters and respective fire severity classes (Section 3.3 - Burned area and severity data). Finally, losses on carbon stocks were estimated by multiplying carbon stocks by combustion factors. In step 4, losses in carbon stocks due to forest fires were estimated as the difference between carbon pools in the pre- and post-fire environment.

3.4.3.2. Monetary valuation

Monetary valuation of carbon stock reduction (Euro) in areas burned at different severity levels was carried out by applying a unit value transfer method based on the social costs of carbon (SCC) and a market price-based valuation method based on carbon credit prices. Social costs of carbon data were collected from the scientific literature, namely the estimates by Nordhaus (2017) of 16.16 Euro/Mg C (Europe) and 105.31 Euro/Mg C (Global) and estimates by Barrage and Nordhaus (2024) of 213.48 Euro/Mg C (Global). SCC estimates from Nordhaus (2017) and Barrage and Nordhaus (2024) were converted from US dollars to Euro using the average exchange rate of 0.9015 (in 2015) and 0.8931 (in 2019), respectively, and inflation-adjusted to 2017 euros. Carbon credit price data were collected from the Market Climate Trade platform (https://market.climatetrade.com – accessed in October 2024). Minimum

(134.40 Euro/Mg C), average (179.38 Euro/Mg C) and maximum (209.97 Euro/Mg C) carbon credit prices were collected from seven carbon offset forest projects in Spain, as currently there is no carbon market in Portugal. These prices were adjusted for inflation to 2017 euros. Then, SCC and carbon credit prices were multiplied by carbon stocks (Mg) estimated in pre- and post-fire environments and monetary losses in carbon stock reduction due to forest fires were calculated as the difference between them.

Table 4: Combustion factors per fire severity class for modelling carbon stock losses in the Northern Portugal test site. NB: Not burned; VL: Very low fire severity; L: Low fire severity; M: Moderate fire severity; H: High fire severity; VH: Very high fire severity. AGB: above-ground biomass; BGB: Below-ground biomass; DOM: Dead organic matter.

Cover type	**Carbon pool**	**Combustion factor (%) per fire severity class**				**Data sources**
		NB or VL	L-M	M-H	H-VH	
Forest	AGB: tree leaves & branches	0%	38%	55%	81%	Balde et al. (2023)
	AGB: understory	0%	30%	70%	90%	Li et al. (2018)
	BGB: tree roots	0%	0%	20%	50%	SeverusPT project (field plots)
	DOM: litter	20%	54%	89%	100%	SeverusPT project (field plots) Campbell et al. (2007) De Santis and Chuvieco (2009) Li et al. (2018)
	DOM: standing trees	1%	21%	38%	48%	
	DOM: down trees	11%	30%	47%	68%	
	DOM: stumps	11%	30%	47%	68%	
Shrubs	AGB	0%	30%	70%	90%	Li et al. (2018)
	BGB	0%	11%	20%	50%	SeverusPT project (field plots) Fonseca et al. (2022)

3.4.4. Soil retention reduction

3.4.4.1. Biophysical quantification

Losses in soil mass (Mg) in areas burned at varying levels of severity were used as a biophysical indicator of reduced soil retention capacity due to forest fires. Biophysical estimates were based on the application of the Morgan-Morgan-Finney (MMF) soil erosion model (Morgan 2001). The MMF model predicts the soil erosion rate, i.e., the amount of soil loss in a given year (Morgan 2001) based on soil properties, topography, precipitation, and land cover. Soil erosion is modelled considering: 1) the water phase, which determines the energy of the rainfall available to detach soil particles from the soil mass and the volume of runoff; 2) the sediment phase, where rates of soil particle detachment by rainfall and runoff are assessed alongside the transporting capacity of runoff; 3) subsequently, predictions of total particle detachment and transport capacity are compared, and the erosion rate is equated to the lesser of the two rates (Morgan 2001). The MMF has been applied and calibrated for Portugal to predict post-fire soil erosion resulting from fire-induced changes in vegetation according to burn severity levels in shrublands and forests (Parente et al. 2022).

The application of the MMF model to the Northern Portugal test site followed three main steps. In step 1, input data collection and processing were carried out (Table 5) and fire severity data (Section 3.3 - Burned area and severity data) was incorporated into the model under previously calibrated model parameters (Parente et al. 2022). In step 2, mathematical equations corresponding to the different phases of the modelling process were collected from the existing scientific literature on the MMF model (Morgan, 2001; Morgan & Duzant, 2008; Parente et al., 2022), and model outputs were generated. In step 3, model outputs were used to estimate soil retention loss as the difference between 1) the ecosystem's capacity to retain soil in the pre-fire environment, determined by subtracting structural soil erosion (i.e. in the absence of vegetation) and the proportion of soil eroded in the presence of unburned vegetation, and 2) the ecosystem's capacity to retain soil in the post-fire environment, calculated by subtracting structural soil erosion (i.e. in the absence of vegetation) and the proportion of soil eroded in the presence of burned vegetation, according to fire severity levels.

3.4.4.2. Monetary valuation

Monetary valuation of soil retention reduction (Euro) due to forest fires was carried out by applying a unit value transfer method based on 1) soil replacement costs and 2) post-fire soil erosion mitigation costs. Data on soil replacement costs were collected from estimates by Marta-Pedroso et al. (2007) for Portugal. The unit value collected was 5.35 Euro/Mg after adjusting for inflation to 2017 euros. This value was then multiplied by the mass of soil retained in the pre-fire and post-fire environment as estimated from MMF model outputs. The monetary value of soil retention reduction due to forest fires was calculated by the difference between the values of the soil mass retained in the pre-fire and post-fire environments. Data on the costs of post-fire soil erosion mitigation treatments were collected from the meta-analysis conducted by Girona-García et al. (2023). The collected unit value was 867.5 €/Mg (after adjusting for inflation to 2017 euros) and corresponds to the median costs of soil erosion mitigation treatments for forest fires that took place in Portugal. For soil retention losses, a treatment prioritisation threshold was applied, i.e., it was assumed that treatments were carried out in areas where the erosion rate exceeds 1 Mg/ha. Then, the amount of soil mass lost due to fires in these areas was estimated, and this value was multiplied by the estimated costs of post-fire soil erosion treatments to obtain the monetary value for the reduction in soil retention due to forest fires.

Table 5: Parameters, data inputs and sources used to run the MMF model in the Northern Portugal test site.

Parameter	Input	Symbol	Data source
Precipitation	Annual rainfall	R	C3S (2022)
	Days with rain	Rn	C3S (2022)
	Intensity of erosive rain	I	Morgan and Duzant (2008)
Soil	Bulk density	BD	Panagos et al. (2024)
	Soil detachability index	K	Panagos et al. (2014)
	Soil surface cohesion	COH	Hengl et al. (2017) Morgan (2001)
	Effective hydrological depth of soil	EHD	Parente et al. (2022) Vieira et al. (2014)
	Max. Soil moisture storage capacity	MS	Hengl et al. (2017) Morgan (2001) Vieira et al. (2014)
Landform	Slope steepness	S	EEA (2012)
Land cover	Rainfall interception	A	Parente et al. (2022)
	Canopy cover	CC	Parente et al. (2022) Fernández et al. (2010)
	Plant height	PH	Parente et al. (2022) Fernandes (2009) Fernandes (2001)
	Actual to potential evapotranspiration ratio	Et/E0	Fernandez et al. (2010) Vieira et al. (2014)
	Crop cover management	C	Parente et al. (2022) Panagos et al. (2015)
	Ground cover	GC	Parente et al. (2022) Fernandes (2003)

3.4.5. Nature-based tourism reduction

3.4.5.1. Biophysical quantification

Losses of recreation opportunity areas (ha) provided by forest and shrubland ecosystems were used as a biophysical indicator of reduced nature-based tourism potential due to forest fires. The Ecosystem Services Mapping Tool – ESTIMAP (Zulian et al., 2013) was applied by following three main steps to modelling 1) the provision of nature-based recreation opportunities (pre-fire environment), 2) the provision of nature-based recreation opportunities (post-fire environment), and 3) the reduction of nature-based recreation opportunities due to forest fires, as the difference between the provision of nature-based recreation opportunities in the pre- and post-fire environment.

In step 1, the provision of nature-based recreation opportunities in Northern Portugal in the pre-fire environment was modelled. Five main factors were used to generate the recreation opportunity map (RP map, in Fig. 4), namely the degree of naturalness, characteristics of natural areas (e.g., nature protection status, presence of geosites), the presence and proximity to inland waters, to cultural sites (e.g., monuments, religious buildings, archaeological sites), and to tourism infrastructures (e.g., tourist offices, tourism attractions). Then, a proximity map was computed based on distances from urban areas and from roads (P map, Fig. 4). Finally, the recreation opportunity spectrum map (ROS map, Fig. 4) i.e., the reclassification of land cover according to the range of opportunities available and the proximity to the potential users, was derived by combining the recreation provision map (RP) with the proximity map (P). Data used in this step is summarized in Table 6.

Table 6: Data inputs and sources used to run the ESTIMAP model in the Northern Portugal test site.

Input	**Component**	**Source**
Land use and cover	Natural and landscape value of land use	DGT (2018)
Natural conditions	Protection status	ICNF (2018)
	Geological heritage	ICNF (2011)
Infrastructure elements	Infrastructure elements	OpenStreetMap
Cultural elements	Cultural elements	OpenStreetMap
Water courses	Water courses	APA (2006)
Urban settlements	Urban settlements	DGT (2020)
Roads	Roads	OpenStreetMap

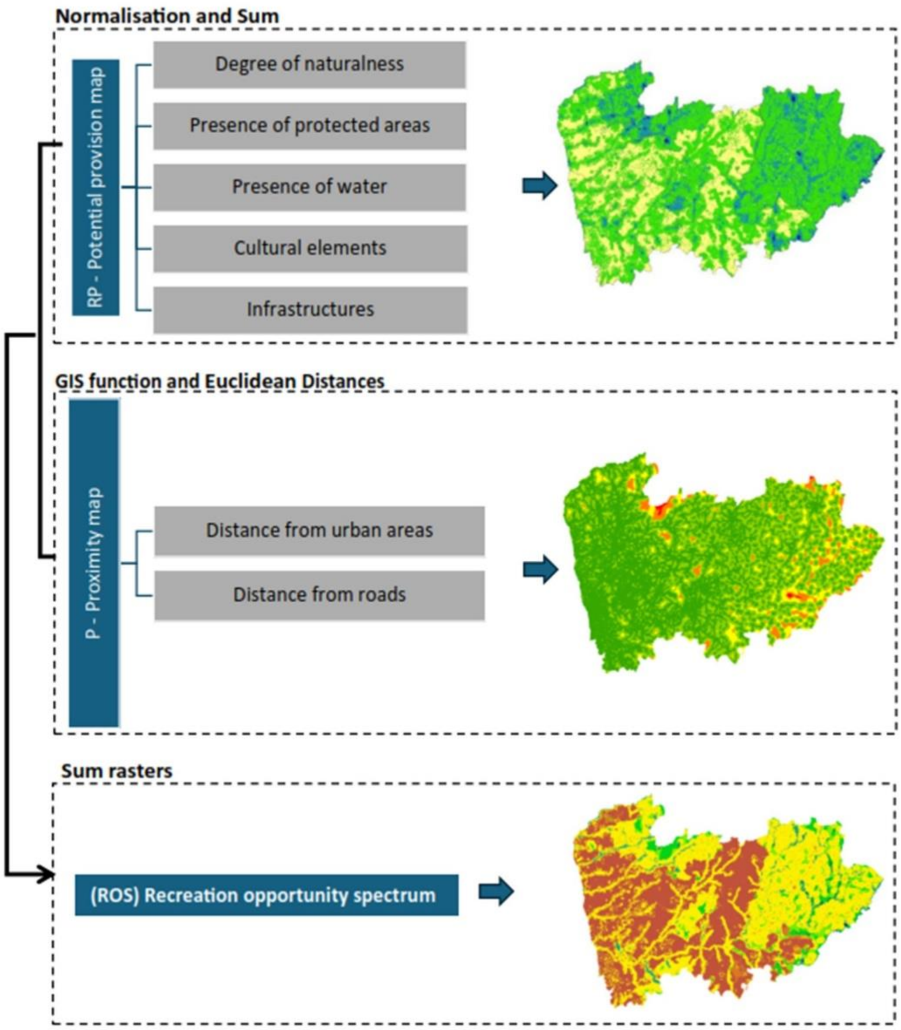

Figure 4: Diagram of the ESTIMAP modelling workflow applied to the Northern Portugal test site.

In step 2, the supply of nature-based recreation opportunities in the post-fire environment was modelled. First, the impact of forest fires that took place in 2017 in Northern Portugal was integrated into the ESTIMAP model. To this end, the naturalness map generated in Step 1 was overlapped with spatial data on fire perimeters and reclassified according to fire severity classes (Section 3.3 – Burned area and severity data). For example, whenever high-severity forest fires impacted areas of high naturalness, the degree of naturalness of those areas decreased to the lowest level. Thus, this approach assumes that the recreation opportunities may shift depending on fire severity. After that, a new recreation provision (RP) map for the post-fire environment was generated following the same process described in Step 1. Then, the recreation opportunity spectrum (ROS) in the post-fire environment, was derived by combining the recreation provision map (RP) impacted by forest fires with the proximity map (P) previously generated in Step 1. In step 3, the loss of nature-based recreation opportunities (ha) due to forest fires was assessed by the difference in areas of high and moderate recreation opportunity classes between pre- and post-fire environments.

3.4.5.2. Monetary valuation

Monetary valuation of nature-based tourism reduction (Euro) due to forest fires was carried out by applying a unit value transfer method based on estimates by (Mendes et al., 2021) for the value of public recreation within and outside protected areas in Portugal. First, unit values were collected, namely 130,7€/ha and 32,2€/ha in forests, and 70,4€/ha and 32,2€/ha in shrublands (after adjusting for inflation to 2017 euros), within and outside protected areas, respectively. Then, the recreation opportunity spectrum (ROS) map in the pre-fire environment was sorted by forest and shrubland areas within and outside protected areas and unit values were assigned. A similar procedure was adopted for the ROS map in the post-fire environment, except that a depreciation factor was applied to the unit value according to fire severity impacts on recreation opportunity areas previously modelled in Step 2 of the biophysical quantification, namely 0.50 for class 2: low to moderate severity, 0.75 for class 3: moderate to high severity, and 0.90 for class 4: high to very high severity. Finally, unit values of public recreation (€/ha) in pre- and post-fire environments were multiplied by the area of each recreation opportunity class. The difference in the monetary value of areas of high and moderate recreation opportunity classes between pre- and post-fire environments was used to estimate the monetary losses of nature-based tourism due to forest fires.

4. Results

4.1. Timber provision reduction

4.1.1. Biophysical quantification

Biophysical estimates of wood biomass in pre-fire and post-fire environments and timber production losses caused by forest fires in 2017 are shown in Table 7. Losses in timber production caused by forest fires corresponded to 65% of timber production in the pre-fire environment, totalling 1,732,488.1 m^3 of wood. The spatial distribution of timber production losses due to forest fires in Northern Portugal is shown in Fig. 5.

Table 7: Timber production (pre- and post-fire environment) and losses due to forest fires in maritime pine, eucalyptus and oak forests in biophysical units (m^3) for the Northern Portugal test site.

	Maritime pine	**Eucalyptus**	**Oaks**	**Total**
Timber pre-fire (m^3)	1,585,824.80	788,871.81	294,995.30	2,669,691.91
Timber loss due to fires (m^3)	1,009,721.85	564,382.97	158,383.28	1,732,488.10
Timber post-fire (m^3)	576,102.95	224,488.84	136,612.01	937,203.81

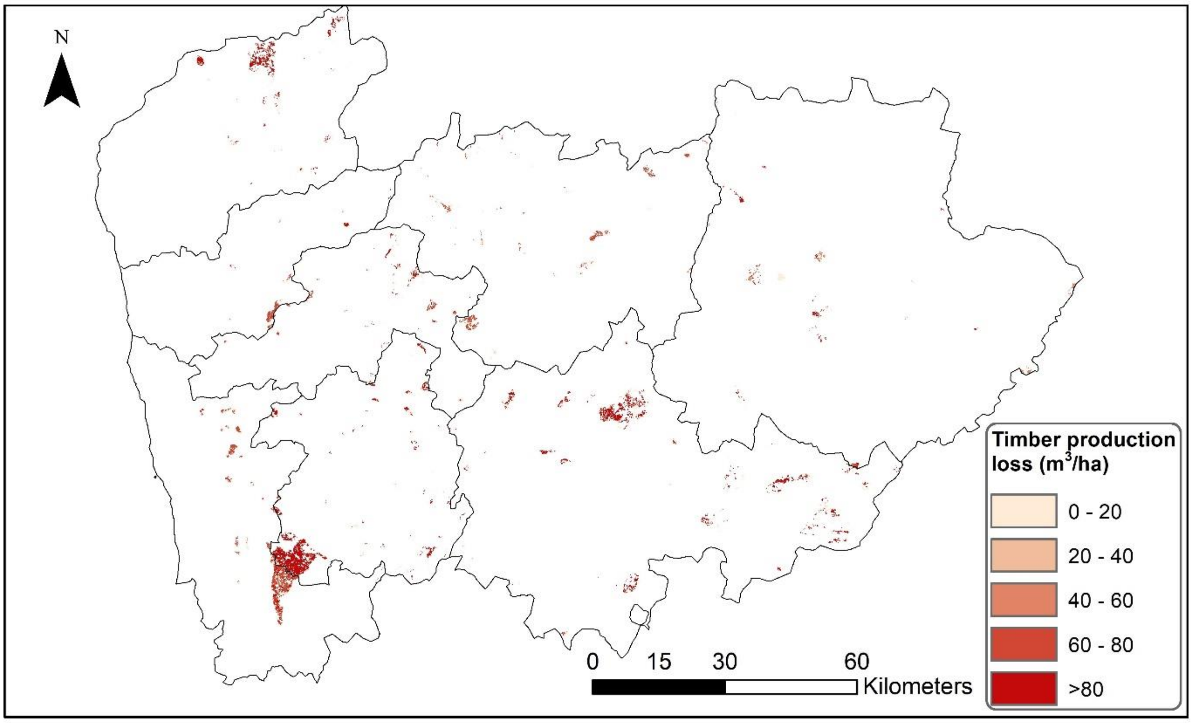

Figure 5: Spatial distribution of timber production losses (m^3/ha) due to forest fires in 2017 in Northern Portugal.

4.1.2. Monetary valuation

Monetary estimates of timber production in pre- and post-fire environments and losses due to forest fires in 2017 are shown in Table 8. Total monetary losses in timber production caused by forest fires ranged from 21.3 to 69.8 million Euros.

Table 8: Timber production (pre- and post-fire environment) and losses due to forest fires in maritime pine, eucalyptus and oaks forests in monetary units (Million Euro) for the Northern Portugal test site.

		Min	**Mean**	**Max**
Maritime pine	Timber pre-fire (Mln €)	12.7	39.7	72.4
	Timber loss due to fires (Mln €)	8.1	25.3	46.1
	Timber post-fire (Mln €)	4.6	14.4	26.3
Eucalyptus	Timber pre-fire (Mln €)	13.7	18.6	25.5
	Timber loss due to fires (Mln €)	9.8	13.3	18.2
	Timber post-fire (Mln €)	3.9	5.3	7.2
Oaks	Timber pre-fire (Mln €)	6.3	7.8	10.3
	Timber loss due to fires (Mln €)	3.4	4.2	5.5
	Timber post-fire (Mln €)	2.9	3.6	4.8
Total	Timber pre-fire (Mln €)	32.7	66.0	108.1
	Timber loss due to fires (Mln €)	21.3	42.7	69.8
	Timber post-fire (Mln €)	11.4	23.3	38.3

4.2. Cork provision reduction

4.2.1. Biophysical quantification

Biophysical estimates of cork production in pre-fire and post-fire environments and cork production losses caused by forest fires in 2017 are shown in Table 9. Losses in cork production caused by forest fires corresponded to 52% of cork production in the pre-fire environment, totalling 88,032.2 kg of cork. The spatial distribution of cork production losses due to forest fires in Northern Portugal is shown in Fig. 6.

Table 9: Cork production (pre- and post-fire environment) and losses due to forest fires in cork oak areas in biophysical units (kg) for the Northern Portugal test site.

	Total
Cork production pre-fire (kg)	168,580.6
Cork production loss due to fires (kg)	88,032.2
Cork production post-fire (kg)	80,548.5

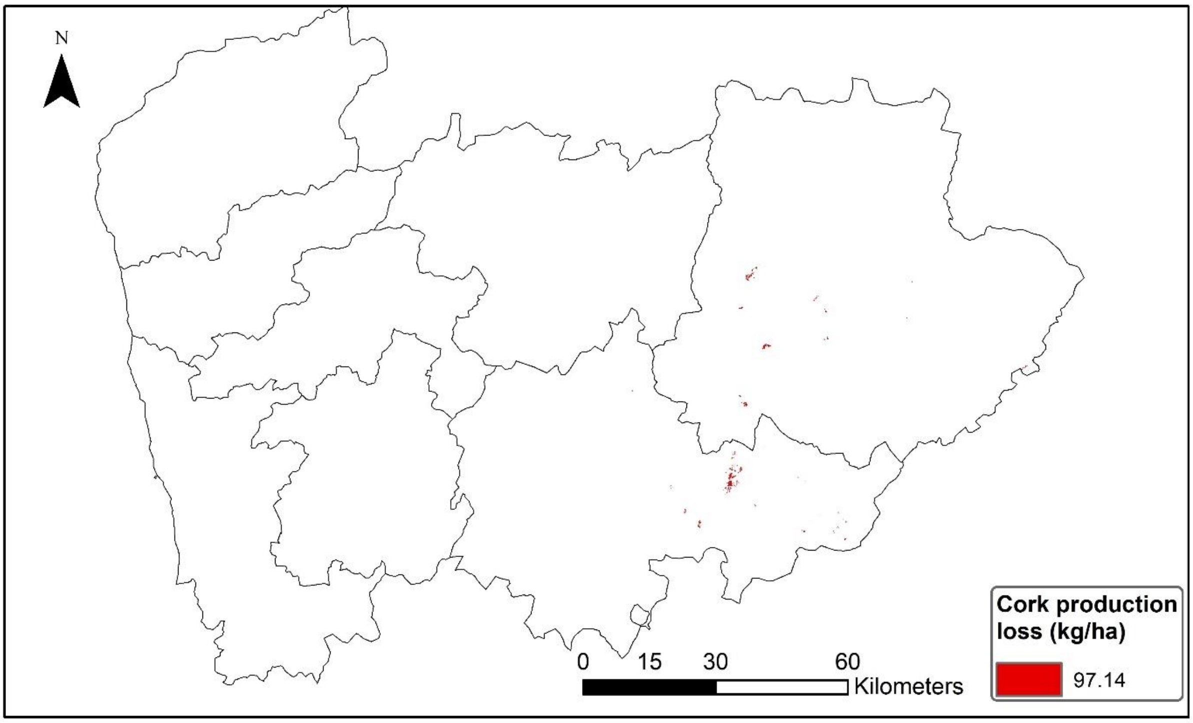

Figure 6: Spatial distribution of cork production losses (kg/ha) due to forest fires in 2017 in Northern Portugal.

4.2.2. Monetary valuation

Monetary estimates of cork production in pre- and post-fire environments and losses due to forest fires in 2017 are shown in Table 10. Monetary losses in cork production caused by forest fires ranged from 8.4 to 51.3 million Euros.

Table 10: Cork production (pre- and post-fire environment) and losses due to forest fires in cork oak forests in monetary units (Million Euro) for the Northern Portugal test site.

	Min.	**Mean**	**Max.**
Cork production pre-fire (Mln €)	16.0	61.7	98.3
Cork production loss due to fires (Mln €)	8.4	32.2	51.3
Cork production post-fire (Mln €)	7.6	29.5	47.0

4.3. Carbon storage reduction

4.3.1. Biophysical quantification

Biophysical estimates of carbon stored in pre- and post-fire environments and carbon losses caused by forest fires in 2017 are shown in Table 11. Losses in carbon stored caused by forest fires corresponded to 17% of carbon stored in the pre-fire environment, totalling 308,662.7 Mg of carbon. The spatial distribution of carbon losses due to forest fires in Northern Portugal is shown in Fig. 7.

Table 11: Carbon stored (pre- and post-fire environment) and losses due to forest fires in forest and shrubland areas in biophysical units (Mg) for the Northern Portugal test site.

	Forest	**Shrublands**	**Total**
Carbon stored pre-fire (Mg)	1,320,847.2	545,507.8	1,866,354.9
Carbon stored loss due to fires (Mg)	182,442.5	126,220.2	308,662.7
Carbon stored post-fire (Mg)	1,138,404.7	419,287.6	1,557,692.3

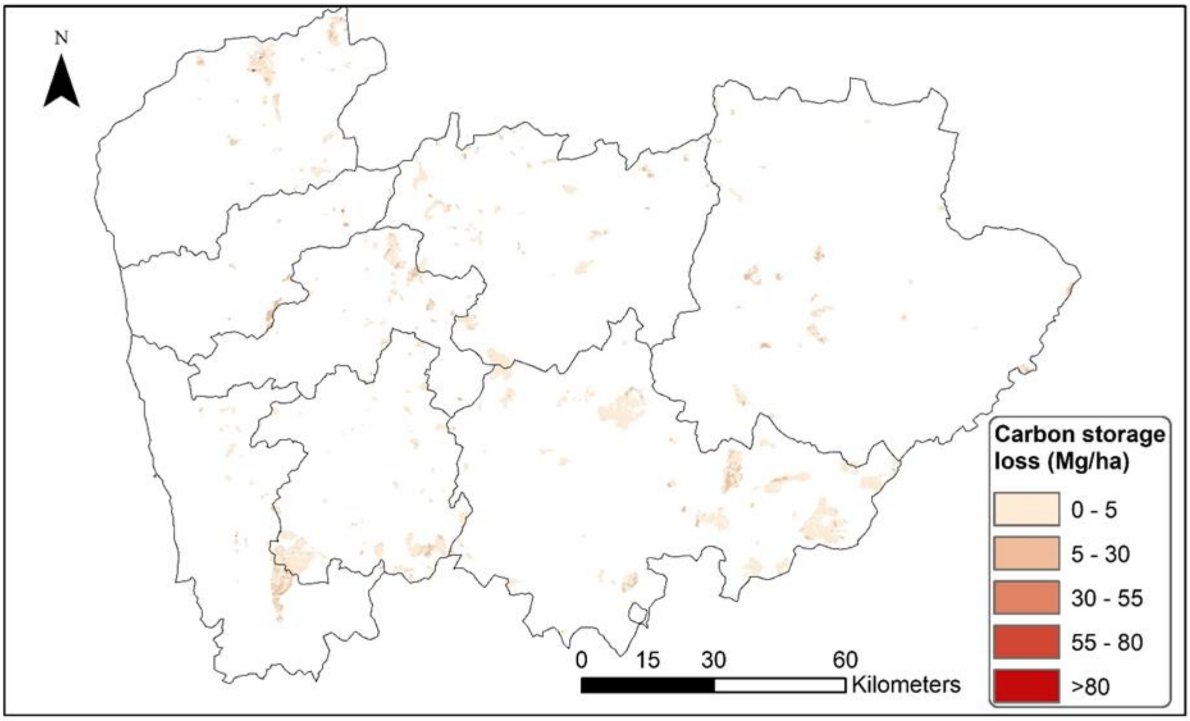

Figure 7: Spatial distribution of carbon losses (Mg/ha) due to forest fires in 2017 in Northern Portugal.

4.3.2. Monetary valuation

Monetary estimates of carbon stored in pre- and post-fire environments and losses due to forest fires in 2017 are shown in Table 12. Monetary losses in carbon stored caused by forest fires ranged from 5.0 to 65.9 million Euros based on Social Costs of Carbon (SCC) value and from 41.5 to 64.8 million euros based on carbon market prices.

Table 12: Carbon stored (pre- and post-fire environment) and losses due to forest fires in forest and shrubland areas in monetary units (Million Euro) for the Northern Portugal test site.

		SCC			Market prices		
		Value*	Value**	Value***	Min.	Mean	Max.
Forest	Carbon stored pre-fire (Mln €)	21.3	139.1	282.0	177.5	236.9	277.3
	Carbon stored loss due to fires (Mln €)	2.9	19.2	38.9	24.5	32.7	38.3
	Carbon stored post-fire (Mln €)	18.4	119.9	243.0	153.0	204.2	239.0
Shrublands	Carbon stored pre-fire (Mln €)	8.8	57.4	116.5	73.3	97.9	114.5
	Carbon stored loss due to fires (Mln €)	2.0	13.3	26.9	17.0	22.6	26.5
	Carbon stored post-fire (Mln €)	6.8	44.2	89.5	56.4	75.2	88.0
Total	Carbon stored pre-fire (Mln €)	30.2	196.5	398.4	250.8	334.8	391.9
	Carbon stored loss due to fires (Mln €)	5	32.5	65.9	41.5	55.4	64.8
	Carbon stored post-fire (Mln €)	25.2	164	332.5	209.4	279.4	327.1

Source: *Nordhaus (2017) – Europe estimates; **Nordhaus (2017) – Global estimates; ***Barrage & Nordhaus (2024) – Global estimates.

4.4. Soil retention reduction

4.4.1. Biophysical quantification

Biophysical estimates of soil mass in pre- and post-fire environments and soil mass losses caused by forest fires in 2017 are shown in Table 13. Losses in soil mass retention caused by forest fires corresponded to 9% of soil mass retained in the pre-fire environment, totalling 333,206.7 Mg of soil. The spatial distribution of soil mass losses due to forest fires in Northern Portugal is shown in Fig. 8.

Table 13: Soil mass retained (pre- and post-fire environment) and losses due to forest fires in forest and shrubland areas in biophysical units (Mg) for the Northern Portugal test site.

	Forest	Shrublands	Total
Soil mass retained pre-fire (Mg)	2,549,174.7	1,199,611.6	3,748,786.4
Soil mass retained loss due to fires (Mg)	226,580.6	106,626.2	333,206.7
Soil mass retained post-fire (Mg)	2,322,594.1	1,092,985.5	3,415,579.6

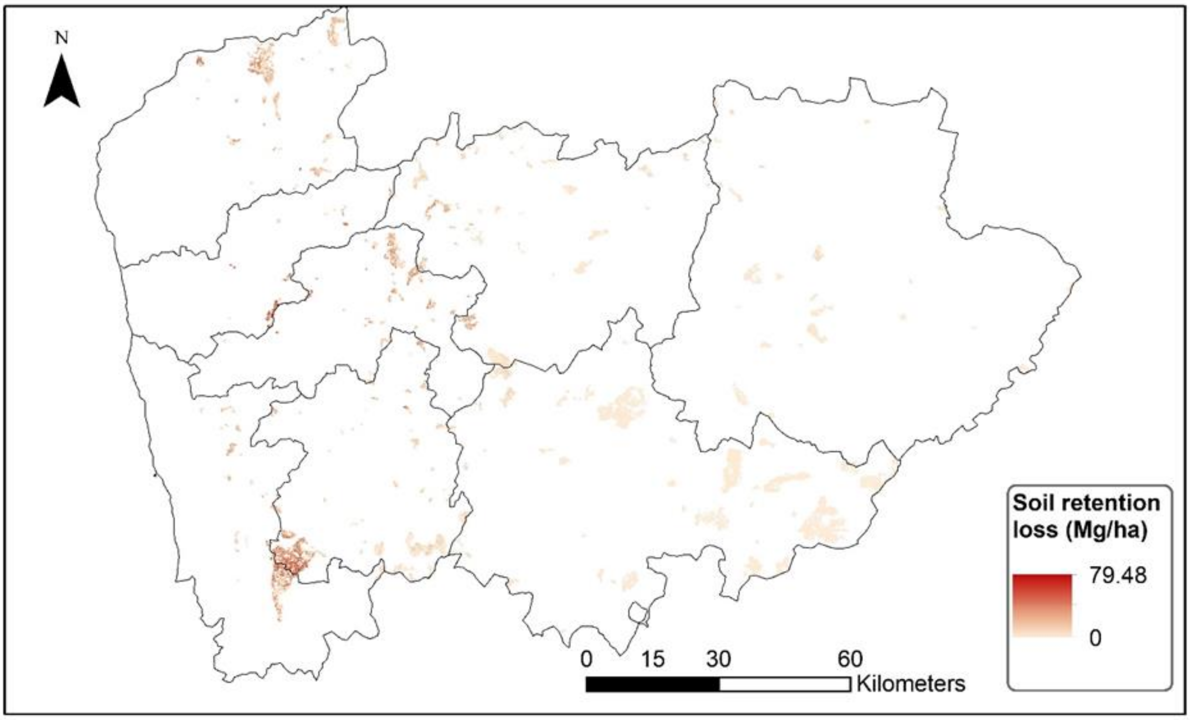

Figure 8: Spatial distribution of soil mass losses (Mg/ha) due to forest fires in 2017 in Northern Portugal.

4.4.2. Monetary valuation

Monetary estimates of soil mass retained in pre- and post-fire environments and losses due to forest fires in 2017 are shown in Table 14. Monetary losses in soil mass caused by forest fires were 1.8 million euros based on soil replacement costs and ranged from 124.4 to 653.6 million Euros based on soil erosion mitigation treatment costs.

Table 14: Soil mass retained (pre- and post-fire environment) and losses due to forest fires in forest and shrubland areas in monetary units (Million Euro) for the Northern Portugal test site.

		Soil replacement costs	Soil erosion mitigation costs		
			Min.	Mean	Max.
Forest	Soil mass retained pre-fire (Mln €)	13.6	972.0	2,211.4	5,106.6
	Soil mass retained loss due to fires (Mln €)	1.2	84.6	192.5	444.4
	Soil mass retained post-fire (Mln €)	12.4	885.6	2,014.8	4,652.7
Shrublands	Soil mass retained pre-fire (Mln €)	6.4	457.4	1,040.7	2,403.1
	Soil mass retained loss due to fires (Mln €)	0.6	39.8	90.6	209.1
	Soil mass retained post-fire (Mln €)	5.8	416.7	948.2	2,189.5
Total	Soil mass retained pre-fire (Mln €)	20.1	1,429.4	3,252.0	7,509.7
	Soil mass retained loss due to fires (Mln €)	1.8	124.4	283	653.6
	Soil mass retained post-fire (Mln €)	18.3	1,302.3	2,963.0	6,842.2

4.5. Nature-based tourism reduction

4.5.1. Biophysical quantification

Biophysical estimates of recreation opportunity areas in pre- and post-fire environments and losses in recreation opportunity areas caused by forest fires in 2017 are shown in Table 15. Losses in recreation opportunity areas caused by forest fires corresponded to 94% of recreation opportunity areas in the pre-fire environment, totalling 3,8575.4 ha. The spatial distribution of losses of recreation opportunity areas due to forest fires in Northern Portugal is shown in Fig. 9.

Table 15: Recreation opportunity areas (pre- and post-fire environment) and losses due to forest fires in forest and shrubland areas in biophysical units (ha) for the Northern Portugal test site.

	Forest	Shrublands	Total
Recreation opportunity pre-fire (ha)	17753.5	23265.5	41019.0
Recreation opportunity due to forest fire (ha)	16203.8	22371.6	38575.4
Recreation opportunity post-fire (ha)	1549.7	893.9	2443.6

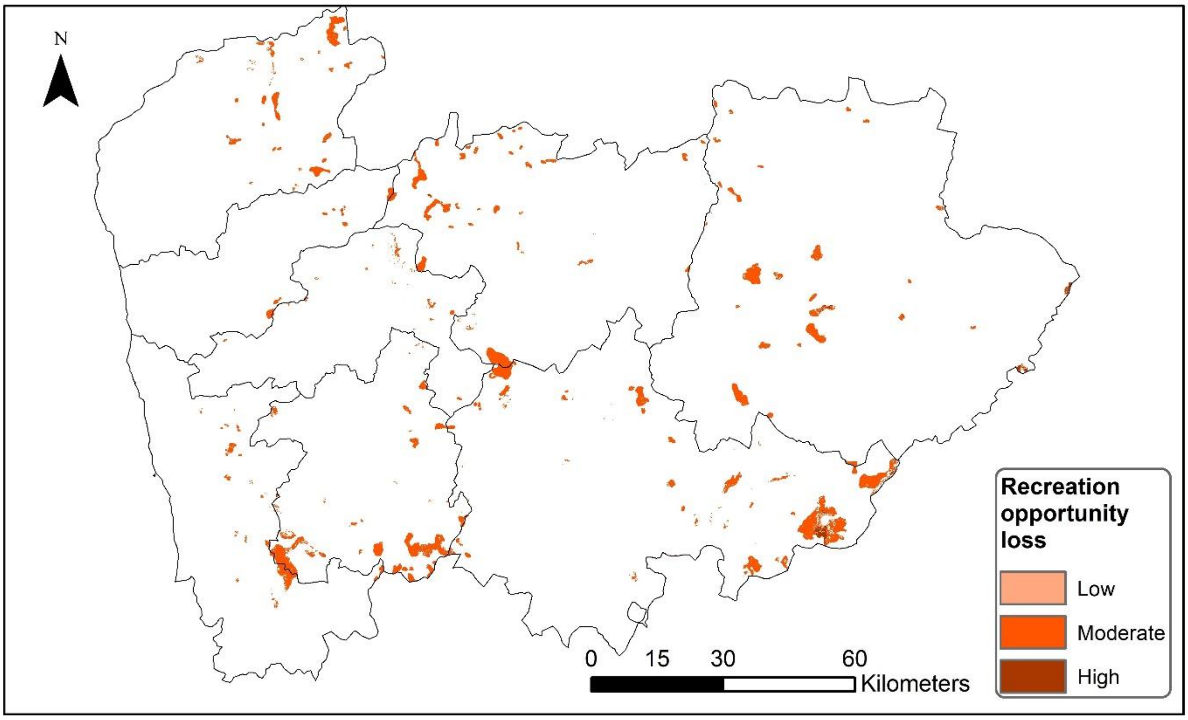

Figure 9: Spatial distribution of losses in recreation opportunity areas due to forest fires in 2017 in Northern Portugal. High losses: transition from high to low/reduced Recreation Opportunity Spectrum (ROS) classes; Moderate losses: transition from moderate to low/reduced ROS classes; Low losses: transition from high to moderate ROS, between pre- and post-fire environments.

4.5.2. Monetary valuation

Monetary estimates of recreation opportunity areas in pre- and post-fire environments and losses due to forest fires in 2017 are shown in Table 17. Monetary losses in recreation opportunity areas caused by forest fires were 2.82 million Euros.

Table 17: Recreation opportunity areas (pre- and post-fire environment) and losses due to forest fires in forest and shrubland areas in monetary units (Million Euro) for the Northern Portugal test site.

	Forest	Shrublands	Total
Recreation opportunity pre-fire (Mln€)	1.38	1.57	2.95
Recreation opportunity loss due to fire (Mln€)	1.29	1.54	2.82
Recreation opportunity post-fire (Mln€)	0.09	0.03	0.12

5. Integration in ecosystem accounts

This section aims to integrate negative externalities in SEEA-EA supply tables following the approach proposed in SELINA Deliverable D5.5 (Figure 1). It specifically addresses the decrease in ecosystem services supply as negative externalities caused by forest fires in the Northern Portugal test site, which primarily result from poor practices and socioeconomic changes in the forestry and agricultural sectors. The following ecosystem accounting tables incorporate negative externalities from forest fires in both biophysical and monetary units related to biomass production: timber provision (Tables 18 and 19) and cork provision (Tables 20 and 21); global climate regulation: carbon storage (Tables 22 to 24); soil and sediment retention: soil retention (Tables 25 to 27); and recreation-related services: nature-based tourism (Tables 28 and 29).

5.1. Timber provision

Table 18: Ecosystem accounting table in biophysical units, including timber provision (gross and net) and forest fires impacts (externality).

	Unit	**Maritime pine**	**Eucalyptus**	**Oaks**
Gross service	m^3	1,585,824.8	788,871.8	294,995.3
Forest fires externality	m^3	1,009,721.8	564,383.0	158,383.3
Net service	m^3	576,103.0	224,488.8	136,612.0

Table 19: Ecosystem accounting table in monetary units, including timber provision (gross and net) and forest fires impact (externality).

	Unit	**Maritime pine**			**Eucalyptus**			**Oaks**		
		Min	Mean	Max	Min	Mean	Max	Min	Mean	Max
Gross service	Mln €	12.7	39.7	72.4	13.7	18.6	25.5	6.3	7.8	10.3
Forest fires externality	Mln €	8.1	25.3	46.1	9.8	13.3	18.2	3.4	4.2	5.5
Net service	Mln €	4.6	14.4	26.3	3.9	5.3	7.2	2.9	3.6	4.8

5.2. Cork provision

Table 20: Ecosystem accounting table in biophysical units, including cork provision (gross and net) and forest fires impact (externality).

	Unit	Cork oak
Gross service	kg	168,580.6
Forest fires externality	kg	88,032.2
Net service	kg	80,548.5

Table 21: Ecosystem accounting table in monetary units, including cork provision (gross and net) and forest fires impact (externality).

	Unit	Cork oak		
		Min	Mean	Max
Gross service	Mln €	16.0	61.7	98.3
Forest fires externality	Mln €	8.4	32.2	51.3
Net service	Mln €	7.6	29.5	47.0

5.3. Carbon storage

Table 22: Ecosystem accounting table in biophysical units, including carbon storage (gross and net) and forest fires impact (externality).

	Unit	Forest	Shrublands
Gross service	Mg	1,320,847.2	545,507.8
Forest fires externality	Mg	182,442.5	126,220.2
Net service	Mg	1,138,404.7	419,287.6

Table 23: Ecosystem accounting table in monetary units (based on SCC – social cost of carbon), including carbon storage (gross and net) and forest fires impact (externality).

	Unit	Forest			Shrublands		
		Value*	Value**	Value***	Value*	Value**	Value***
Gross service	Mln €	21.3	139.1	282.0	8.8	57.4	116.5
Forest fires externality	Mln €	2.9	19.2	38.9	2.0	13.3	26.9
Net service	Mln €	18.4	119.9	243.0	6.8	44.2	89.5

Source: *Nordhaus (2017) – Europe estimates; **Nordhaus (2017) – Global estimates; ***Barrage & Nordhaus (2024) – Global estimates.

Table 24: Ecosystem accounting table in monetary units (based on carbon market prices), including carbon storage (gross and net) and forest fires impact (externality).

	Unit	Forest			Shrublands		
		Min	Mean	Max	Min	Mean	Max
Gross service	Mln €	177.5	236.9	277.3	73.3	97.9	114.5
Forest fires externality	Mln €	24.5	32.7	38.3	17.0	22.6	26.5
Net service	Mln €	153.0	204.2	239.0	56.4	75.2	88.0

5.4. Soil retention

Table 25: Ecosystem accounting table in biophysical units, including soil retention (gross and net) and forest fires impact (externality).

	Unit	Forest	Shrublands
Gross service	Mg	2,549,174.7	1,199,611.6
Forest fires externality	Mg	226,580.6	106,626.2
Net service	Mg	2,322,594.1	1,092,985.5

Table 26: Ecosystem accounting table in monetary units (based on soil replacement costs), including soil retention (gross and net) and forest fires impact (externality).

	Unit	Forest	Shrublands
Gross service	Mln €	13.6	6.4
Forest fires externality	Mln €	1.2	0.6
Net service	Mln €	12.4	5.8

Table 27: Ecosystem accounting table in monetary units (based on soil erosion mitigation costs), including soil retention (gross and net) and forest fires impact (externality).

	Unit	Forest			Shrublands		
		Min	Mean	Max	Min	Mean	Max
Gross service	Mln €	972.0	2,211.4	5,106.6	457.4	1,040.7	2,403.1
Forest fires externality	Mln €	84.6	192.5	444.4	39.8	90.6	209.1
Net service	Mln €	885.6	2,014.8	4,652.7	416.7	948.2	2,189.5

5.5. Nature-based tourism reduction

Table 28: Ecosystem accounting table in biophysical units, including recreation opportunity areas (gross and net) and forest fires impact (externality).

	Unit	Forest	Shrublands
Gross service	ha	17,753.5	23,265.5
Forest fires externality	ha	16,203.8	22,371.6
Net service	ha	1,549.7	893.9

Table 29: Ecosystem accounting table in monetary units, including recreation opportunity areas (gross and net) and forest fires impact (externality).

	Unit	Forest	Shrublands
Gross service	Mln €	1.38	1.57
Forest fires externality	Mln €	1.29	1.54
Net service	Mln €	0.09	0.03

6. Discussion

Applying the analytical framework developed at the Northern Portugal test site allowed for the identification of strengths, weaknesses, research needs and potential relevance for policy development and implementation.

Strengths

- Comprehensive and straightforward methodology to assess negative externalities due to effects of forest fires on ecosystem service supply, both in biophysical and monetary terms, thus allowing integration into ecosystem accounting frameworks.
- Assessment based on national and international public databases that facilitate data collection and modelling processes.
- Integrating spatially explicit data on observed burned area and severity enhanced the modelling of negative externalities due to forest fires.
- An analytical framework suitable to support the development and implementation of national and international environmental policies (e.g., integrated fire management, nature restoration plans, provision and payment for ecosystem services schemes).

Weaknesses

- Potential uncertainties in biophysical modelling due to data collection from multiple sources or the adoption of modelling assumptions due to limited or lack of spatial and temporal data or field measurements covering the test site.
- Potential uncertainties in the monetary valuation of provisioning ES due to likely inter-annual fluctuation of forest products market prices.
- Potential uncertainties in the monetary valuation of regulating and cultural ES due to the application of value transfer methods, which may not account for all factors determining their value in the test site.

Research needs

- Explore the suitability of remote sensing products to overcome data limitations related to spatial, temporal or thematic resolution or spatial and temporal extent.
- Integrate post-fire ecosystem recovery analysis to enhance the assessment of medium- and long-term impacts of forest fires.

- Assess the usefulness and suitability of different economic and sociocultural-based methods to improve understanding of the impacts of forest fires on ES.
- Expand the analysis of negative externalities to other ES and include the impacts of forest fires on human health.
- Examine the socio-ecological impacts of forest fires under alternative land management and climate change scenarios.

Policy relevance:

The application of this analytical framework is relevant to integrating natural capital into decision-making and implementing European environmental policies aimed at addressing the challenges posed by ongoing global change, such as restoration of degraded or vulnerable ecosystems - Nature Restoration Law (Regulation (EU) 2024/1991)) to achieve more fire-resilient ecosystems and landscapes and prevent catastrophic forest fires that put people and property at risk (Lelouvier et al., 2021). Moreover, it can assist in the operationalisation of planning and land management instruments at the national level, such as the National Plan for Integrated Rural Fires Management (Decree-Law No. 82/2021) or the Landscape Transformation Programme (Resolution of the Council of Ministers No. 49/2020) by fostering fire-smart strategies aimed at preventing and mitigating socio-ecological impact of forest fires in vulnerable territories (Regos et al., 2023; Sil et al., 2024).

Also, it can contribute to the development of national environmental policy, for example, the National Strategy of Nature Conservation and Biodiversity for 2030 (Resolution of the Council of Ministers No. 55/2018), through the operationalisation of strategic axes fostering the provision and payment for ecosystem services schemes, by supporting the assessment of services provided by ecosystems or supporting the design of sustainable management strategies (e.g., certification of forest products or preventive land-based fire management by land-owners) aimed at enhancing the condition of ecosystems and mitigating the impacts of forest fires, thereby increasing the capacity to provide ES and generate positive externalities (Fernandes and Simões, 2024; Santos et al., 2019).

In addition, it can contribute to the development of the National Restoration Plans (Regulation (EU) 2024/1991) by promoting post-fire research aimed at ensuring the reestablishment of ecological functions essential for protecting nature and humans against the harmful impacts of forest fires (Gonçalves et al., 2025). Furthermore, it can contribute to filling the gap in evaluating the socio-ecological impacts of forest fires (Pacheco and Claro, 2023), by conveying important information to the Portuguese government and regional/local administration beyond the conventional analysis of forest fire impacts focused on fire suppression costs and losses of human infrastructure (Mavsar, 2009; Poduška and Stajić, 2024). Besides, it can be relevant in overcoming the lack of ecosystem accounting frameworks in Portugal and paving the way for their development and implementation (Lange et al., 2022).

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Annex 4: Negative externalities related to forest fires and intensive agriculture in Peloponnese, Greece

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1. Preface

This report is an Annex of Deliverable D5.5 Final report: Specifying and testing how externalities and EDS can be included in ecosystem accounts. It presents in detail the analyses that were carried out in the test site Peloponnese (Greece) for the purpose of Task 5.1.

2. Introduction

2.1. Site description

The Test Site of Peloponnese (**Figure 1**) lies in Western Greece and includes the Prefectures (now Regional Units of the Region of Western Greece) of Achaia and Ilia. Achaia and Ilia are comprised by a rich natural and cultural environment, that supports a growing tourism sector, however still being not overcrowded in the high season (summer months). Achaia is a mix of urban centres (including Patras, the third largest city of Greece), high mountains (with many peaks above 2000 m), and an extensive coastline. Ilia is the homeland of the Olympic Games and of iconic historical sites such as Ancient Olympia, while simultaneously hosting RAMSAR and Natura 2000 protected areas and vast sandy beaches expanding over many kilometres in the shore of Ionian Sea. Over the past decade, these regions have experienced tourism growth (with visitors from all around the globe), shifts in economic activity, and demographic challenges.

Over the past decade, both Achaia and Ilia have increasingly focused on nature-based recreation to attract visitors beyond traditional cultural and beach tourism. Some of the key developments include:

- Mountain and Ecotourism (Achaia):
  - Kalavryta has emerged as a premier mountain tourism destination, with the Kalavryta Ski Center attracting over 100,000 visitors annually. The ski resort is a major winter draw, while summer activities include hiking, cave exploration (Cave of the Lakes), and religious tourism (Mega Spilaio Monastery).
  - Chelmos-Vouraikos UNESCO Global Geopark, covering 654 km², is a major attraction for geotourism, with unique rock formations, waterfalls, and a historic rack railway (Odontotos) route.
  - Vouraikos Gorge and Mount Erymanthos have become key areas for hiking and trekking tourism, especially among domestic visitors.

- Coastal and Wellness Tourism (Ilia):
    - Kyllini and Kaiafas Thermal Springs: Ilia's thermal spas have seen a rise in wellness tourism, with Kaiafas being recognized for its medicinal properties.
    - Blue Flag Beaches: Ilia has several Blue Flag-certified beaches, including Kourouta, Zacharo, and Kyllini, which attract thousands of Greek and European tourists in the summer.
    - Strofylia and Kotychi Wetlands: Shared among Achaia and Ileia, these wetlands offer excellent opportunities for birdwatching and eco-tourism, particularly in the Strofylia Forest and surrounding lakes and wetland systems.

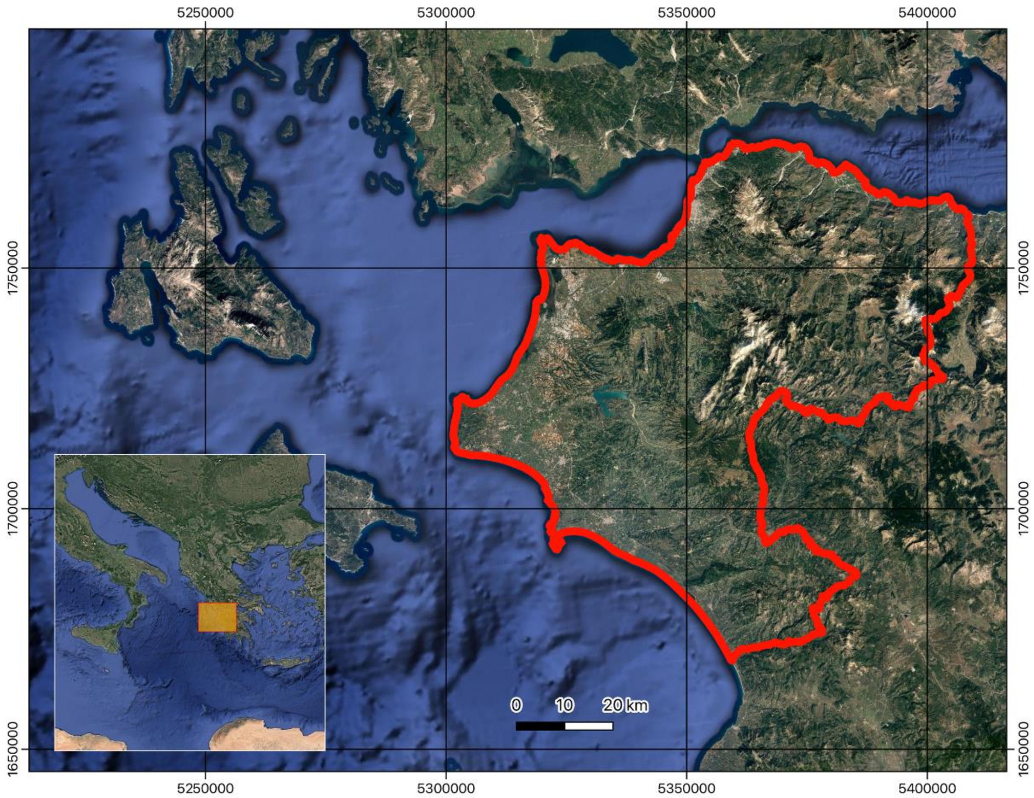

Figure 1: Test Site of Peloponnese, located at the western part of Greece. The red line includes an updated (more extensive) area in Peloponnese, in order to better capture the targeted externalities.

Tourism peaks in summer, but Kalavryta's (Achaia) winter ski tourism helps balance this issue in the region. In 2019, tourism directly contributed ~€200 million to the local economy. Tourism-related sectors (hotels, restaurants, transport) employ approximately 10,000–12,000 people in Achaia and Ilia, accounting for 7%–9% of total employment. The cruise industry in Katakolo (Ileia) alone generates an estimated €25–30 million annually, benefiting local businesses and excursion operators.

Achaia and Ilia have seen population declines, consistent with national trends. The combined population in 2021 was approximately 480,000, down from 520,000 in 2011, a decline of 8%. Patras (Achaia's capital) remains the region's economic and population centre, with 213,000 residents, while Pyrgos (Ilia's capital) has ~25,000 residents. Rural depopulation is a major concern, particularly in Ilia's inland villages, where aging populations and youth outmigration are prominent. Unemployment has been persistently high, though improving post-pandemic. Agriculture remains a dominant sector in Ilia, with tourism acting as a secondary income source for many local residents, while nature- and agricultural- based recreation is a prominent need to be covered, in order to support the locals' income.

Main ecosystem services supplied at the Test Site are food production from cultivations and livestock, recreation in nature, biodiversity conservation, and climate regulation / climate change mitigation by its extensive natural areas.

2.2. Negative externalities and ecosystem disservices

The most important negative externalities and ecosystem disservices in the region are (a) forest fires (natural forest fires are considered as ecosystem disservice, and human induced fires as negative externalities), that affect yearly the Test Site, with small, large or mega-fire events, and are more frequent during the last two decades mainly due to forest management and rural areas' abandonment, combined with human-induced climate change and (b) intensive agricultural practices (negative externality), that have replaced, especially during the last decades, vital natural ecosystems (e.g. coastal lakes, rivers and wetlands). Due to the fact that the region's economy mainly lies on agricultural sector and tourism activity, we selected to assess the impact on nature recreation potential of those two externalities. Moreover, in the region there is an ongoing development shift from intensive agricultural to sustainable tourism activities, focusing on the unique natural characteristics found in the area. These externalities are crucial to be assessed, since the region has been almost totally destroyed by the 2007 mega-fire, that even today affects cultivations, tourism and infrastructure (e.g. due to post-fire flooding, or due to not adequate restoration in many natural areas that affect visitors' preferences and recreation experience).

3. Data and methods

In the Test Site of Peloponnese, we collected all available data from open source platforms, regional, and local agencies in order to identify the externalities present in the region, and to select appropriate data to conduct an accounting process, based on one or more externalities. Forest fires have been included in their spatial and temporal aspect, and intensive agriculture as an existing externality that decreases nature recreation potential. The analysis and mapping procedure was made in the Google Earth Engine (GEE) platform, using the relevant datasets (GEE assets) prepared by UPATRAS team.

3.1. Forest fires

3.1.3 Biophysical quantification

To assess the forest fire impact on the Test Site's ecosystems, we used the available EFFIS layer, including all registered forest fires occurred in the area since 2007, as a vector (.shp) file, that includes the date of each fire event (per fire polygon). This allows to quantify the fire area, as well as the ecosystem types burnt and the time of the fire incident, in order to calculate the potential restoration response time of the different ecosystem types.

3.2. Intensive agriculture

3.2.1 Biophysical quantification

We have considered that only non-intensive and traditional agricultural practice contributes to recreation potential (e.g. for agrotourism), and only those categories are integrated in the model. More precisely, in order to keep a standardised and replicable approach on intensive agriculture identification, we selected the Corine Land Cover (CLC) data set for Greece, and in particular the agriculture classes that are considered as intensive: Permanently irrigated land (CLC class 2.1.2), Rice fields (2.1.3), Annual crops associated with permanent crops (2.4.1). Olive groves (class 2.2.3), even if considered as an intensive agricultural use, were included in our study as non-intensive, since this land use contributes to various agro-tourism projects in the region, while many olive groves are cultivated until the present day in a traditional way (low or no irrigation, no fertilisers etc.).

3.3. Externalities impact on nature recreation potential

The data and steps followed to assess the externalities impact on nature recreation potential in the Test Site were:

- Land Cover Data: Use of Corine land cover data to classify different land types with recreation suitability scores (i.e., natural ecosystems, and traditional cultivations).
- Calculate Proximity to Roads: Higher recreation potential in areas near main road network (paved roads). We imported road network data (OSM) and calculated a proximity buffer (i.e., 1000 meters) using an exponential decay to reduce influence with distance.
- Integrate Natura 2000 Protected Areas and National Parks: Overlay of Natura 2000 and National parks boundaries and application of a 50% boost in recreation potential within these regions.
- Assess Proximity to Blue Flag Beaches: Creation of proximity buffers around Blue Flag beaches, i.e. beaches where a series of stringent environmental, educational, safety, and accessibility criteria must be met and maintained (for details see https://www.blueflag.global/), applying a gradual boost in potential up to 1000 meters.
- Analyse Terrain Suitability (Slope and Elevation): Use of elevation and slope data to identify areas with gentle slopes (<15°) and scenic elevations (300-1000 m).
- Incorporate Archaeological and Cultural Sites: Overlay with archaeological site boundaries and application of a 20% boost in recreation potential.
- Integrate Wildfire Impact with Temporal Recovery: Adjustment of recreation potential based on wildfire impact over time (since 2007), with initial reduction and gradual recovery (i) set recreation potential to its lowest (0.0) in the year of the fire and the following year, (ii) for general areas, increase recreation potential by 5% per year, starting the third year after the fire, (iii) For coniferous forests above 800 m, apply a slower recovery rate of 1% per year, up to a maximum increase of 0.9.

To represent the reduction in recreation potential in terms of intact areas for recreation, the resulting value (0.00-1.00) of the recreation potential raster was applied as a weighting factor to the corresponding areas of each land use class (in terms of mean value).

3.3.1 Monetary valuation

We calculated the monetary price of nature recreation based on the Methodology of estimating the value of forest land in Greece (Albanis et al. 2018), officially adopted by the Greek State. In order to value the forest recreation in Greece, and for various regions of the country, studies have been implemented to assess the value of recreation based on the Travel Cost (TCM) and Contingent Valuation Methodologies (CVM). We should note that exchange values, i.e., values based on actual or imputed market transactions are preferred in the SEEA framework. However, other valuation approaches (e.g., welfare-based, non-market valuation, etc.) are sometimes referenced even if they are not the primary focus in core SEEA EA accounting. In the latest manual (United Nations, 2024) many none exchange valuation methods are described in sections 9.3.5 to 9.3.7. Thus, WTP methods (CVM or CV) as standalone methods or through benefit transfer methodology can be used when exchange values are not feasible.

The annual value of recreation was estimated based on the benefit transfer method, indirectly based on the above mentioned TCM and CVM, according to the formula:

$Vr = Area_r * MARr$,

Where,
Vr is the annual recreational value in €,
$Area_r$ is the area of the forest in which recreation is exercised in ha, and
MARr is the annual recreational value per hectare in €/ha.

Based on a meta-analysis of contingent valuations and travel cost estimates, the average annual recreation value of MARr is 94.30 €/ha/year. This price refers to high and mature forests. It is proposed to reduce this price for other types of forest vegetation. For evergreen broadleaves, a price reduction of 80% to 75.40 €/ha and year, for phryganic ecosystems a reduction of 70% to 66.00 €/ha and year and for grassland areas a price reduction of 50% to 47.20 €/ha and year. Here we present the recreation in nature value for woodland and forest ecosystems.

The calculation of the nature-based recreation value was calculated at the ecosystem type level, using areas for the ecosystem types, as derived from assigning and mapping of CLC classes to ecosystem types (level 2). For the correspondence between CLC classes and ecosystem types, we followed the crosswalk proposed by the European Environment Agency (2023).

4. Results

In this section, the combined externalities impact of intensive agriculture and forest fires is assessed and presented in a thematic map.

4.1. Externalities impact on nature recreation

4.1.1. Biophysical quantification

In **Figure 2,** four indicative maps are presented for the years 2007, 2018, 2022 and 2023, providing the recreation potential by integrating ecosystem recovery rate from the fires occurred since 2007. The result depicts accurately the best areas for recreation in nature, as well as the sites that are affected by forest fires and less attractive for recreation. Blue areas that appear only in the 2022 and 2023 maps, in comparison with the 2007 and 2018 maps, represent well recovered areas after the impact of forest fires, and now considered to have an adequate recreation potential.

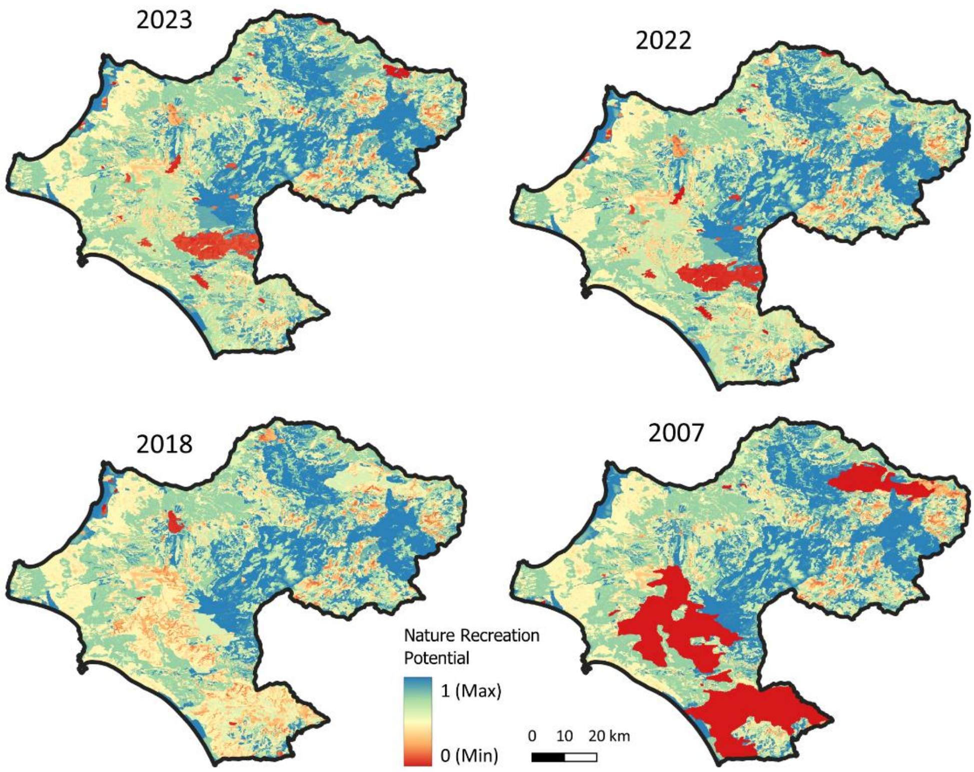

Figure 2: Recreation potential in nature, with integration of the intensive agriculture and forest fire externalities in the Peloponnese study region (for the years 2007, 2018, 2022 and 2023).

4.1.2. Monetary valuation

Based on the aforementioned methodology, in Table 1 the recreation value for each ecosystem type is presented, weighted by the potential recreation factor, as expressed in mean value for the total area of each ecosystem type included in the study area.

Table 1: Monetary valuation of recreation potential in forests and forested areas, affected by forest fires (2007).

Year / Ecosystem types (level 2)	Area (ha)	R-Index	MARr	ΔR-index	Recreation value	Δ-Recreation value
2021						
Broad-leaved forest	9425.42	0.953	94.3	-	846,623.55 €	-
Coniferous forest	37159.42	0.936	94.3	-	3,280,050.75 €	-
Mixed forest	14517.94	0.836	94.3	-	1,144,455.93 €	-
Transitional woodland-shrub	82530.45	0.738	75.4	-	4,592,394.80 €	-
2022						
Broad-leaved forest	9425.42	0.955	94.3	0.00	848,690.84 €	2,067.30 €
Coniferous forest	37159.42	0.935	94.3	0.00	3,275,859.09 €	-4,191.66 €
Mixed forest	14517.94	0.837	94.3	0.002	1,146,550.52 €	2,094.60 €
Transitional woodland-shrub	82530.45	0.749	75.4	0.011	4,660,241.19 €	67,846.40 €
2023						
Broad-leaved forest	9425.42	0.956	94.3	0.002	850,113.76 €	1,422.92 €
Coniferous forest	37159.42	0.936	94.3	0.001	3,280,583.58 €	4,724.49 €
Mixed forest	14517.94	0.844	94.3	0.01	1,155,909.73 €	9,359.21 €
Transitional woodland-shrub	82530.45	0.750	75.4	0.00	4,664,089.40 €	3,848.20 €

5. Integration in ecosystem accounts

5.1. Fire impact on nature recreation

In Table 2, the fire impact on nature recreation is presented for woodland and forest ecosystem types of the region affected by forest fires, using as proxy indicators (a) the mean value of recreation potential from each ecosystem type polygon and (b) the available area for recreation as calculated after applying the recreation potential as a weighting factor.

Table 2: Recreation in nature potential and available (weighted area) for recreation accounting table.

Year / Ecosystem types (level 2)	Area (ha)	R-Index	Area (weighted) for recreation	Δ-Area (weighted) for recreation
2021				
Broad-leaved forest	9425.42	0.953	8977.98	-
Coniferous forest	37159.42	0.936	34783.15	-
Mixed forest	14517.94	0.836	12136.33	-
Transitional woodland-shrub	82530.45	0.738	60907.09	
2022				
Broad-leaved forest	9425.42	0.955	8999.903	21.92254
Coniferous forest	37159.42	0.935	34738.7	-44.4503
Mixed forest	14517.94	0.837	12158.54	22.21208
Transitional woodland-shrub	82530.45	0.749	61806.91	899.8196
2023				
Broad-leaved forest	9425.42	0.956	9014.992	15.08928
Coniferous forest	37159.42	0.936	34788.8	50.10065
Mixed forest	14517.94	0.844	12257.79	99.2493
Transitional woodland-shrub	82530.45	0.750	61857.95	51.03715

This approach provides results for the total area per woodland and forest level 2 ecosystem type of the study area. However, more detailed tables can be extracted for each year only for the areas affected by one or more forest fires, in the given time frame of the study (2007 – 2023).

6. Discussion

The proposed methodology tries to exemplify the integration of externalities in the ecosystem accounting framework, using the decreasing ecosystem service of recreation in nature as a proxy. The strength of this approach is that it provides in a standardised way the integration of the continuously updated EFFIS layer (for the forest fires events) and by this easily document changes (impact) on related ecosystem services (here on recreation potential), based on relevant modeling approaches.

This approach could be also used to capture change in ecosystem condition, based on the post-fire recovery potential of the different ecosystem types. However, this type of analysis is in contrast with the SEEA EA accounting framework, that captures condition changes from externalities via extent, functional, structural and landscape characteristics indicators (that respond to post-fire conditions). By this, a revision of the SEEA EA framework, could be beneficial, if accounting for externalities is to be adopted, as a discrete accounting part. Otherwise, the proposed method could be applied to document ecosystem services change due to externalities and simultaneously support ecosystem condition assessment and accounting in the original SEEA EA accounting framework.

The herein presented example is considered as highly relevant for management and policy decisions for the Test Site, as well as for Greece, since tourism has been severely affected during the last years by smaller or bigger wildfires. The proposed approach provides estimation on how an ecosystem service can recover after the impact of an externality, and by this support efforts and resources for better wild fire prevention, forest and forested areas management and decisions on how to deal with a strategic planning for recreation in nature.

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