Building the foundations of sustainable environmental management: understanding the influence of natural capital on ecosystem services

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Abstract

Global indicators of ecosystem extent and condition have declined by 47%, relative to their earliest estimated states. Natural capital is the worlds' stocks of these natural assets, which supply a wide range of ecosystem services that directly or indirectly produce value for people. This decline in our natural capital is likely to have repercussions for the ecosystem services it supplies. To support and inform sustainable and effective environmental management decisions for the provision of our vital ecosystem services we must fully understand the linkages between them and natural capital. Yet many existing approaches only assess a limited number of ecosystem services and natural capital assets, and therefore miss important synergies and trade-offs. Furthermore, there has been very little exploration into the context dependency of these linkages and the evidence underlying them; natural capital to ecosystem service linkages may be of different relevance to decision makers depending on their desired application.

This thesis follows the creation of the Linking Natural Capital Attribute Groups to Ecosystem Services (LiNCAGES) platform to support collation, exploration and synthesis of evidence on linkages between natural capital and ecosystem services and its communication in environmental decision making. The thesis shows how the LiNCAGES platform allows for the holistic investigation of natural capital and ecosystem service linkages while accounting for the context dependency of a user's decisions. Furthermore, this thesis reveals how accounting for relationships between multiple natural capital attributes can reveal new indirect trade-offs and synergies between ecosystem services and how these are affected by context dependency. Additionally, the thesis highlights the importance of understanding the strengths and limitations of evidence on natural capital that underpins maps of ecosystem service provision, which are frequently used to support environmental decisions. Maps of present and future ecosystem service provision created using evidence from the literature, expert scoring and an existing model were found to vary considerably by the region, ecosystem service, and future scenario mapped.

The work presented in this thesis provides new insight into the complexities and context dependencies in natural capital to ecosystem service linkages and relationships between natural capital attributes, as well as in the evidence used to communicate such linkages via ecosystem service provision maps. This deeper understanding contributes to the support of sustainable and effective environmental management decisions necessary for the preservation of our vital ecosystem services.

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Declaration

I declare that other than where the contribution of others is specified, this thesis is entirely my own work and has not been submitted for the award of any other degree at this or any other university.

George Linney

Statement of authorship for multi-authored chapters

Chapter 3: A visualization platform to analyze contextual links between natural capital and ecosystem services

Linney, G.N., Henrys, P., Blackburn, G.A., Maskell, L.C., Harrison, P.A., 2020. A visualization platform to analyze contextual links between natural capital and ecosystem services. Ecosyst. Serv. 45, 101189. https://doi.org/10.1016/j.ecoser.2020.101189

The concept and methodological approach for this chapter was decided upon through a series of discussions between the authors. As first author, I was solely responsible for all analysis involved in this paper including the development of the LiNCAGES platform. Once the analysis was complete, I produced a draft manuscript, including all figures and tables. This draft was then revised in response to feedback from the other authors.

Chapter 4: Effect of linkages between natural capital attributes on ecosystem service provision

Linney, G.N., Maskell, L.C., Henrys, P., Blackburn, G.A., Harrison, P.A., 2020. Effect of linkages between natural capital attributes on ecosystem service provision. Submitted to Ecosystems.

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Chapter 5: Vive la différence: evidence matters to ecosystem service mapping

Linney, G.N., Dunford-Brown, R., Henrys, P., Blackburn, G.A., Maskell, L.C., Harrison, P.A., 2020. Vive la différence: evidence matters to ecosystem service mapping. Submitted to People and Nature.

The concept and methodological approach for this chapter was decided upon through a series of discussions between the authors. Robert Dunford-Brown created the spreadsheet that breaks down the IAP2 land use classes and links them to CORINE land cover classes. I was responsible for all other analysis including the adaption of this spreadsheet to use the scores developed by (Stoll et al., 2015), the creation of the evidence type maps and their comparisons. Once the analysis was complete, I produced a draft manuscript, including all figures and tables. This draft was then revised in response to feedback from the other authors.

1 Introduction

1.1 Background and context

We are totally dependent upon the natural world (Dasgupta, 2021). It supplies us with our basic life support systems (IPBES, 2019); every oxygen-laden breath we take and every mouthful of food we eat (Dasgupta, 2021). Yet, humanity has been degrading the natural environment at rates far greater than ever before (Dasgupta, 2021; IPBES, 2019). Global indicators of ecosystem extent and condition have declined by 47%, relative to their earliest estimated states (IPBES, 2019). To prevent further degradation of our life support systems we must fully understand and communicate the value of our natural environment. To do this we use the term ecosystem services; the contributions that ecosystems make to human well-being (Haines-Young and Potschin, 2018).

Ecosystem services range from: provisioning services (tangible things that can be exchanged or traded e.g. timber), regulating services (ecosystem outputs that are not consumed but affect people e.g. air quality), and cultural services (non-material ecosystem outputs that have symbolic, cultural or intellectual significance e.g. aesthetic landscapes) (Maes et al., 2013). Ecosystem services are becoming more and more mainstreamed into policy and decision making with the establishment of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services in 2012 (IPBES, 2022).

Ecosystem service assessment involves accounting for all the attributes that contribute to ecosystem service provision including those affecting both supply and demand. This can be framed as a cascade linking the attributes of supply and demand that are important for ecosystem service delivery (Figure 1). For example, the **driver** of habitat restoration increases the **natural capital asset** of streamside habitat area, which promotes water filtration and retention, providing the **ecosystem service** of reduced sediment load in the stream, which provides the **benefit** of clean water. Clean water can be **valued** through avoided treatment costs at water treatment works.

In order to successfully assess ecosystem services we must incorporate all parts of the cascade and also include the synergies and trade-offs between all ecosystem service cascades (Potschin-Young et al., 2018).

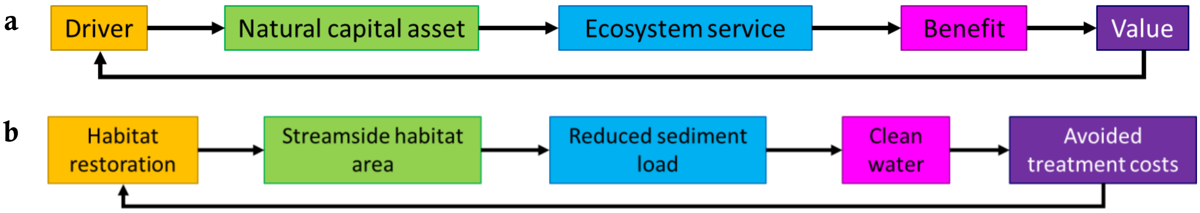


Figure 1: Modified version of the cascade adapted from the cascades model used by Haines-Young and Potschin (2011) *and Harrison et al.* (2017)*, (b) An example cascade for clean water provision.*

There are many tools available to assess ecosystem services, which use a wide range of methods and techniques. Data approaches involve the use of ecosystem service indicators to represent the high complexity of the human-environment system involved in the provision of an ecosystem service (Maes et al., 2016; Müller and Burkhard, 2012). For example, stream macroinvertebrates can be used to represent the ecosystem service of water quality (Maskell et al., 2013). Alternatively, transfer approaches involve taking estimates from one site and applying them to another. For example, land cover types can be assigned expert derived scores based on their ability to provide various ecosystem services e.g. Burkhard et al. (2009). Additionally, many models and modelling platforms exist for ecosystem service assessment with Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) and ARtificial Intelligence for Ecosystem Services (ARIES) being two of the most popular. See Section 2.3 for more details about these modelling platforms.

However, many existing approaches focus on an individual ecosystem service (Gutierrez-Arellano and Mulligan, 2018; Harrison et al., 2014; Lefcheck et al., 2015; Ziter, 2016) and as a result fail to account for important trade-offs and synergies involved in the supply of multiple ecosystem services, whether that be direct trade-offs between ecosystem services or indirect trade-offs between the natural capital assets that are important to the supply of ecosystem services.

Furthermore, the use of ecosystem service indicators relies on the assumption that an indicator can represent the complexity of the whole human-environment system that influences the provision of the ecosystem service (Maes et al., 2016). For example, Harrison et al. (2014) found that the links between biodiversity and the delivery of ecosystem services requires further research and evidence gathering. This leads to limitations in using indicators to assess ecosystem services that are strongly connected to biodiversity, such as the cultural service of bird watching (Maes et al., 2016).

Moreover, trade-offs and synergies, more generally referred to as linkages throughout the remainder of this thesis, between all parts of the cascade are context dependent (Adhikari and Hartemink, 2016; Harrison et al., 2014). This context dependency refers to variation in the presence of such trade-offs and synergies with aspects such as ecosystem type, spatial and temporal scale, geographical location and study method. For example, forests typically only store half as much carbon when in an urban environment compared to when in a more natural environment (Zhao et al., 2010). As a result of this context dependency, some linkages may be more or less relevant to decision makers depending on their context or desired application. This context dependency is overlooked in data transfer methods such as expert elicitation, and is not explored in many ecosystem service assessments (Duncan et al., 2015).

Assumptions, limitations, and subjectivity is also present in the methods and evidence types that are used to investigate ecosystem service provision. Models can be complex and seen as black boxes (a model which produces useful information without revealing information about its internal workings) by decision makers (Harrison et al., 2018; Shoyama et al., 2017; Vorstius and Spray, 2015). Although this may be necessary to account for the complexity of the environment, it does mean that decision makers may not be fully aware or understand the assumptions and limitations underlying the model or method. On the other hand, transfer methods such as expert elicitation can oversimplify the complexities of the human environment system and rely on subjective judgements (Burkhard et al., 2012; Campagne et al., 2020). As a result effective communication of the differences between assessment methods and their assumptions is essential to inform sustainable environmental management (Campagne et al., 2020; Schulp et al., 2014).

1.2 Objectives

Fully accounting for the entire ecosystem service cascade across all ecosystem services and all contexts is not within the scope of this thesis. Therefore, the thesis focuses on the provisioning side of the ecosystem service cascade, specifically the linkage between natural capital and ecosystem services and the associated synergies and trade-offs across ecosystem service cascades. By only focussing on one link in the cascade, but for as many ecosystem services and contexts as possible, this thesis aims to promote the move away from "by service" ecosystem service assessments to a more holistic assessment of multiple ecosystem service provision. Focusing on the link between natural capital and ecosystem services this thesis aims to achieve the following objectives:

- Holistically investigate the link between natural capital and ecosystem services while accounting for context dependency.
- Investigate how accounting for interlinkages across ecosystem service cascades can reveal new indirect natural capital to ecosystem service linkages.
- Compare how the evidence type used for natural capital to ecosystem service linkages can influence maps of ecosystem service provision.
- Show how different evidence types can be combined to further support environmental decision making based off linkages between natural capital and ecosystem services.
- Create a platform to support collation, exploration, and synthesis of evidence on linkages between natural capital and ecosystem services and its communication in environmental decision making and research.

1.3 Thesis organisation and structure

This thesis comprises six chapters (plus appendices). In chapter 1 (this chapter), the aims and structure of the thesis are outlined. Chapter 2 provides an overview of existing literature on the concepts of ecosystem services and natural capital, how they fit within the overall ecosystem service cascade, and current methods and tools for assessing ecosystem service provision. Following this review, three analytical chapters are presented.

Chapter 3 describes the development and application of the LiNCAGES (Linking Natural Capital Attribute Groups to Ecosystem Services) platform. LiNCAGES provides a flexible tool that researchers can use to support the collation, exploration, and synthesis of literature-based evidence for linkages between 42 natural capital attributes and 13 ecosystem services. Through using LiNCAGES to filter and weight the evidence by their specific context or desired application, decision makers can access credible and salient evidence on important natural capital to ecosystem service linkages that occur under their context. An application of the LiNCAGES platform through a hypothetical use case scenario of a small-scale European forest manager is presented. This chapter also shows how the LiNCAGES platform provides a resource for researchers to identify key gaps in the evidence base and to work collaboratively to target and collate additional evidence that can strengthen the foundations of sustainable environmental management.

Chapter 4 builds on Chapter 3 by exploring how accounting for interactions between natural capital attributes may reveal further (indirect) natural capital attribute to ecosystem service linkages. A long-term, large-scale monitoring dataset was used to explore and cluster potential inter-relationships between natural capital attributes. These clusters were then combined with literature-based evidence for natural capital to ecosystem service linkages as identified from the LiNCAGES platform. This revealed new potential indirect positive and negative natural capital to ecosystem service linkages and enabled the identification of new potential trade-offs and synergies between ecosystems services that decision-makers may not be aware of. Chapter 4 also investigates how context, in terms of geographic location, may influence which new potential linkages are revealed. The importance of understanding both the relationship between multiple natural capital attributes and how they are linked to ecosystem services when making management decisions is illustrated through the scenario of an upland farmer wishing to promote soil carbon concentration to improve ecosystem service delivery.

Chapter 5 explores how the complex information underlying ecosystem service provision identified in Chapter 3 and Chapter 4 can be communicated in a spatially explicit manner that is quick and easy for decision makers to understand. Maps of ecosystem service provision were created based upon literature evidence for natural capital to ecosystem service linkages from the LiNCAGES platform, expert scoring evidence from land use to ecosystem service matrices and modelled evidence from an integrated modelling platform. Comparison of the maps for both current conditions and a range of future scenarios allowed disagreements between the evidence type maps to be explored. Chapter 5 highlights the impact that the type of evidence underpinning ecosystem service maps may have on decision making, emphasizing the importance of effectively communicating the assumptions and limitations of the evidence underlying ecosystem service provision maps.

Chapter 6 concludes the thesis, summarising how this thesis has met the objectives set out above in Section 1.2 and the importance of these findings in supporting sustainable environmental management, including the implementation of some of the outputs of this thesis in the LiNCAGES platform. Finally, this chapter outlines the further work required to contribute to the operationalisation of the ecosystem service cascade to support sustainable environmental decision making.

The first of these chapters has been published (see statement of authorship for full reference and authorship details). The second analytical chapter is currently under review with the journal Ecosystems (at the time of writing this thesis) and the final analytical chapter is currently under review with the journal People and Nature (at the time of writing this thesis). An overall reference list, collated from the reference lists for each individual chapter, is provided at the end of this thesis. The appendices provide the supplementary material published alongside each chapter.

2 Literature review

2.1 What are natural capital and ecosystem services

2.1.1 History of the concept of natural capital and ecosystem services

The idea that natural systems provide benefits that support human wellbeing is as old as humans themselves (Costanza et al., 2017). These natural resources that human society draws upon can be defined as natural capital, which includes both Earth's ecosystems and the underpinning geo-physical systems (Figure 2) (Haines-Young and Potschin, 2018; Maes et al., 2013; Smith et al., 2017).

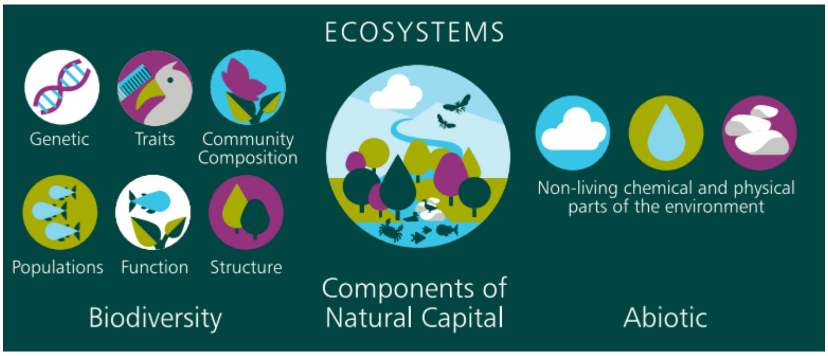


Figure 2: Components of natural capital adapted from Dasgupta (2021).

Natural capital generates a flow of benefits to human society. This flow is known as ecosystem services. The term 'nature's services' first appeared in the academic literature in Westman (1977) and was later developed into 'ecosystem services' by Ehrlich and Ehrlich (1981). In 1997 a paper by Costanza et al. (1997) on valuing the world's ecosystem services, along with an accompanying book by Daily et al. (1997) describing the ecosystem services used in the paper in detail, brought the ecosystem service concept into the research and policy spotlight. Over the past 25 years the ecosystem service concept has been developed and iterated upon, with the number of publications on ecosystem services increasing dramatically (Costanza et al., 2017; Mulder et al., 2015; Wang et al., 2021). It was pushed further into the forefront of the policy agenda with the publishing of the Millennium Ecosystem Assessment (MA) in 2005, an assessment of the consequences of ecosystem change on human wellbeing involving 1,360 experts worldwide (Burkhard and Maes, 2017). Since the publication of the MA, various frameworks and assessments have been developed to improve understanding and evidence related to ecosystem services including The Economics of Ecosystems and Biodiversity (TEEB), Common International Classification of Ecosystem Services (CICES), and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES). This has led to a wide variety of definitions and framings for the concept itself (Box 1), with the most recent rebranding of ecosystem services to "natures contributions to people" first introduced by Diaz et al. (2018) and used in the IPBES (2019) global and regional assessments. However, due to its infancy, and debate surrounding the use of the term "natures contributions to people" e.g. De Groot et al. (2018), the term ecosystem service is used from here on.

Box 1: Definition of ecosystem services over time by different frameworks adapted and updated from Braat and de Groot (Braat and de Groot, 2012).

- Ecosystem Services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life (Daily et al., 1997)
- Ecosystem Services are the benefits human populations derive, directly or indirectly, from ecosystem functions (Costanza et al., 1997)
- **MA**: Ecosystem Services are the benefits people obtain from ecosystems (MA, 2005)
- **TEEB**: Ecosystem Services are the direct and indirect contributions of ecosystems to human well-being (TEEB, 2010)
- **CICES**: Ecosystem services are the contributions that ecosystems make to human well-being, and distinct from the goods and benefits that people subsequently derive from them (Haines-Young and Potschin, 2018)
- **IPBES**: Natures contributions to people are all the contributions, both positive and negative, of living nature (diversity of organisms, ecosystems, and their associated ecological and evolutionary processes) to people's quality of life (Díaz et al., 2018)

2.1.2 Defining and classifying ecosystem services

Final ecosystem services are defined as the outputs of ecosystems that most directly affect the well-being of people. Whether something is regarded as a final ecosystem service depends on the context. For example, if the water in a lake is used directly as a source for drinking, then it could be regarded as a final service. Yet, for the ecosystem service recreational fishing, the fish caught would be regarded as a final service (Haines-Young and Potschin, 2018)

. Therefore, it is important to define final ecosystem services clearly, to avoid the problem of double counting when we calculate their values (Burkhard and Maes, 2017).

CICES uses a hierarchical classification for ecosystem services (Figure 3) that is designed to link back to other ecosystem service classifications (Table 1) (Costanza et al., 2017)). CICES has a five level hierarchical structure (Figure 3) allowing for greater clarification on what ecosystem services are included within each class (Maes et al., 2013).

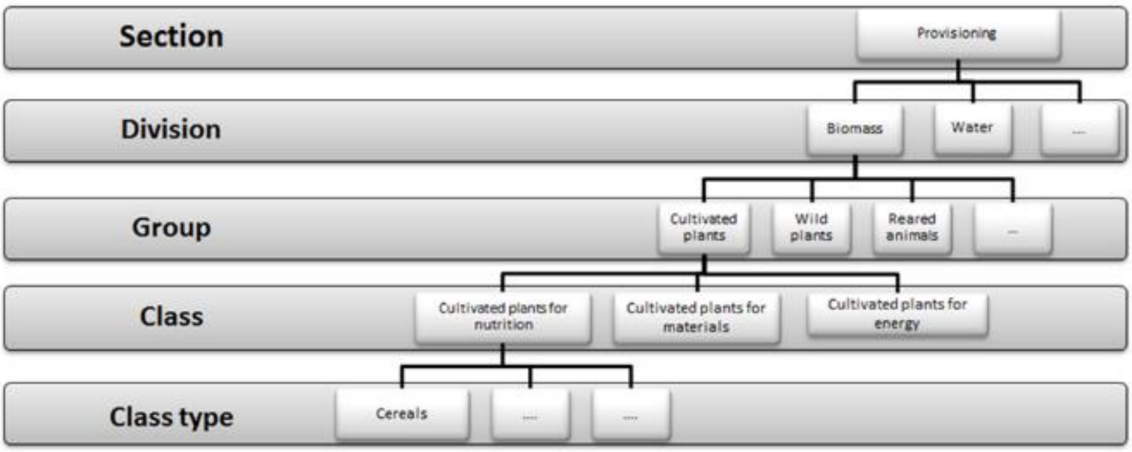


Figure 3: The hierarchical structure of CICES V5.1, illustrated with reference to a provisioning service 'cultivated plants' which at Group level has no end-use associated with it; this category is subsequently disaggregated at class level as 'Cultivated terrestrial plants (including fungi, algae) grown for nutritional purposes' which is defined as 'the ecological contribution to the growth of cultivated, land-based crops that can be harvested and used as raw material for the production of food'. This can be represented as 'cereals' at class type level. Taken from Haines-Young and Potschin (2018).

At the highest level of the classification there are three sections: provisioning; regulating and maintenance; and cultural. Below that in the hierarchy are so-called divisions of ecosystem services of which there are eight unique classes. Provisioning services are tangible things that can be exchanged or traded, as well as consumed or used directly by people in manufacturing (e.g., timber). Regulating and maintenance services include ecosystem outputs that are not consumed but directly affect people (e.g., air quality). Cultural services include all non-material ecosystem outputs that have symbolic, cultural or intellectual significance (e.g. aesthetic landscapes) (Maes et al., 2013).

Table 1 Comparison of four of the main ecosystem services classification systems used worldwide and their differences and similarities adapted from Maes et al. (2013) *and Diaz et al.* (2018).

MA Categories	TEEB Categories	CICES v4.3 group*	Natures Contributions to People categories
			Maintenance of options
Food (fodder)	Food	Biomass [Nutrition]	Food and feed
		Biomass (Materials from plants, algae and animals for agricultural use)	
Fresh water	Water	Water (for drinking purposes) [Nutrition]	Regulation of freshwater quantity, location and timing
		Water (for non-drinking purpose) [Materials]	
Fibre, timber	Raw materials	Biomass (fibres and other materials from plants, algae and animals for direct use and processing)	Materials, companionship and labor
Genetic resources	Genetic resources	Biomass (genetic materials from all biota)	Medicinal, biochemical and genetic resources
Biochemicals	Medicinal resources	Biomass (fibres and other materials from plants, algae and animals for direct use and processing)	
Omamental resources	Omamental resources	Biomass (fibres and other materials from plants)	Materials, companionship and labor
			Maintenance of options
		Algae and animals for direct use and processing	
		Biomass based energy sources	Energy
		Mechanical energy (animal based)	Materials, companionship and labour
Air quality regulation	Air quality regulation	[Mediation of] gaseous/air flows	Regulation of air quality
Water purification and water treatment	Waste treatment (water purification)	[Mediation of] waste, toxics and other nuisances] by biota	Regulation of freshwater and coastal water quality
		[Mediation of] waste, toxics and other nuisances] by biota	
Water regulation	Regulation of water flows	[Mediation of] liquid flows	Regulation of freshwater quantity, location and timing
	Moderation of extreme flows		
Erosion regulation	Erosion prevention	[Mediation of] mass flows	
Climate regulation	Climate regulation	Atmospheric composition and climate regulation	Regulation of climate
			Regulation of ocean acidification
			Regulation of hazards and extreme events
Soil formation (supporting service)	Maintenance of soil fertility	Soil formation and composition	Formation, protection and decontamination of soils and sediments
Pollination	Pollination	Lifecycle maintenance, habitat and gene pool protection	Pollination and dispersal of seeds and other propagules
Pest regulation	Biological control	Pest and disease control	Regulation of detrimental

			organisms and biological processes
Disease regulation			Medicinal, biochemical and genetic resources
Primary production Nutrient cycling (supporting services)	Maintenance of life cycles of migratory species (incl. nursery service)	Life cycle maintenance, habitat and gene pool protection	Habitat creation and maintenance
		Soil formation and composition	
	Maintenance of genetic diversity (especially in gene pool protection)	[Maintenance of] water conditions	
		Lifecycle maintenance, habitat and gene pool protection	
Spiritual and religious values	Spiritual experience	Spiritual and/or emblematic	Supporting identities
Aesthetic values	Aesthetic information	Intellectual and representational interactions	Physical and psychological experiences
Cultural diversity	Inspiration for culture, art and design	Intellectual and representational interactions	Physical and psychological experiences
			Learning and inspiring
		Spiritual and/or emblematic	Supporting identities
Recreation and ecotourism	Recreation and tourism	Physical and experimental interactions	Physical and psychological experiences
Knowledge systems and educational values	Information for cognitive development	Intellectual and representational interactions	Learning and inspiring
		Other cultural outputs (existence, bequest)	

*Explanatory information from CICES division level [between squared brackets] and from CICES class level (between parentheses).

2.1.3 The cascade model

Despite the differences in definition and classification of what an ecosystem service is between each of the frameworks (Table 1), all frameworks agree that there is some kind of 'pathway' for delivering ecosystem services that goes from natural capital at one end through to the well-being of people at the other (Harrison et al., 2017; Potschin and Haines-Young, 2016). This can be described as a 'cascade' (Figure 4).

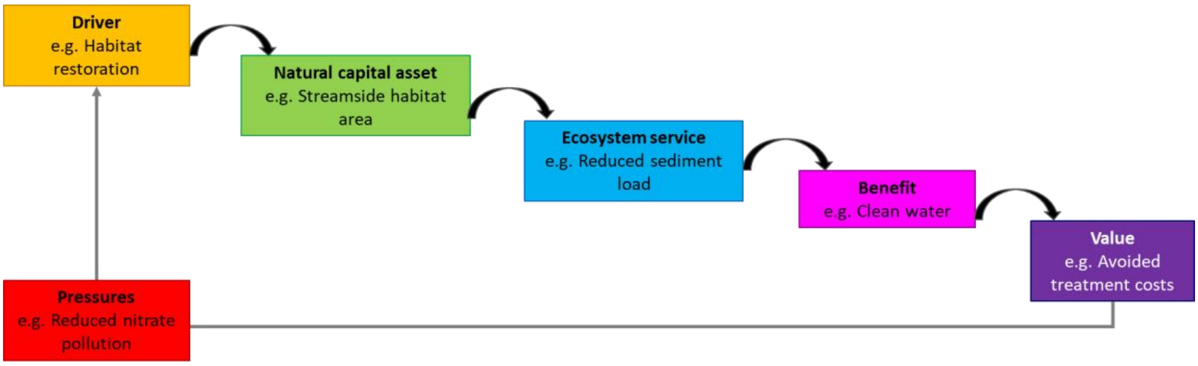


Figure 4: The cascade model adapted from Haines-Young and Potschin (2018) *and Harrison et al.* (2017). *See appendix for the original figures from Haines-Young and Potschin* (2018) *(Figure A1) and Harrison et al.* (2017) *(Figure A2).*

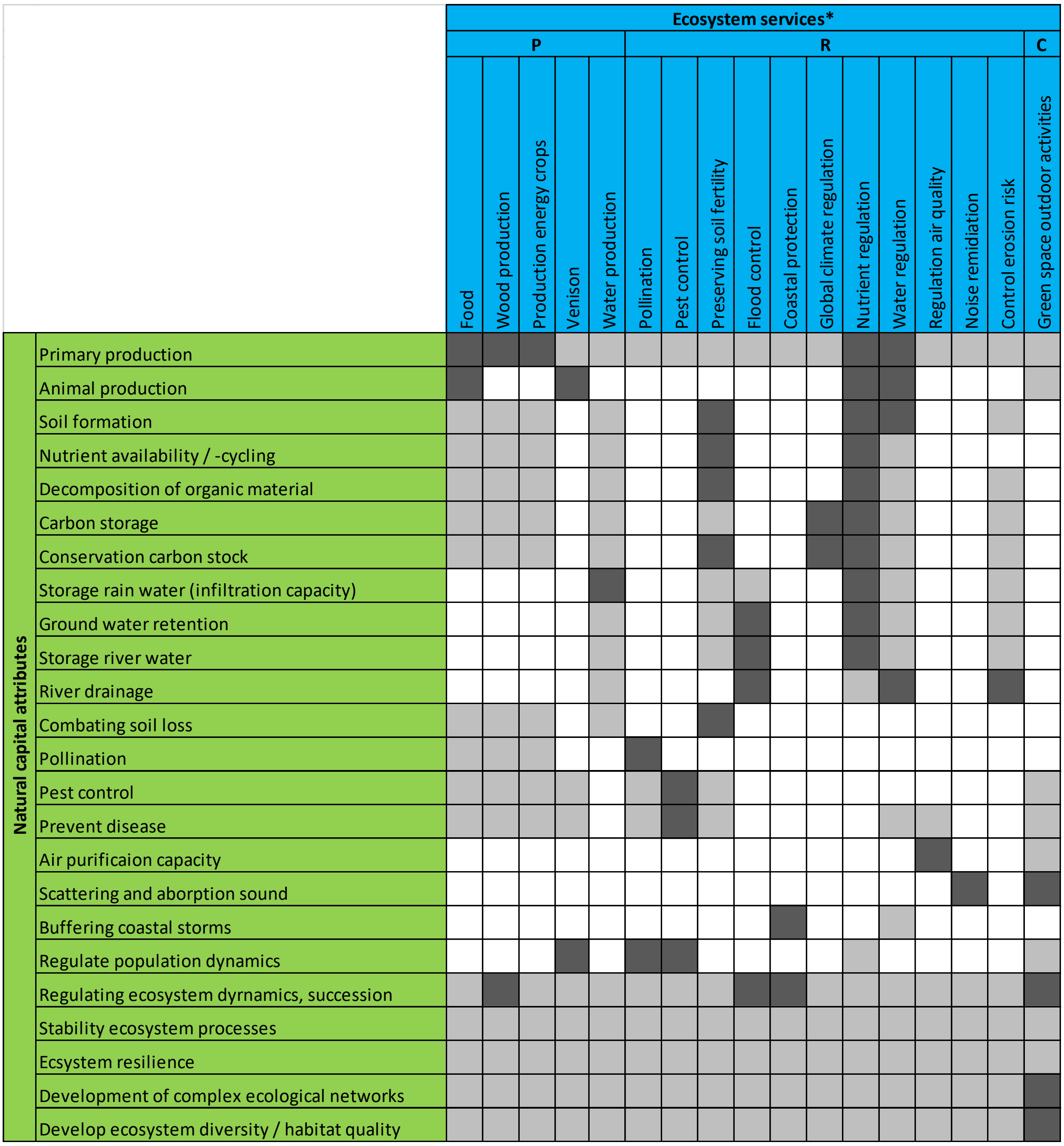
1.1.1.1 Natural capital asset

In the cascade model (Figure 4) the natural capital asset node is characterised by ecosystem properties (stock, condition and structure), and ecosystem functions that represent flow or processes (Harrison et al., 2017). For example, in the case of the ecosystem service timber production, a woodland is a natural capital asset which has underlying biophysical structural properties that include tree composition, soil type and nutrient status (Haines-Young and Potschin, 2018). These properties provide functions such as such as hardness, strength, and durability that make the wood material good for use as a building material.

1.1.1.2 Ecosystem service

An ecosystem service may depend on multiple functional attributes of the natural capital assets (Maseyk et al., 2017; Wu and Li, 2019). For example, timber production is also dependent on the branch and stem characteristics of the stand, stem density and stand age (Potschin and Haines-Young, 2016). Similarly, an attribute can give rise to several ecosystem services (Harrison et al., 2014; Smith et al., 2017). For example, a stand of trees can slow the passage of runoff, and those same trees can offer benefits in terms of shelter and recreational activities (Potschin and Haines-Young, 2016). Further examples of natural capital attributes supporting ecosystem services are shown in Table 2.

Table 2: Representation of natural capital attributes steering (dark grey) or supporting (light grey) an ecosystem service. White fields demonstrate indirect effects. Adapted from Burkhard and Maes (2017).



**Provisioning (P), Regulating (R), Cultural (C)*

1.1.1.3 Benefits, values and pressures

The benefit node represents something that can change someone's wellbeing. Benefits can be important to people. The extent of their importance to a person is expressed by the value that the person assigns to them (Van Oudenhoven et al., 2012). Alongside monetary values, people can express the importance they attach to benefits using moral, aesthetic or spiritual criteria. These values influence the degree to which people and societies choose to act to manage the pressures on ecosystems and ultimately the benefits they deliver to society. This feedback is highlighted in the arrow running from the values node back to the driver node via the pressures node (Figure 4). For example, wetlands (a natural capital asset) provide habitat for bacteria which break down excess nitrogen. This results in the removal of nitrogen from the water (an ecosystem service) resulting in clean water (a benefit). People can value improved water quality in multiple ways (e.g., by expressing their willingness to pay for clean water) (Burkhard and Maes, 2017). This value in turn limits pressures such as nitrate pollution and wetland habitat degradation, via policies such as the Water Framework Directive (in the UK), to maintain the benefit of clean water.

2.1.4 Synergies and trade-offs between cascades

The cascade model provides a useful framework for the analysis of each individual ecosystem service (Maes et al., 2013) as it allows for the operationalisation of the delivery of an ecosystem service by breaking it up into measurable entities (Boerema et al., 2017). As such, it has been widely used and adapted by many other researchers in varying degrees of complexity (e.g. Boerema et al., 2017; Harrison et al., 2017; Hernández-Morcillo et al., 2013; Van Oudenhoven et al., 2012). Several of these adaptions stress the importance of quantifying each step in the cascade to fully operationalise the assessment of ecosystem services in decision making (Boerema et al., 2017; Harrison et al., 2017). Additionally, other authors highlight the importance of taking account of trade-offs and synergies between the items of the cascade across multiple socioecological systems (Harrison et al., 2017). These trade-offs and synergies between the nodes of the cascade are shown visually by Figure 5.

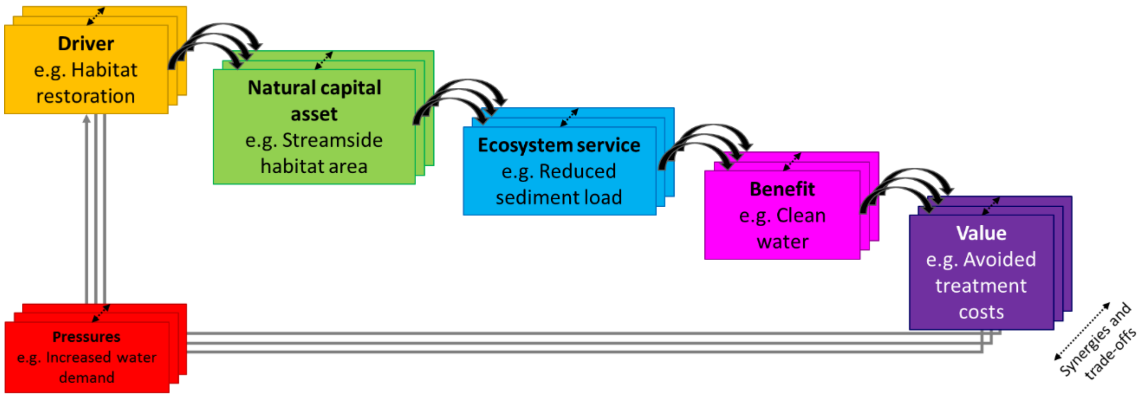


Figure 5: The cascade model from Figure 3 showing potential trade-offs and synergies between the cascades.

Figure 5 shows that the nodes of each cascade can feature trade-offs and synergies with the nodes of other cascades. For example, the natural capital asset soil carbon concentration has a synergistic linkage to soil moisture content (Falloon et al., 2011).

Methods for quantifying ecosystem service provision should try to incorporate all parts of the cascade and the synergies and trade-offs between cascades (Potschin-Young et al., 2018). Each of the steps in the cascade model can be quantified using biophysical, economic or social-cultural valuation methods (Burkhard and Maes, 2017). A range of methods for ecosystem service assessment are available, as well as guidance on choosing the most appropriate methods depending on context using decision trees (Harrison et al. 2018). Yet no method or tool exists that accounts for all of these interactions between nodes of the cascade and the interactions between cascades. To contribute to the solving of this large goal, this thesis focusses on one specific link in the cascade: the link between natural capital and ecosystem services and all associated trade-offs and synergies between natural capital and ecosystem services.

2.2 Existing methods for linking natural capital to ecosystem services

There is a lack of methods linking natural capital assets to final ecosystem services (Wong et al., 2015). Quantification of this link is an important step in operationalizing the concept for management and decision-making (Boerema et al., 2017). Currently there are three main approaches that are used to measure and evaluate the biophysical supply of ecosystem services:

metrics and indicators using primary data; transfer approaches using secondary data and land cover proxies; and simulation models.

2.2.1 Metrics and indicators

Ecosystem service indicators are used to represent and communicate the high complexity in human-environment systems that underlies the ecosystem service. They aim to make it possible for policy-makers to understand the condition, trends and rate of change in ecosystem services (Maes et al., 2016; Müller and Burkhard, 2012). For example, stream macroinvertebrate metrics can be calculated from observation made directly in the field and then used to reflect established relationships between diversity and water quality (Maskell et al., 2013). So typically primary data is used to quantify metrics and indicators (Burkhard and Maes, 2017) and expert opinion is used to qualitatively connect ecosystem characteristics to ecosystem services (Wong et al., 2015). Using the indicator approach, trade-offs and synergies between ecosystems services can be investigated through relationships between the primary data used for their indicators. Examples of such studies include Maskell et al. (2013), Qin et al. (Qin et al., 2015) and Sylla et al. (2020).

A wide range and diversity of indicators exist. Maes et al. (2016) reviewed the quality of 327 ecosystem service indicators for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. Only one fifth of these were assigned the highest quality label (H), 124 the medium quality label (M), 103 the low quality (L), and 36 the grey label for which the expert panel did not assign a quality label. The quality of these ecosystem service indicators by ecosystem and ecosystem service section is shown in Figure 6, as well as the descriptions of the quality labels.

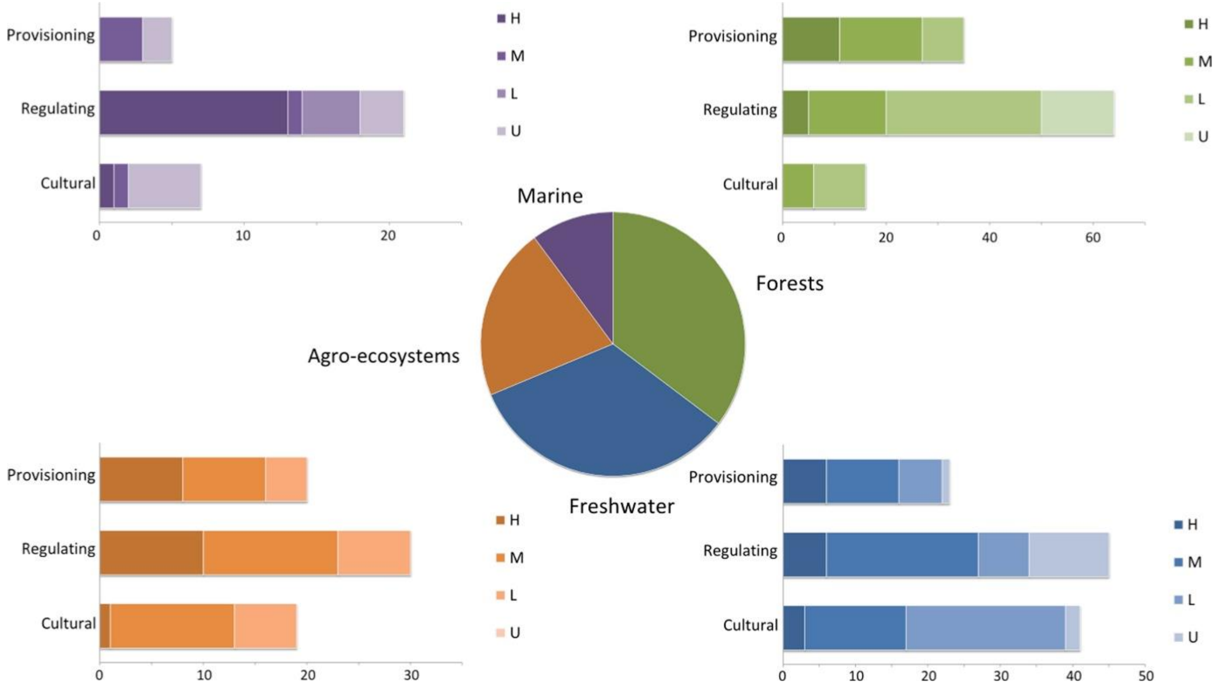


Figure 6: Number and classification of the indicators proposed to map ecosystem services in the European Union. The letters refer to the quality label of indicators: H for high, M for medium†, L for low‡ and U for unknown.*

**High (H): indicators which have harmonised, spatially explicit data available to measure the quantity of ecosystem service provision at the European scale that can be easily understood by policy makers and non-technical audiences.*
†Medium (M) represents medium quality indicators with available data that can measure the quantity of ecosystem service but either harmonised, spatially explicit data is not available at the European scale, the indicator only partially captures the ecosystem service, or the indicator can result in different interpretations by the user.
‡Low (L) represents low quality indicators that measure the condition of an ecosystem or quantity of an ecosystem service where no harmonised, spatially explicit data is available that only provided the information at an aggregated level and requires additional clarification to non-technical users.

This large abundance of available indicators as well as their broad range in quality (Figure 6) is due to many factors. As shown by Figure 4 many components contribute to the provision of an ecosystem service. Some indicators represent potential supply of an ecosystem services, whereas others represent actual supply (Boerema et al., 2017; Heink et al., 2016). For example, the best indicator for actual use of fish for food may be tonnes of fish landed per year (a "benefit" in Figure 4), whereas the best indicator for the potential use of fish for food may be fish population size (a "natural capital attribute" in Figure 4) (Heink et al., 2016).

Measurement cost as well as quantification and data scarcity challenges set practical and spatial limits on the application of indicators (Burkhard and Maes, 2017; Heink et al., 2016). This often leads to the use of only a small group of potentially representative variables to represent the service (Layke et al., 2012; Müller and Burkhard, 2012). This lower representation of ecosystem services can lead to oversimplification of the cascade model (Figure 4). For example, in the case of the ecosystem service of climate regulation, carbon sequestration is often the only component measured yet there are other essential components including climate moderation and sequestration of methane and nitrogen dioxide (Boerema et al., 2017). Pragmatism influences the type of indicators used to represent ecosystem service provision; data for indicators of potential ecosystem service provision are usually easier to acquire and therefore these indicators are more commonly used. Additionally, data availability is also context dependent. For example, less data is available to measure the state of marine ecosystem services than for terrestrial and freshwater systems (Figure 6).

Some indicators are not as strongly connected to the ecosystem service they represent as others (Maes et al., 2016). For example, Harrison et al. (2014) found that the links between species diversity and the delivery of ecosystem services requires further research and evidence gathering. This is particularly relevant to cultural services such as bird watching which are strongly connected to biodiversity. Figure 6 shows a high proportion of lower quality indicators for cultural services compared to other ecosystem services.

Even when considering a single indicator, different methods and data types are available as shown by Table 3; each with their own strengths and limitations.

Table 3: Examples of different methods to measure ecosystem service indicators; adapted from Burkhard & Maes (2017)

Ecosystem services*	Indicator	Direct measurement	Indirect measurement	Model
Cultivated crops	Crop yield (tonne/ha/year)	Crop statistics (obtained through official reporting)	Remote sensing of crop biomass using NDVI and aerial photo analysis for long temporal changes coupling structural observations with remote sensing information	Crop production models
Water (Nutrition)	Water abstracted (m3/year)	water statistics (obtained through official reporting)	Remote sensing of water bodies and soil moisture	Water balance models
Biomass (Materials)	Timber growing stock (m3/ha) and timber harvest (m3/ha/year)	Forest stand measurements and forest statistics	Remote sensing of forest biomass using NDVI	Timber production models
Mediation of waste, toxics and other nuisances	Area occupied by riparian forests (ha)	Site observations	Earth observation land cover data	
	Nitrogen and sulphur removal in the atmosphere or in water bodies (kg/ha/year)	Measurement of deposition of NO2 and SO2; field measurements of denitrifications in water bodies	Remote sensing of canopy structure (leaf area index)	Transport and fate models for N and S
Flood protection	Area of floodplain and wetlands (ha)	Site observations	Elevation models and data; aerial photo analysis; remote sensing of land cover	Modelling water transport

**CICES class*

These factors weaken the assumption that a single indicator can be representative of the ecosystem service as a whole (Boerema et al., 2017), and rather indicators are only highlighting one aspect of a broad concept. Therefore, transparency is essential when indicators are applied as the best choice of indicator depends on the purpose of the ecosystem service assessment and its audience (Heink et al., 2016; Müller and Burkhard, 2012). For example, in the case of pollination, a scientist could be interested in the diversity and density of different bee and bumblebee populations, a farmer may wish to know how far he can rely on wild pollination to help pollinate his fruit trees, whilst a biodiversity policy officer may need to know if, at the national scale, pollination services are declining or increasing (Burkhard and Maes, 2017). Despite the limitations, indicators of ecosystem services can be the most accurate method of quantifying ecosystem services.

2.2.2 Transfer approaches using secondary data and land cover proxies

Transfer approaches are a much lower cost method for ecosystem service assessment. These involve taking estimates from one site and applying them to another. One of the most common examples of the use of transfer approaches in ecosystem service assessment is the use of a matrix linking spatially explicit biophysical landscape units to ecosystem service supply and demand using expert judgements e.g. Burkhard et al. (2009). This transfers the expert derived values of land cover to other locations as shown by Figure 7.

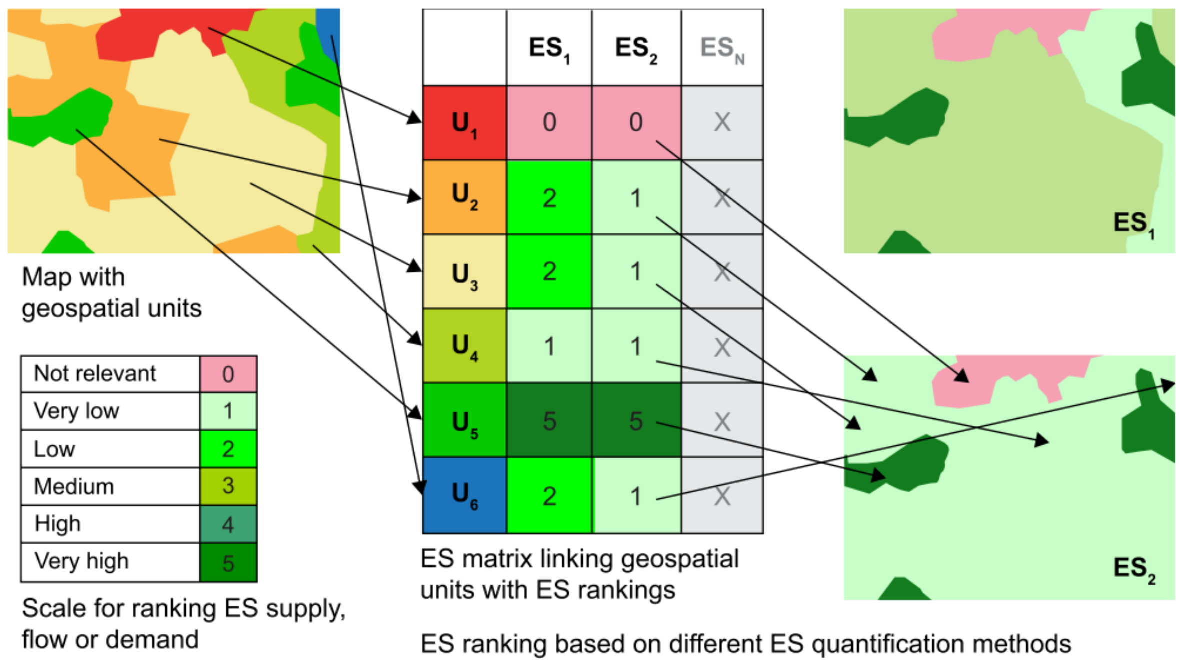


Figure 7: Overview of the ecosystem service matrix transfer approach (Burkhard and Maes, 2017).

The flexibility of the matrix approach allows the method to be applied to all spatial and temporal scales (Burkhard and Maes, 2017). It is fast, transparent, and easy to use and communicate (Burkhard et al., 2009; Harrison et al., 2018) and is much less costly than the primary data approach (Wong et al., 2015). Wong et al. (2015) found two thirds of the studies they reviewed measured ecosystem services using secondary data and land cover. However, this simplicity comes at a cost as this method is entirely land use driven and therefore makes the assumption that the dominant land cover is the principle driver of supply of all of the services (Burkhard et al., 2009; Schulp et al., 2014).

Land use alone lacks information regarding important components of ecosystem condition that supports ecosystem service capacities, such as soil type and quality, water availability, geomorphology or overall ecosystem integrity (Campagne et al., 2020). Furthermore, this aggregation of complex information (Burkhard et al., 2012) can give a false impression of completeness (Harrison et al., 2018; Shen et al., 2021). For example, the class of ''water bodies'' provides several ecosystem services but in reality various qualities of water bodies determine the real ecosystem service provision potential (Kopperoinen et al., 2014). More complex inputs can be used to combat this dependency, e.g. Kopperoinen et al. (2014), however this reduces the simplicity, manageability, generality, and comparably of the simple matrix based approach (Campagne et al., 2020).

Expert derived scores for ecosystem service provision are also subjective to the expert who came up with them and are not based on causal relationships between ecosystems characteristics and final services (Wong et al., 2015). Furthermore, many of these studies use the global values from Costanza et al. (1997), which are only rough guidelines (Boerema et al., 2017). Despite limitations, transfer approaches are very good for building awareness of ecosystem service provision in the absence of data or detailed models (Burkhard and Maes, 2017).

2.2.3 Modelling systems

Biophysical models can be empirical, or process based. Empirical models relate management and environmental factors to ecosystem functions through statistical relationships (Wong et al., 2015). The Universal soil loss equation is a widely used example that quantifies the regulation of soil erosion, $A = R \times K \times LS \times C \times P$ where: A = predicted soil loss, R = rainfall and runoff, K = soil erodibility, LS = slope length and steepness, C = crop management and P = support practices (such as farming direction or strip cropping). However, these relationships are correlative and not necessarily causal so may not fully describe system behaviours and interactions (Korzukhin et al., 1996).

The assumptions of empirical models are in statistical theory whereas the assumptions of process-based models are rooted in causal mechanisms grounded in ecological theory based on a theoretical understanding of relevant ecological processes (Cuddington et al., 2013; Wong et al., 2015). They use explicit assumptions of how the system works (Cuddington et al., 2013) and therefore show greater transparency. This leads to increased confidence in extrapolation of process-based models. Yet, developing this understanding of ecological processes requires more resources than empirical-based models (Choquet et al., 2021). Despite the merits of process-based modelling, the best approach is context and application dependent. When comparing empirical and process-based modelling to quantify soil-supported ecosystem services Choquet et al. (2021) found that process-based modelling is effective only in deep, homogeneous, and cultivated soils whereas empirical modelling is effective over a larger range of soils, but mostly for provisioning services.

Ecological production functions are process-based models where the output is a final ecosystem service (Wong et al., 2015). They specify the output of ecosystem services provided by an ecosystem given its condition and process. This allows researchers to quantify the impact of landscape change on ecosystem service provision (Tallis et al., 2011). Examples of ecological production functions include the relationships of habitat changes to fisheries production, pollination to crop yields and ecosystem conditions to air quality (Wong et al., 2015).

Integrated assessment models link both empirical and process based sectoral models such that the outputs of one model are used as inputs to another. For example, an agricultural model may calculate water availability from irrigation based on rainfall. Yet to identify whether the irrigation water is available to use, a water allocation model that splits water availability between sectors must be integrated with the agricultural model (Burkhard and Maes, 2017). As a result, these models allow for the incorporation of more parts of the cascade model (Figure 4) and more interactions between cascades (Figure 5). However, integrated models tend to be more time consuming and technically challenging (Burkhard and Maes, 2017).

2.3 Available tools for quantifying and assessing ecosystem service provision

There are currently many available tools for quantifying and assessing ecosystem service provision. These tools use one or a combination of the methods outlined in section 2.2.

2.3.1 Environmental Benefits from Nature (EBN) tool

The EBN tool scores (on a scale of 0-10) the ability of different habitat types to deliver 18 ecosystem services. These scores are established from literature review and expert consultations (Smith et al., 2021). The scores are modified by applying multipliers based on 40 indicators of habitat condition and spatial location, and then multiplied by the area of habitats, as well as by multipliers to reflect delivery risk and the time taken for new habitats to reach their target condition. The outputs show the change in provision of ecosystem services from the baseline to the post-development situation (Smith et al., 2021).

2.3.2 Spatial Evaluation for Natural Capital Evidence (SENCE)

SENCE (https://www.envsys.co.uk/sence/) is a rule based model that maps ecosystem services by looking at parcels of land and considering its land cover/habitat type, its geology and soil, its position in the landscape (e.g. steep slope, next to an urban area), and how it is managed. Rules are developed based on local knowledge and expert understanding of how habitat attributes deliver ecosystem services, and a relative value (high, medium, low) is assigned to each element in each dataset, with weightings applied when different datasets are combined (Medcalf et al., 2012). From this method, maps are derived showing the relative importance of the parcels of land for ecosystem services supply (Vorstius and Spray, 2015).

2.3.3 Co$ting nature (forestry research, 2017).

Co$ting nature (http://www.policysupport.org/costingnature) incorporates preloaded detailed global datasets, spatial empirical models for biophysical and socioeconomic processes and scenarios for climate and land-use. It also allows interventions (policy options) or scenarios of change and conservation priorities to be applied to understand their impact on ecosystem service delivery (Bagstad et al., 2013). By combining more than 140 input maps, the tool calculates the spatial distribution of 18 ecosystem services expressed in relative terms (0 - 1).

2.3.4 InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs)

The most established suite of process-based modelling systems is InVEST (https://naturalcapitalproject.stanford.edu/software/invest), which has been widely used for multiple contexts (Burkhard and Maes, 2017). InVEST models are based on ecological production functions and economic valuation methods to create spatially explicit values for 18 ecosystem services (Tallis et al., 2011). Outputs are given in both physical and monitory terms. InVEST is useful for understanding the consequences of alternative decisions when little information exists about a system and it is necessary to rely on more generalized functional relationships (Boumans et al., 2015). However, InVEST does not specifically model interactions between ecosystem services and is weaker for cultural ecosystem services; as is the case with many of the biophysical models (Burkhard and Maes, 2017).

2.3.5 LUCI (Land Utilisation and Capability Indicator)

LUCI is a decision support tool that can model ecosystem service condition and identify locations where interventions or changes in land use might deliver improvements in ecosystem services (Sharps et al., 2017). LUCI, like InVEST, uses ecological production functions (where possible) to model physical processes (Jackson et al., 2013). Outputs show parts of the landscape that currently provide ecosystem services and areas where management interventions could enhance or degrade services (Bagstad et al., 2013). LUCI has a unique, built-in trade-off tool, which allows the user to identify locations where there is potential for "win-wins" (where multiple services might benefit from interventions), or trade-offs (with one service benefitting from interventions while another is reduced) (Sharps et al., 2017). LUCI is designed to use simple algorithms and outputs to transparently communicate ecosystem service trade-offs in settings with stakeholders and decision makers, and allows near-real-time results to be presented in public forums (Bagstad et al., 2013).

2.3.6 EcoServ-GIS

EcoServ-GIS maps the capacity of an ecosystem to supply a service as well as areas of demand for the service on a county scale (Bellamy et al., 2014). EcoServ uses readily available datasets and is accessible to non-experts. It uses available UK datasets to create a base map assigning a habitat type to each parcel of land. Process-based ecosystem service models then use this base map to create capacity and demand models based on either look-up tables or indicators for ecosystem processes (Bagstad et al., 2013). These models are not production functions as they use models to proxy a service of interest (Bagstad et al., 2013). The maps are then overlaid to identify areas of ecosystem service provision (Vorstius and Spray, 2015). EcoServ-GIS is now available as an R package called EcoservR and can be found at (https://ecoservr.github.io/EcoservR/).

2.3.7 IAP2 (Integrated Assessment Platform 2)

The IMPRESSIONS Integrated Assessment Platform 2 combines ten sectoral models and simulates cross-sectoral interactions between agriculture, forestry, biodiversity, water resources and flooding under varying climate and socio-economic conditions (Harrison et al., 2019). IAP2 splits Europe into 10` x 10` cells and simulates values for ecosystem service indicators for each cell. It can also map ecosystem service provision for future plausible scenarios quantified using three Representative Concentration Pathways (2.6, 4.5 and 8.5) and four Shared Socioeconomic Pathways (1, 3, 4 and 5) out to 2100. Scenarios can be run for the three time slices 2020s, 2050s and 2080s (Harrison et al., 2019).

2.3.8 MIMES (Multiscale Integrated Model of Ecosystem Services)

Another integrated model is MIMES, an ecosystem-based management tool. It integrates georeferenced datasets, with diverse information sources on human and natural systems to create production functions. These production functions assess the value of ecosystem services at different spatial levels under different future scenarios (Bagstad et al., 2013; Boumans et al., 2015). The system elements are grouped in spheres each representing processes that generate natural and human system flows. This allows MIMES to capture the dynamics and feedbacks of multiple ecosystem service productions and demands simultaneously (Boumans et al., 2015). For this reason MIMES is well suited to examine trade-offs under various economic, policy, and climate scenarios in space and over time (Boumans et al., 2015). However, MIMES has heavy data input requirements as well as complex outputs which require training and familiarity with the system to understand.

2.3.9 ARIES (ARtificial Intelligence for Ecosystem Services)

ARIES was developed as an online platform to allow the building and integration of models such as those discussed above. It allows the most appropriate ecosystem services model to be assembled automatically from a library of modular components, driven by context-specific data and machine-processed ecosystem services knowledge (Villa et al., 2014). Consequently, ARIES moves away from the idea that one model should fit all circumstances. While other methods such as InVEST and LUCI focus on using known biophysical relationships (where possible) to model physical processes, ARIES, in addition to standard modelling approaches can also use probabilistic methods (Bayesian networks) if there is insufficient local data to use in biophysical equations (Sharps et al., 2017). This Bayesian framework relies on the data itself to inform functional relationships rather than predefining the relationships as is the case with deterministic models (Villa et al., 2014).

2.4 Summary and challenges

To better inform the environmental management decisions currently degrading our natural capital assets we must understand all complexities involved in the generation of the ecosystem services and benefits they provide. Previous work has used a wide variety of assumptions and methods of varying complexity to assess and communicate the value of our natural capital assets to decision makers. These methods underpin an abundance of interactive tools and platforms. This recent increase in interactive tools and platforms is due to the great complexity of the natural and socioeconomic systems and processes that influence the provision of ecosystem services and benefits, and the need to synthesise such information so that the outputs are relevant to the needs of a wide variety of decision makers.

As a result of this great complexity most assessments of ecosystem service provision take a "by ecosystem service" approach. In doing so they fail to account for the important trade-offs and synergies between the nodes of the cascade (Figure 4), potentially leading to unexpected undesirable outcomes when informing environmental management decisions. Furthermore, many studies do not consider the context dependency of the links between nodes of the cascade or the context dependency of the synergies and trade-offs between nodes of different cascades. Decision makers need to know if the processes, synergies, and trade-offs underlying ecosystem services are present in their own context. Finally, each of the different approaches and tools available for investigating and assessing ecosystem services use different methods, assumptions, evidence types, resulting in varying levels of representation of the complexities of the cascade. Awareness of the influence of choice of method and evidence on the outputs provided by previous studies and existing tools is essential so that decision makers can make educated and informed choices.

There is a clear need for exploration of the linkages between each node of the cascade and the trade-offs and synergies between different cascades, while also accounting for context dependencies and maintaining awareness of the methods, data and assumptions used to evidence such linkages and trade-offs. This would be an immense task and is out of the scope of a PhD thesis. Therefore, this thesis applies this approach to one link in the cascade: the link between natural capital assets and ecosystem services. Key aspects of this approach are integrated into a decision support tool. The interactive nature of this tool allows decision makers to navigate the complexities underlying the influence of natural capital on ecosystem service by synthesizing and exploring evidence that is relevant to their needs.

3 A visualization platform to analyze contextual links between natural capital and ecosystem services

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3.1 Abstract

To prevent further loss of our vital ecosystem services we must understand the linkages to their supporting natural capital attributes. Systematic literature reviews synthesise evidence of natural capital attribute to ecosystem service (NC-ES) linkages. However, such reviews rarely account for the context dependency of evidence that is derived from individual studies undertaken for a particular purpose, at a specific spatial scale or geographic location. To address this deficiency, we developed the LiNCAGES (Linking Natural Capital Attribute Groups to Ecosystem Services) platform for investigating the context dependency of literature-based evidence for NC-ES linkages. We demonstrate the application of the LiNCAGES platform using the OpenNESS systematic literature review of NC-ES linkages. A hypothetical use case scenario of a small-scale European forest manager is described. We find evidence for many NC-ES linkages, and trade-offs and synergies between services, is severely diminished or non-existent under certain contexts, such as larger spatial scales and European study location. The LiNCAGES platform provides a flexible tool that researchers can use to support collation, exploration and synthesis of literature-based evidence on NC-ES linkages. This is vital for providing credible and salient evidence to stakeholders on important NC-ES linkages that occur under their context, to guide effective management strategies.

3.2 Introduction

Natural capital is the world's stock of natural assets, which supplies a wide range of ecosystem services that directly or indirectly produce value for people (Smith et al., 2017). Ecosystem services are vital for human existence and good quality of life, yet global indicators of ecosystem extent and condition have declined by 47%, relative to their earliest estimated states (Díaz et al., 2019). This is likely to have repercussions for ecosystem services. To prevent further loss of ecosystem services we must understand how they are influenced by attributes of natural capital (de Bello et al., 2010; Díaz et al., 2019; Harrison et al., 2014; Ricketts et al., 2016), so that manageable natural capital attributes that are essential for ecosystem service delivery can be identified (Harrison et al., 2014; Maseyk et al., 2017). However, due to their broad and complex nature, investigation of natural capital attribute to ecosystem service (NC-ES) linkages must incorporate a holistic approach and account for context dependency (Adhikari and Hartemink, 2016; Gutierrez-Arellano and Mulligan, 2018; Harrison et al., 2014).

Holistic investigation is important as an attribute of natural capital can support the provision of one ecosystem service, while at the same time antagonising another service (Harrison et al., 2014; Maseyk et al., 2017). Furthermore, ecosystem services themselves can interact with each other both positively and negatively, as some regulating ecosystem services underpin the delivery of other services, particularly provisioning services (Boerema et al., 2017; Raffaelli, 2006; Ziter, 2016). For example, pollination is critical for the delivery of food production (Harrison et al., 2014). However, using land for food production reduces or removes the provision of some regulating and cultural services, such as atmospheric regulation, erosion control, air and water quality regulation and recreation that would be provided if the land were forested (Maes et al., 2012).

Many of the NC-ES linkages evidenced in the literature are highly context dependent (Adhikari and Hartemink, 2016; Harrison et al., 2014). This context dependency includes aspects such as ecosystem type, spatial and temporal scale, geographical location, and study method.

Ecosystem type influences NC-ES linkages (Feld et al., 2009; Hevia et al., 2017; Maskell et al., 2013). For example, urban forests typically store about half as much carbon as natural forests, so are less effective in providing the ecosystem service of atmospheric regulation (Zhao et al., 2010). This is thought to be due to their younger age structure (Zhao et al., 2010). NC-ES linkages are influenced by the spatial scale at which the natural capital attributes operate and the scale at which the ecosystem service is delivered (Burkhard et al., 2012; de Bello et al., 2010; Haines-Young and Potschin, 2011; Hevia et al., 2017; Maskell et al., 2013; Raffaelli, 2006). Duncan et al. (2015) found that assessing NC-ES linkages at large spatial scales resulted in significant information loss of the mechanisms underpinning NC-ES linkages, as key ecosystem functions work at finer scales. Additionally, Ricketts et al. (2016) found that broader spatial scale studies might evidence more positive NC-ES linkages as they capture a greater variation of natural capital attributes. The temporal scale of a study has also been shown to influence NC-ES linkages (Cimon-Morin et al., 2013). For example, different pollinator species begin flight at different times in the day due to differing body size, warm-up rates and ambient flight temperature (Kremen, 2005). As a result, coarser temporal scales may not capture the influence of some of these species on pollination services. Experiment type has also been known to affect NC-ES linkages, with Balvanera et al. (2006) finding more positive NC-ES linkages where environmental variables could be controlled best, such as greenhouse experiments.

These context dependent aspects also interact with each other. For example, the ecosystem type under investigation can affect study temporal scale, e.g., due to the difficulty in maintaining experimental setup in a hostile environment (Raffaelli, 2006). The aspects are also influenced by other pragmatic factors such as the time available in a research studentship or grant (Martnez-Harms and Balvanera, 2012; Raffaelli, 2006), the maximum plot size that could be handled by the researcher, and the space available for the work (Raffaelli, 2006). These limitations lead to the completion of experimental NC-ES studies mostly at small scales, with the larger scale studies using secondary source evidence and modelling (Martnez-Harms and Balvanera, 2012).

The context dependency of evidence on NC-ES linkages is not explored in many studies (Duncan et al., 2015), making the transferability of empirical evidence and its synthesis, difficult. Furthermore, evidence for NC-ES linkages in the literature is highly fragmented (de Bello et al., 2010; Smith et al., 2017) and can be difficult to locate through standard search engines due to the vagueness and imprecision of ecosystem service definitions (Boerema et al., 2017; Englund et al., 2017). While it can be argued that this encourages creativity and transdisciplinary collaboration, it also leads to difficulty in finding and synthesising relevant information from the literature (Boerema et al., 2017).

Previous studies have attempted to overcome these limitations and synthesise the literature on NC-ES linkages using systematic review methodologies. However, the majority of these studies fail to incorporate a holistic approach by focussing on individual ecosystem functions, taxonomic groups or ecosystem services (Gutierrez-Arellano and Mulligan, 2018; Harrison et al., 2014; Lefcheck et al., 2015; Ziter, 2016). Most studies focus on species level natural capital attributes, yet functional group and population level natural capital attributes are also vital for underpinning the supply of ecosystem services (Ricketts et al., 2016).

Seven systematic reviews attempted to be more holistic in their design Balvanera et al. (2006), de Bello et al. (2010), Harrison et al. (2014), Hevia et al. (2017), Ricketts et al. (2016), Schwarz et al. (2017) and Smith et al. (2017). For comparison of these reviews, see Table B1. The number of studies used in the systematic reviews varies considerably from 103 in Balvanera et al. (2006) to 780 in Smith et al. (2017), as do the number of ecosystem services investigated from four in Ricketts et al. (2016) to 13 in both Smith et al. (2017) and Hevia et al. (2017). Some reviews were limited to a specific ecosystem, e.g., Schwarz et al. (2017) focused on urban environments, and some reviews recorded significantly more study aspects that could identify context dependency, with the greatest amount recorded by Smith et al. (2017). See Appendix B Section 4 for study aspects recorded by Smith et al. (2017).

These reviews use a variety of methods from vote counting to meta analysis, with the majority favouring some form of vote counting approach due to the widely varying disciplines involved in

NC-ES research (Smith et al., 2017). This vote counting aproach is a major limitation, as it assumes equal contribution of evidence from the studies. Admittedly de Bello et al. (2010) and Hevia et al. (2017) did filter for studies that showed significant linkages. Harrison et al. (2014) attempted to add a strength of evidence parameter to the NC-ES linkages that were extracted from the studies they considered, but this was later abandoned by Smith et al. (2017)., who reverted to vote counting when building on this work due to the use of many incompatible indicators and approaches in the literature base. However, Smith et al. (2017). did not attempt to account for the context of the NC-ES linkages when assigning a weight to the evidence provided by a particular study. This is most likely due to the subjectivity of the context of a NC-ES linkage, as certain contexts may be more useful for specific research or stakeholder questions.

This study aims to address these limitations by building on the work of Smith et al. (2017). through developing a platform for Linking Natural Capital Attribute Groups to Ecosystem Services (LiNCAGES). LiNCAGES aims to support the dialogue between science and policy by improving stakeholder's understanding of important natural capital to ecosystem service linkages, and associated trade-offs and synergies, relevant to their own context. Thus, it provides scientific evidence that is more salient to their needs. For example, a local landowner may prefer to use evidence from local scale studies from a similar landscape, whilst a policymaker may prefer national scale studies covering multiple ecosystems. LiNCAGES also aims to provide a resource for researchers by enabling consistent collation of the fragmented knowledge base on natural capital and ecosystem services, to identify key gaps in evidence. This fosters collaborative working, to target and collate additional evidence that can strengthen the sustainable management of natural capital for the benefit of people and biodiversity. This paper describes the development and features of the LiNCAGES platform and its application to a hypothetical use case.

3.3 Methods

The LiNCAGES platform (available at: https://shiny-apps.ceh.ac.uk/LiNCAGES/) was developed and tested using the Operationalisation of Natural Capital and Ecosystem Services (OpenNESS) database (Smith et al., 2017). We chose OpenNESS as it provides a recent and substantial evidence base pertaining to a wide range of ecosystem services and natural capital attributes, in addition to recording the largest amount of context dependent aspects of all of the review studies we considered. The OpenNESS database consists of a systematic literature search of 780 peer-reviewed journal articles published in the English language across 13 ecosystem services, targeting 60 papers per ecosystem service. It used a standardised protocol based on customized keywords developed by Harrison et al. (2014) and covered articles published up until the end of June 2014 (Pérez Soba et al., 2017; Smith et al., 2017). See Appendix B Section 2 for the list of key words used to create the OpenNESS database. Some journal articles were split into multiple studies if they addressed more than one ecosystem, location or ecosystem service, and were entered separately into the database (Pérez Soba et al., 2017).

The database includes: four provisioning ecosystem services (food production (crops), freshwater fishing, timber production, water supply); seven regulating ecosystem services (air quality regulation, atmospheric regulation (carbon sequestration), mass flow regulation (erosion protection), water quality regulation (water purification), water flow regulation (flood protection), pollination and pest regulation); and two cultural ecosystem services (species-based recreation and aesthetic landscapes) (Pérez Soba et al., 2017). For each article, the reviewer recorded the direction of each of the linkages between the ecosystem service the study was investigating and 42 natural capital attributes (30 biotic and 12 abiotic). See Appendix B Section 3 for a list of all the natural capital attributes and their definitions. The reviewer classified each of these 42 NC-ES linkages as positive, negative, unclear, both (positive and negative), or not mentioned. An unclear linkage direction was assigned when the study mentions that the ecosystem service is affected but does not give an indication of the direction. To avoid confusion for the user of the LiNCAGES platform, we grouped “unclear” and “both (positive and negative)" linkage directions underneath the umbrella term of "unclear". The OpenNESS database classified the natural capital attributes of soil and geology as categorical and therefore they could not be assigned a direction of relationship, so the direction of NC-ES linkages with these natural capital attributes were classified as unclear (Pérez Soba et al., 2017).

In this use case scenario, we investigate the context dependency of the study aspects: spatial scale, temporal scale and location as an example, though the LiNCAGES platform can be used to investigate a further 13 context dependent aspects present in the OpenNESS database. For all the context dependent study aspects available in the LiNCAGES platform, see Appendix B Section 4.

The LiNCAGES platform accounts for these context dependent study aspects by allowing the user to filter or weight them according to their needs. Filtering allows the user to remove all the studies that feature a certain aspect and then create visualisations from the filtered studies. If a study contains multiple options for the same study aspect, e.g. where it spans multiple continents, the LiNCAGES platform will only filter out the study once the user has chosen to filter all of the options it contains. In cases where filtering may be considered too strict, for example when the user wishes to prioritise certain types of studies above others, but not lose studies entirely, which would reduce their sample size, weighting can be used. Weighting allows the user to attach greater importance to specific contexts using a two-tier hierarchical weighting system, described visually in the user input section of Figure 8. For hierarchical weighting the user weights the level 2 aspects (e.g. 'local', 'national') relative to each other and then assigns an overall weight (level 1 aspect weight) to their level 2 weighting choices for each of the level 1 aspects (e.g. spatial scale, temporal scale). The total weight of the study is then calculated as shown by the method flowchart in Figure 8; see also Appendix B Section 5 for a worked example of calculating the total study weight based on example user assigned weights. To help inform the definition of weights, the LiNCAGES platform shows the frequency of studies that feature each context dependent study aspect.

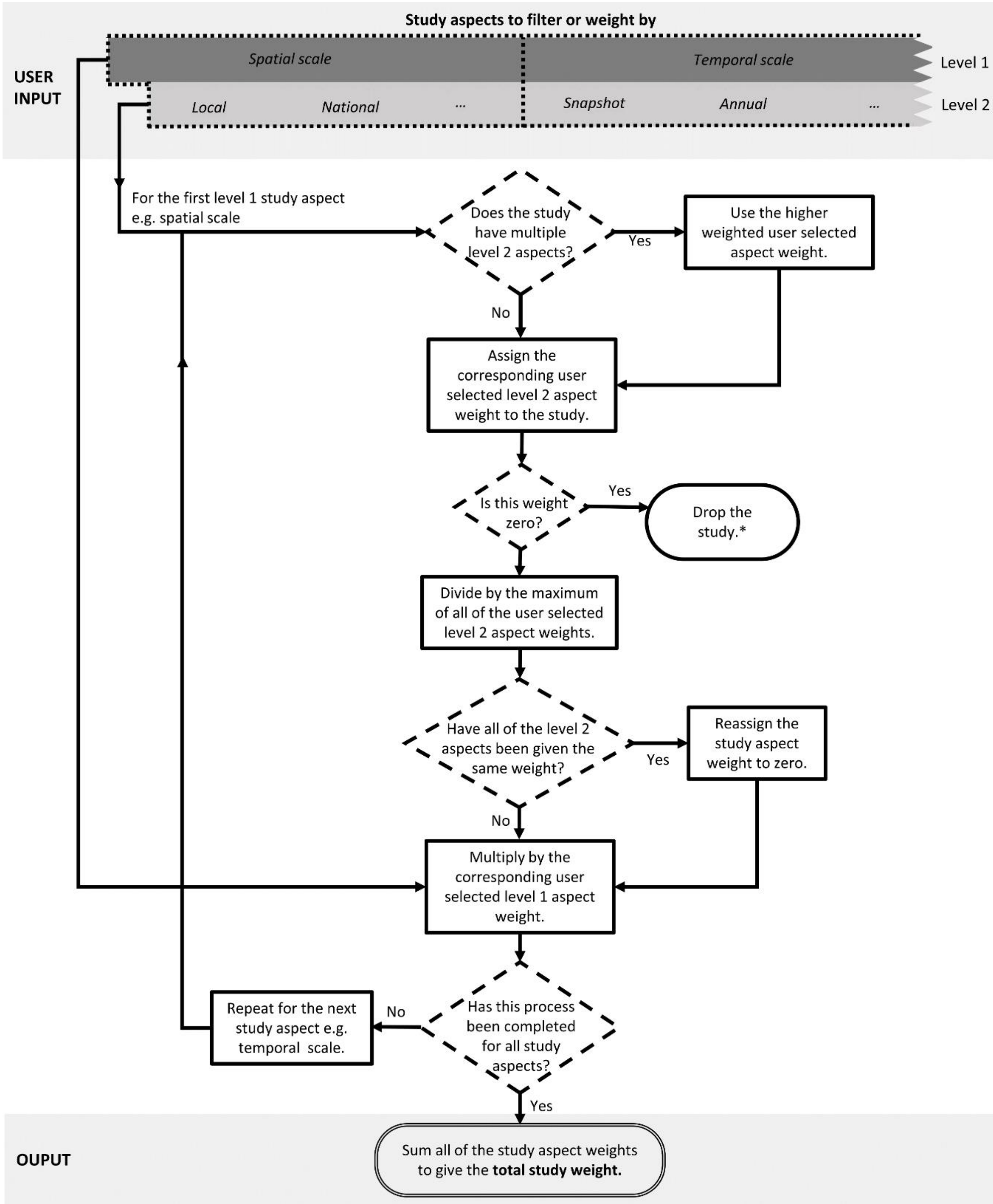


Figure 8: Method flowchart used by the LiNCAGES platform for filtering and hierarchically weighting the studies. The user input section features a visual representation of the two-tier hierarchical weighting system used in The LiNCAGES platform. The torn effect on the right-hand side of the user input section indicates that the hierarchy continues to include a further 14 level 1 aspects. Likewise, the level 2 aspect box with "…" indicates that there are actually more level 2 aspects to weight by than are shown in this visual representation.

**Assigning '0' to an aspect indicates the user wishes to filter out articles featuring that level 2 aspect.*

The total study weight is assigned to all of the NC-ES linkages for which that study provides evidence. This process is repeated for all studies included in the OpenNESS database. Then the sum of all of the study weights that support each of the included NC-ES linkages is calculated and can be visualised as either a stacked bar plot or network diagram.

The network diagrams produced by the LiNCAGES platform are fully interactive and allow the user to select particular NC-ES linkages and extract references for all of the studies that evidence that linkage. Selecting a node (ecosystem service or natural capital attribute) in the network diagram will output a reference table of all of the evidence for NC-ES linkages with that node. If the user has chosen to weight by particular context aspects, these references are ordered by their total weight, allowing the user to identify which studies are more relevant to their chosen context (Table B2). The user can also download a more detailed breakdown of how the weights for each of the studies have been calculated from their chosen weightings (Table B3). A weighted network diagram (Figure B5) and extracted references for a NC-ES linkage (Table B2) can be found in the weighting and filtering worked example in Appendix B Section 5.

LiNCAGES has been strongly informed by both stakeholder and researcher feedback throughout its development. We demonstrated LiNCAGES using an iPad at a variety of workshops and conferences (Lancaster Local Nature partnership, exhibit stand at The Centre for Global Eco-Innovation's Eco-I conference, Natural Capital Initiative Summit and the Lancaster Environment Centre Spring 2019 conference) to collect feedback on functionality, visualisations and ease of use.

3.4 Results

We demonstrate the potential applications of the LiNCAGES platform through discussion of a hypothetical use case scenario. A small-scale European forest manager receives benefits from the ecosystem service of timber production. The forest manager wants to use the LiNCAGES platform to discover which attributes of their natural capital they should promote to maximise the benefits they receive from timber production. Throughout this use case scenario, we use bar plots to visualise the NC-ES linkages as feedback on LiNCAGES from stakeholders and researchers was that bar plots were the easiest visualisation option to interpret and understand quickly. However, other visualisation options are available within LiNCAGES (network diagrams (Figure B5) and radar plots (Figure B6)) which may be more suitable to certain users or applications.

3.4.1 NC-ES linkages for one ecosystem service

First, the forest manager uses the LiNCAGES platform to visualise all the evidence for linkages between the 42 natural capital attributes and the ecosystem service of timber production (Figure 9).

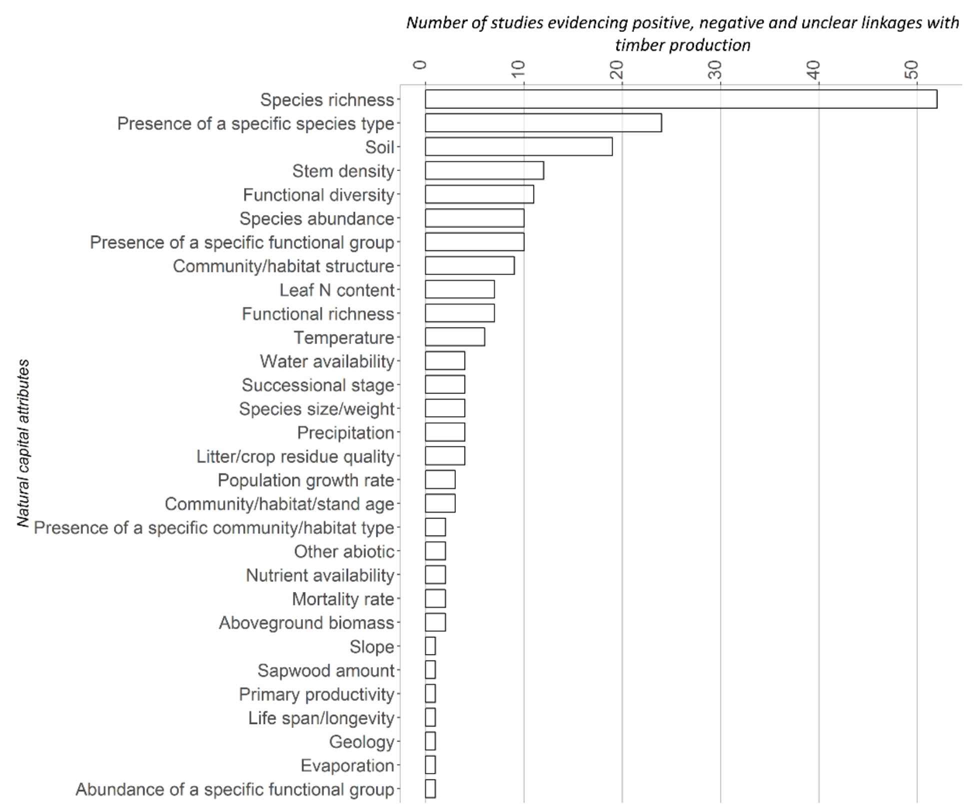


Figure 9: Bar graph of the number of studies evidencing NC-ES linkages with the ecosystem service of timber production for all directions of linkage (positive, negative and unclear), ordered by number of studies providing evidence. Natural capital attributes without evidence for NC-ES linkages with timber production are not shown in this figure.

Figure 9 shows that timber production has 21 linkages with biotic attributes and nine linkages with abiotic attributes. Species richness has the most evidence for linkages with timber production, followed by presence of a specific species type and soil. Overall, species and functional group level natural capital attributes have the most evidence for linkages with timber production. Figure 9 shows the amount of evidence for a linkage with timber production; yet does not show whether the linkage is positive or negative. Therefore, we used the LiNCAGES platform to compare the amount of evidence for positive and negative linkage directions (Figure 10).

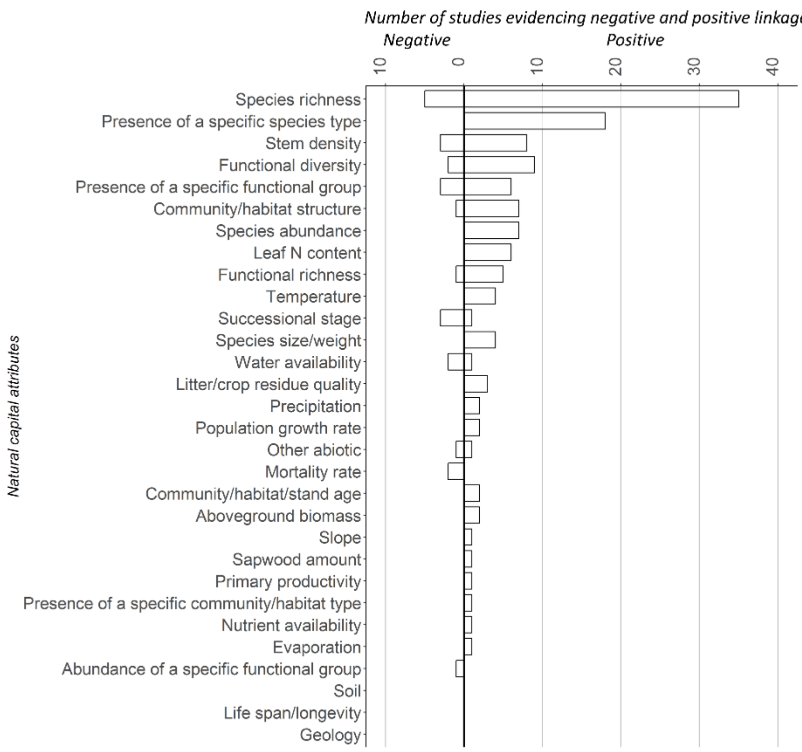


Figure 10: Mirrored bar graph of positive and negative NC-ES linkages with the ecosystem service of timber production. The number of studies evidencing positive linkages and negative linkages is shown on the positive (right) and negative (left) part of the x-axis respectively.

Figure 10 shows that in total, there are 25 positive and 11 negative NC-ES linkages with timber production. By comparing Figure 9 and Figure 10 we find no evidence for positive or negative linkages for the natural capital attributes of soil and geology because the OpenNESS database classified the direction of these NC-ES linkages as unclear (Pérez Soba et al., 2017); this highlights the importance of considering the methodology used to extract the evidence supporting the LiNCAGES platform. Species richness and presence of a specific species type have the most evidence for positive linkages with timber production. This is counterintuitive because positive linkages with presence of a specific species type suggests monocultures are best for timber production (e.g. Paquette and Messier, 2011), whereas positive linkages with species richness suggest mixed species forests are best for timber production (e.g. Bristow et al., 2006; Vilà et al., 2013). Figure 10 shows further uncertainty in the direction of NC-ES linkages; nine of the natural capital attributes have both positive and negative linkages with timber production, with species richness having the most evidence for both positive and negative linkages. Many systematic reviews stop their analysis at this stage (e.g. Balvanera et al., 2006; de Bello et al., 2010; Harrison et al., 2014; Ricketts et al., 2016; Hevia et al., 2017; Schwarz et al., 2017; Smith et al., 2017) and do not investigate this uncertainty in direction.

3.4.2 Context dependency of NC-ES linkages for one ecosystem service

Context dependency could be responsible for both the uncertainty in direction of the NC-ES linkages and the counterintuitive observations shown by Figure 10. The forest manager explores the context dependency of the evidence behind the NC-ES linkages with timber production by using the LiNCAGES platform to filter the evidence to three different contexts: (a) evidence from studies with spatial scales larger than local, (b) evidence from studies with a snapshot temporal scale (short term study) and (c) evidence from studies undertaken in Europe. Figure 11 shows how the amount of evidence for the NC-ES linkages with timber production changes under the three respective contexts, compared to the unfiltered NC-ES linkages in Figure 10, allowing the forest manager to review the influence of context on both the direction and existence of NC-ES linkages.

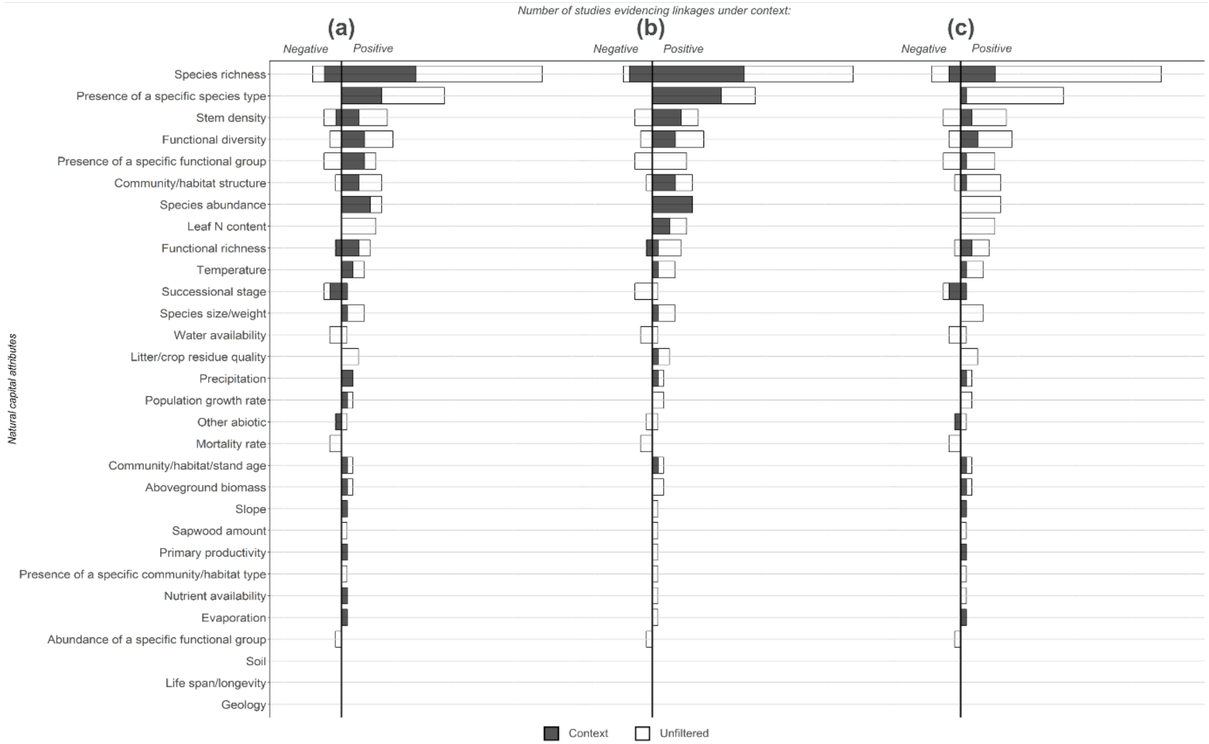


Figure 11: Mirrored overlay bar graphs showing the positive and negative NC-ES linkages with the ecosystem service of timber production, filtered for three different contexts: (a) evidence from studies with spatial scales larger than local, (b) evidence from studies with a snapshot temporal scale and (c) evidence from studies undertaken in Europe. Evidence for positive and negative NC-ES linkages are shown on the positive (maximum= 35) and negative (maximum = 5) part of the x-axis, respectively. To aid comparison, for each context the filtered NC-ES linkages (dark grey) are overlaid onto the unfiltered NC-ES linkages (white) from Figure 10.

Figure 11(a) shows that filtering the evidence for larger spatial scales than local entirely removed evidence for NC-ES linkages for seven natural capital attributes: leaf N content, water availability, litter/crop residue quality, mortality rate, sapwood amount, presence of a specific community/habitat type and abundance of a specific functional group. This suggests that these NC-ES linkages with timber production are only observed at local spatial scales, indicating that some NC-ES linkages with timber production have a strong spatial scale dependence, resulting in studies with coarser spatial scales underestimating the importance of some natural capital attributes for timber production. However, the user must also consider the effect of the differing sample size for these contexts. The majority (37) of studies evidencing NC-ES linkages for timber production have a local spatial scale and about a quarter (14) have a sub-national spatial scale. Very few studies had larger spatial scales.

Similarly, considering evidence from snapshot studies, which consist of nearly half of the studies (27), removes evidence for linkages between 14 natural capital attributes and timber production as shown by Figure 11(b). One of these natural capital attributes is presence of a specific functional group, which lost evidence from all nine studies that supported a positive or negative linkage with timber production. This observation may be due to studies with shorter temporal scales missing longer-term ecological processes.

Finally, as shown by Figure 11(c) filtering for evidence from European studies omitted a large proportion of studies (29) from North America and resulted in the removal of NC-ES linkages with 11 natural capital attributes, including species abundance and leaf N content, which lost evidence from five and seven studies, respectively. Furthermore, contrary to Figure 11(a) and Figure 11(b), under a European context there is very little evidence for NC-ES linkages with presence of a specific species type. This is due to the majority of European studies evidencing NC-ES linkages with timber production in the OpenNESS database focussing on natural forests rather than plantations (e.g. Vilà et al., 2013).

3.4.3 NC-ES linkages with other ecosystem services

Forests are known to generate multiple ecosystem services (Foley et al., 2007; Maes et al., 2012). Without holistic investigation of the NC-ES linkages, the forest manager may choose to promote a natural capital attribute that would have a positive effect on timber production but could lead to unintended consequences for another service they value. For example, the forest manager may receive payments from a water company to maintain good water supply, therefore they wish to avoid promoting natural capital attributes that will be detrimental to water supply. Furthermore, the forest manager is aware of payments for other ecosystem services their forest provides, such as atmospheric regulation and aesthetic landscapes, and therefore is interested in understanding the linkages between these ecosystem services and their natural capital attributes. The forest manager uses the LiNCAGES platform to explore the trade-offs and synergies between the unfiltered NC-ES linkages for timber production and three other ecosystem services: the provisioning service of water supply, the regulating service of atmospheric regulation and cultural service of aesthetic landscapes. Figure 12 shows the amount of evidence for the positive and negative NC-ES linkages for the four services, and how the overall amount and direction of evidence changes as each ecosystem service is cumulatively added.

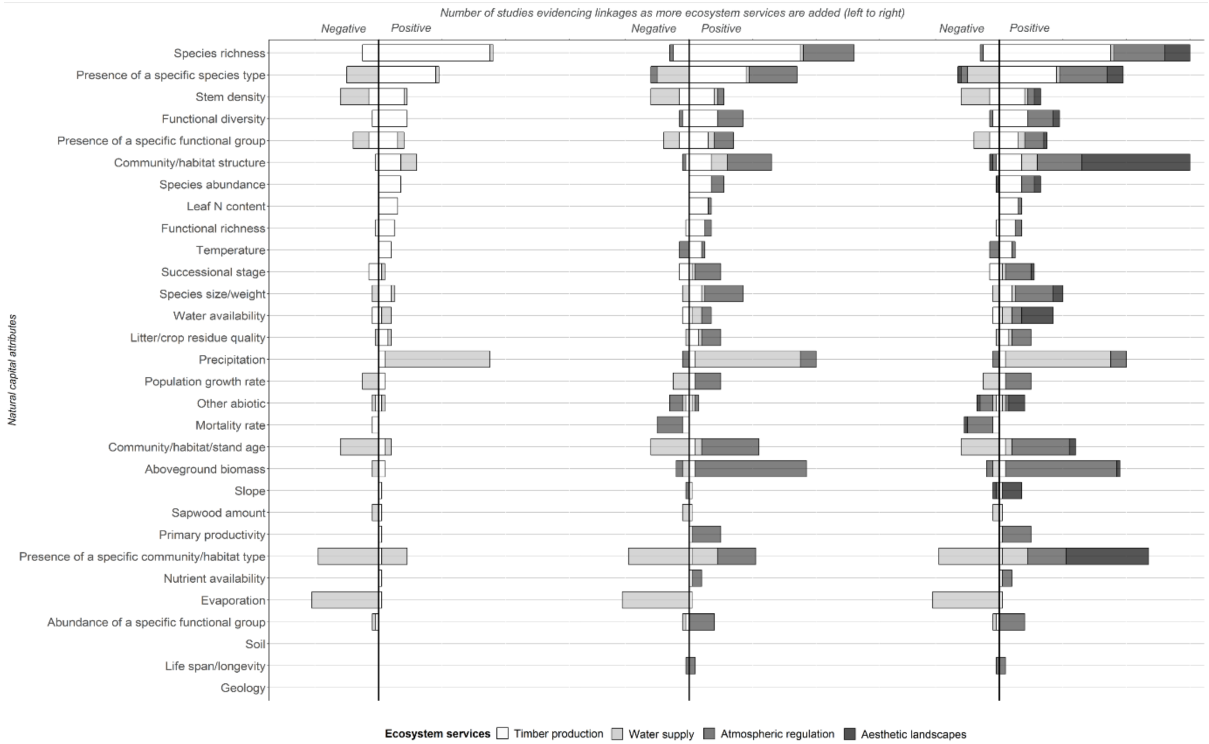


Figure 12: Mirrored stacked bar graphs of the unfiltered positive and negative NC-ES linkages with the ecosystem services: timber production, water supply, atmospheric regulation and aesthetic landscapes. Left to right shows how the overall evidence for each of the NC-ES linkages changes as more ecosystem services are added. Evidence for positive and negative NC-ES linkages are shown on the positive (maximum = 60) and negative (maximum = 21) part of the x-axis respectively. Natural capital attributes without evidence for linkages with timber production are not shown.

Figure 12 shows many trade-offs between the ecosystem services, the majority being with water supply. Stem density and presence of a species type have positive linkages with timber production, yet also have negative linkages with water supply, so enhancing these natural capital attributes could support timber production but degrade water supply. Furthermore, Figure 12 shows that community/habitat stand age leads to trade-offs between water supply and atmospheric regulation. For example, Webb et al. (2012) found a strong inverse relationship between community/habitat stand age and catchment streamflow and Zhao et al. (2009) found that carbon storage increased with stand age in Chinese forests from 4 to 21 years. Figure 12 also shows many synergies between timber production, atmospheric regulation and aesthetic landscapes, particularly for the natural capital attributes community/habitat structure, species richness and presence of a specific species type. Community habitat structure was particularly synergistic, changing from a natural capital attribute of mediocre importance to timber production to one of the most important natural capital attributes for multiple service provision of the four services the forest manager investigated.

However, many of the studies did not identify these synergies and trade-offs themselves. Only one study on timber production identified an interaction with water supply. Fifteen studies evidencing NC-ES linkages for atmospheric regulation identified an interaction with timber production, but only two for water supply. Studies evidencing NC-ES linkages for water supply identified 12 interactions with atmospheric regulation, and studies evidencing NC-ES relationships for aesthetic landscapes did not identify any interactions with any of the ecosystem services in Figure 12.

3.4.4 NC-ES linkages for one natural capital attribute

The forest manager is interested in how they can better manage specific natural capital attributes to deliver multiple ecosystem services. They use the LiNCAGES platform to investigate the linkages between stem density and all the other ecosystem services available in the platform, to assess the potential implications of changing stem density on the provision of these services.

Figure 13 shows the amount of evidence for positive and negative linkages between stem density and all 13 of the ecosystem services in the LiNCAGES platform.

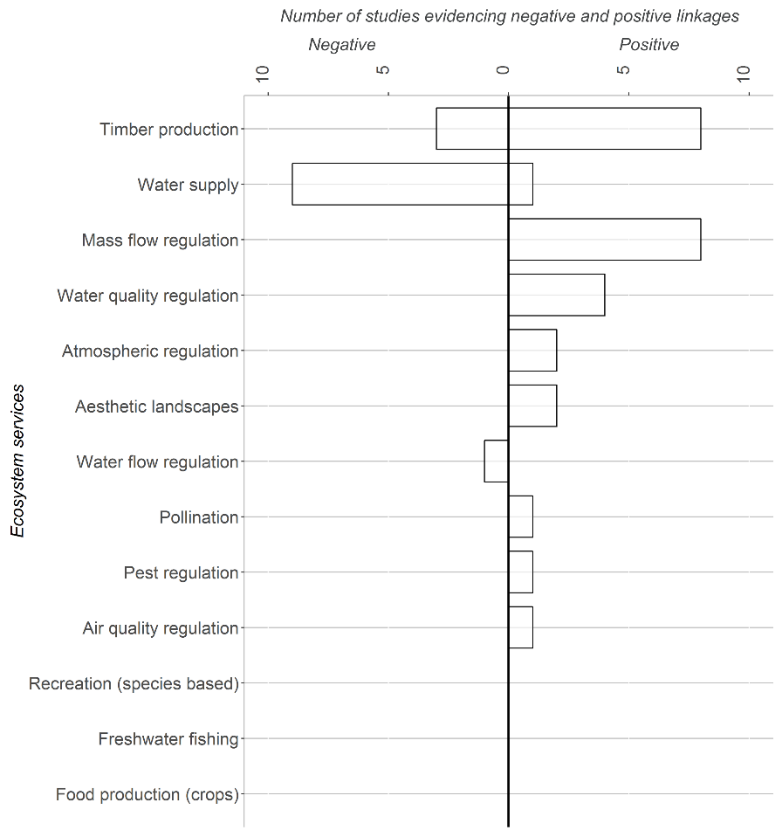


Figure 13: Mirrored bar graph of positive and negative ES-NC linkages with the natural capital attribute of stem density. The amount of evidence for positive and negative ES-NC linkages is shown on the positive (right) and negative (left) part of the x-axis respectively.

Stem density has the most evidence for positive linkages with timber production and mass flow regulation as shown by Figure 13. Higher stem density increases the productivity of forests (e.g. Amoroso and Turnblom, 2006) and reduces soil erosion (e.g. Lin et al., 2014). Stem density also has many negative linkages with water supply and water flow regulation as higher stem density results in significantly less soil water content (Kagawa et al., 2009; Zou et al., 2008).

3.4.5 Context dependency of ES-NC linkages for one natural capital attribute

The forest manager recalls the strong context dependency of NC-ES linkages with timber production (Figure 13). Therefore, before deciding whether to promote stem density, they use the LiNCAGES platform to investigate the context dependency of these ES-NC linkages (Figure 14).

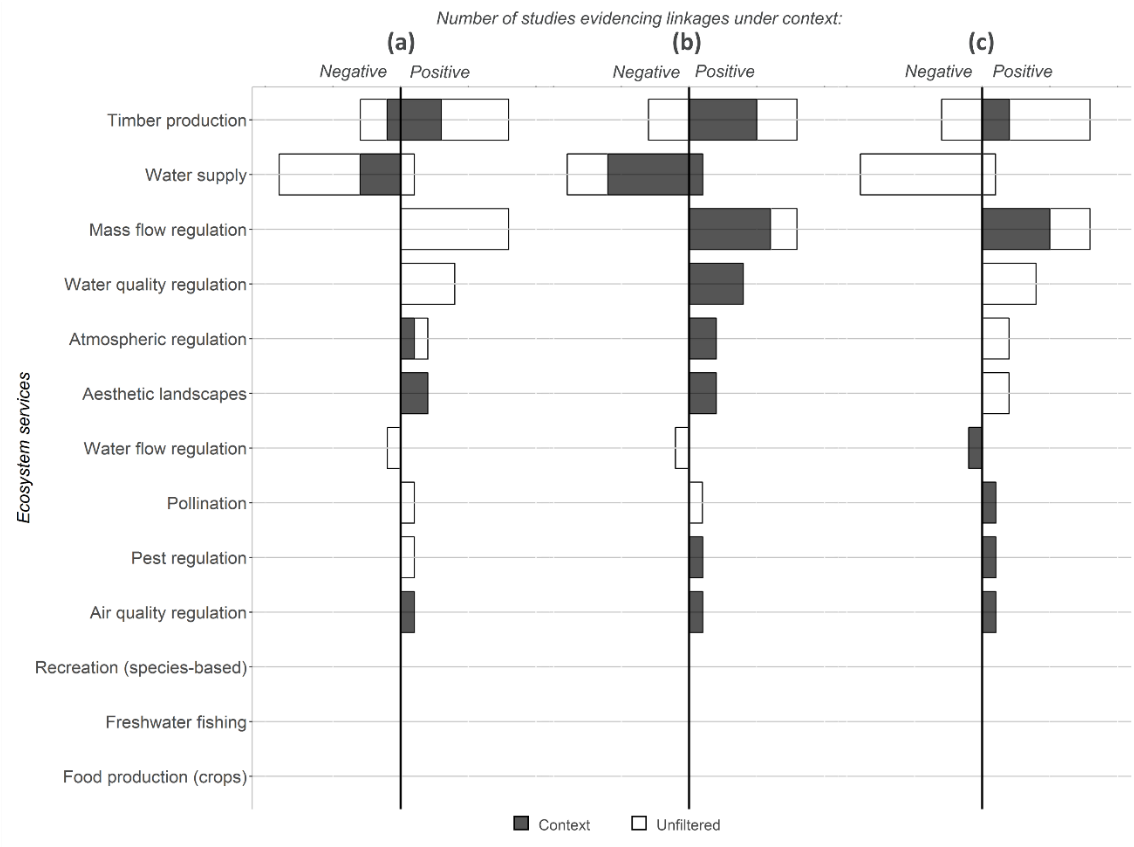


Figure 14: Mirrored overlay bar graph showing the positive and negative ES-NC linkages with the natural capital attribute of stem density filtered for three different contexts: (a) evidence from studies with spatial scales larger than local, (b) evidence from studies with a snapshot temporal scale and (c) evidence from studies undertaken in Europe. Evidence for positive and negative ES-NC linkages is shown on the positive (maximum = 8) and negative (maximum = 9) part of the x-axis, respectively. For comparison, for each context the filtered ES-NC linkages (dark grey) are overlaid onto the unfiltered ES-NC linkages (white) from Figure 6. See Figure B6 for Figure 13 and Figure 14 as radar plots.

Figure 14(a) shows that under the context of spatial scales larger than local, evidence is lost for ES-NC linkages between stem density and the five regulating ecosystem services: mass flow regulation, water quality regulation, water flow regulation, pollination and pest regulation. This highlights the spatial scale dependency of ES-NC linkages, with regulating services most affected. Figure 14(b) shows that snapshot spatial scale represents the unfiltered linkages well, only losing evidence for two ecosystem services (water flow regulation and pollination), which had very little evidence to begin with. The greatest context dependency of linkages with stem density comes from filtering for evidence from European studies. Figure 14(c) shows that under this context, there is no evidence for the strong trade-off between stem density and water supply, and evidence for positive linkages with water quality regulation, atmospheric regulation and aesthetic landscapes is lost. Furthermore, the amount of evidence for linkages with timber production is greatly reduced.

3.4.6 Applying multiple contexts through weighting

Using the knowledge gained throughout this process the forest manager can better decide on the weightings they will apply to the LiNCAGES platform to effectively deploy their context, while accounting for the context dependency of the NC-ES linkages. In this case, hierarchical weighting should be used rather than filtering, as the latter risks significantly reducing the sample size especially when using multiple study aspects simultaneously to apply context. See Appendix B Section 5 for a worked example using hierarchical weighting to investigate contexts (a), (b) and (c) simultaneously.

3.5 Discussion

3.5.1 Use case scenario

By applying the LiNCAGES platform to a hypothetical use case scenario, we illustrated the importance of context in understanding evidence about NC-ES linkages. Our hypothetical use case scenario journeyed through the decision-making process of a small-scale European forest manager using the LiNCAGES platform to explore the benefits, dependencies, synergies and trade-offs associated with the ecosystem service of timber production. The forest manager discovered which natural capital attributes have the most evidence for linkages with timber production, but also that the amount and direction of evidence for NC-ES linkages varies considerably with contexts such as spatial and temporal scale and study location. Other reviews also found strong context dependencies in NC-ES linkages in spatial scale (Cimon-Morin et al., 2013; Ricketts et al., 2016), temporal scale (Cimon-Morin et al., 2013) and location of study (Ricketts et al., 2016).

Our use case scenario has identified trade-offs and synergies that are missing in the literature. This evident lack of a holistic approach in the NC-ES studies reviewed in the OpenNESS database could lead to an underestimation of the value of multiple service provision (Balvanera et al., 2014). We also found these trade-offs and synergies to be context dependent. For example, at spatial scales larger than local, evidence is lost for linkages between stem density and five ecosystem services. This supports the findings of Duncan et al. (2015) who found that assessing NC-ES linkages at larger spatial scales misses key ecosystem functions that work at finer scales.

We demonstrated how multiple contexts can be applied simultaneously through the hierarchical weighting feature of the LiNCAGES platform (Appendix B Section 5). Weighting may be beneficial over filtering in such cases as it allows the user to give preference to certain contexts without filtering out potentially useful studies and considerably reducing the size of the evidence base. LiNCAGES gives responsibility for weighting to the user, allowing them to assign appropriate weights based on their expert knowledge and the specific purpose of their application. This is supported by the exploratory nature of the platform allowing users to quickly and transparently examine different weightings and to fully understand how each of their weighting choices contributes to the overall weight assigned to the study (Table B3). Nevertheless, the user should be cautious when interpreting the results of the weighted analysis; for this reason, LiNCAGES gives a clear indication of when weighting has been used, shown by the x-axis of Figures D3 and D4.

Finally, we have demonstrated LiNCAGES using one use case scenario, focusing on one ecosystem service and investigating three context dependent aspects. There are 13 ecosystem services and a further 13 context dependent aspects (Appendix B Section 4) available to investigate within LiNCAGES. As such, LiNCAGES can be applied to a diverse range of scenarios, e.g., ranging from a national policy-maker interested in understanding the potential impact of nature-based solutions on ecosystem services to a protected area manager who wishes to better understand how protecting certain natural capital attributes might affect the delivery of ecosystem services. Due to the high context dependencies we identified in this use case scenario, we expect other use case scenarios to produce different results according to the needs of the user.

3.5.2 Comparison to other tools and platforms

Most existing tools and platforms for ecosystem service assessment consist of models supported by data input; these are usually GIS based and use remotely sensed data sources (Vorstius and Spray, 2015). To the best of our knowledge, we are not aware of an ecosystem service assessment platform that exclusively uses literature-based evidence for investigating NC-ES linkages. The most similar tool is MESER (Managing Ecosystem Services Evidence Review; https://meser.simomics.com) which provides a searchable literature review on how habitats can be managed to enhance their delivery of ecosystem services. Unlike LiNCAGES, MESER is habitat specific and does not account for other context dependent aspects of its underlying studies.

3.5.3 Limitations and further work

As with any literature-based synthesis, the evidence behind the LiNCAGES platform is likely to include reporting bias as non-significant or less interesting results are less likely to be published (Ricketts et al., 2016; Schwarz et al., 2017). This reporting bias could lead to an underrepresentation of the amount of unclear NC-ES linkages between natural capital attributes and ecosystem services (Schwarz et al., 2017). Additionally, some natural capital attributes and ecosystem services are studied more than others (Adhikari and Hartemink, 2016; Balvanera et al., 2006; Hevia et al., 2017), leading to their potential over representation in the LiNCAGES platform. The OpenNESS database tried to overcome this by recording 60 articles per service (Pérez Soba et al., 2017).

Due to this reporting bias, we ensured that the user can view the number of studies available under their current filtering and weighting choices. This means the LiNCAGES platform can also be used to investigate the reporting bias in the NC-ES linkage literature. For example, Figure 9 showed that 12 of the natural capital attributes did not have evidence for NC-ES linkages with timber production and Figure 13 showed no evidence for NC-ES linkages between stem density and three ecosystem services. Investigating whether these missing NC-ES linkages are legitimate or due to reporting bias can form important research questions to better direct research into NC-ES linkages. Furthermore, the LiNCAGES platform can identify the amount of studies in particular contexts, directing researchers to study contexts that are underrepresented. For example, the LiNCAGES platform shows that very few studies with larger spatial scales than subnational provide evidence of NC-ES linkages with timber production. Similarly, the LiNCAGES platform can also investigate the reporting bias of the natural capital attributes. For example, Figure 11b shows that linkages between species abundance and timber production are evidenced only by studies with snapshot temporal scale.

When using the LiNCAGES platform the user should be aware that only the amount of evidence for a linkage is displayed, as the OpenNESS database did not consider the statistical significance or effect size of linkages, due to the diverse nature of the evidence using many incompatible indicators and approaches (Smith et al., 2017). Additionally, judgement was involved in assessing the direction of the linkage (Smith et al., 2017). Furthermore, pooling evidence can sometimes oversimplify and mislead both scientific syntheses and management interventions (Martnez-Harms and Balvanera, 2012; Ricketts et al., 2016) as definitions of the context dependent aspects can vary between the studies (Englund et al., 2017). For example, Englund et al. (2017) found that studies in different countries have different definitions of spatial scale. For these reasons, the LiNCAGES platform should be used only as a guide, or as a starting point, to improve understanding of the main NC-ES linkages in the literature before the user explores specific aspects of the literature in further detail themselves. We have ensured that the LiNCAGES platform is as transparent as possible to aid with this literature exploration.

Currently the evidence behind the NC-ES linkages in the LiNCAGES platform is exclusively based on the OpenNESS database so the findings may be sensitive to the search terms and search engines used to identify the relevant papers (Ricketts et al., 2016). A list of all search terms used to create the OpenNESS database is given in Appendix B Section 2. To reduce this dependency and to ensure the longevity of the LiNCAGES platform we plan to add the functionality for other researchers to continue to add studies to the LiNCAGES platform in a consistent way to build up knowledge, ensuring that the evidence base can evolve.

3.6 Conclusion

This study follows the development of the novel LiNCAGES platform, and its application to a hypothetical use case scenario. We created LiNCAGES to provide a system for investigating evidence for NC-ES linkages that allows the user to account for the context dependent and sometimes non-holistic nature of this type of evidence. Through the use case scenario, we demonstrated the capabilities and need for the LiNCAGES platform.

Decision-makers in policy, practice and business are increasingly aware of the need to manage natural capital sustainably, but they lack suitable tools and evidence to enable them to assess the impact of different management decisions. In particular, there is a lack of understanding on how natural capital assets influence the capacity of ecosystems to supply different services in specific contexts. The LiNCAGES platform can be used by stakeholders to raise awareness and build understanding of important NC-ES linkages and trade-offs and synergies between service provision under their own context, thus providing scientific evidence that is more salient to their needs. It can also provide a resource for researchers to identify key gaps in this evidence base and to work collaboratively to target and collate additional evidence that can strengthen the foundations of sustainable environmental management.

The LiNCAGES platform can be accessed at: https://shiny-apps.ceh.ac.uk/LiNCAGES/.

4 Effect of linkages between natural capital attributes on ecosystem service provision

George N. Linney, Lindsay C. Maskell, Peter A. Henrys, George A. Blackburn and Paula A. Harrison

This chapter is a replication of a paper that is under review with the journal *Ecosystems.*

4.1 Abstract

Preventing further loss of our vital ecosystem services (ES) requires understanding of linkages to their supporting natural capital attributes (NCA). However, NCA also interact with each other, for example soil carbon concentration is influenced by soil moisture content. These interlinkages may only occur under certain contexts. This study uses a large, yet fine grained monitoring dataset (Countryside Survey 2007) to evidence NCA-NCA interlinkages and combines it with literature-based information from the interactive visualisation platform (LiNCAGES) to reveal potential NCA-ES linkages, for 26 NCAs and 13 ESs.

Accounting for some NCA-NCA interlinkages identified new NCA-ES linkages. For example, accounting for NCA-NCA interlinkages with the NCA soil organic matter content revealed new indirect positive and negative NCA-ES linkages with air quality regulation, aesthetic landscapes, water supply, timber production, pest regulation and freshwater fishing. The presence of some NCA-NCA interlinkages varied with location. We found different NCA-NCA interlinkages in upland and lowland locations.

The importance of understanding both NCA-NCA interlinkages and NCA-ES linkages is illustrated through the scenario of an upland farmer wishing to promote soil carbon concentration to improve ES delivery. This approach provides a decision-maker with more comprehensive evidence of possible NCA-NCA interlinkages and NCA-ES linkages that may be important to their ES provision. It provides a starting point for further data collection or literature review to confirm whether the identified linkages are relevant and important causal

relationships under their context. This approach aims to raise awareness of the need to explore both NCA-NCA interlinkages and NCA-ES linkages before making decisions.

4.2 Introduction

Ecosystem services (ES) are vital for human existence and good quality of life (Díaz et al., 2019). These services are supported by their underlying natural capital (de Bello et al., 2010; Díaz et al., 2019; Harrison et al., 2014; Norton et al., 2016; Ricketts et al., 2016). Natural capital attributes (NCA; see Box 2 for definition) can be positively or negatively linked to the provision of ESs (Burkhard et al., 2012; Maseyk et al., 2017; Smith et al., 2017; Wu and Li, 2019). For example, changing the NCA soil nutrient status through promoting soil nutrient turnover often results in the release of carbon dioxide, enhancing the provision of the ES of crop production while diminishing the ES of carbon storage (Manning et al., 2018).

Box 2: Definitions of key terms used in this study.

Ecosystem service (ES): the contributions that ecosystems make to human well-being, and distinct from the goods and benefits that people subsequently derive from them. These contributions are framed in terms of 'what ecosystems do' for people (Haines-Young and Potschin, 2018).

Natural capital (NC): Natural capital encompasses the elements of nature that directly or indirectly produce value for people, including ecosystems, species, freshwater, land, minerals, air and oceans, as well as natural processes and functions (Pérez Soba et al., 2017)

Natural capital attribute (NCA): biotic and abiotic attributes of natural capital which affect service delivery (Smith et al. , 2017)

NCA-NCA interlinkages: when the promotion of one natural capital attribute will influence another natural capital attribute.

NCA-ES linkages: The direction (positive or negative) of influence a natural capital attribute has on ecosystem service delivery (Linney et al., 2020).

NCAs also interact with each other (Maskell et al., 2013; Norton et al., 2016). For example the NCA soil carbon concentration is influenced by the NCA soil moisture content (Falloon et al., 2011). As a result, accounting for NCA to NCA (NCA-NCA) interlinkages can reveal further indirect NCA to ES (NCA-ES) linkages. For example, as the NCA soil moisture content is interlinked with the NCA soil carbon content, and the NCA soil carbon content is positively linked to the provision of the ES atmospheric regulation (Wiesmeier et al., 2019), by accounting for NCA-NCA interlinkages we find that increasing the NCA soil moisture content will indirectly lead to the provision of the ES atmospheric regulation. These NCA-NCA interlinkages and NCA-ES linkages can be visually represented as shown by the hypothetical schematic in Figure 15a showing all possible NCA-NCA interlinkages and NCA-ES linkages between three NCAs and three ESs.

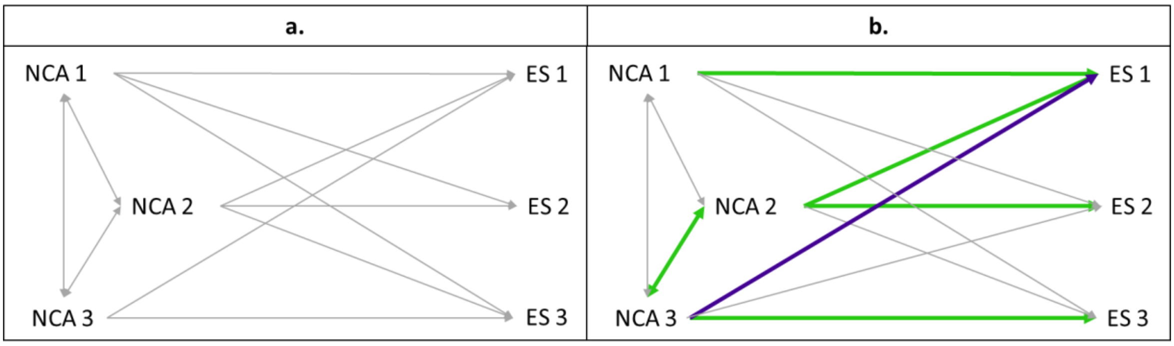


Figure 15: Hypothetical schematic illustrating possible relationships between three NCA and three ESs: (a) shows all possible NCA-ES linkages and NCA-NCA interlinkages, (b) highlights NCA-NCA interlinkages and NCA-ES linkages for which evidence has been found to support their existence with the colour indicating the direction (positive (green), negative (purple)). [1]

When making decisions for sustainable ES delivery it is important to identify both NCA-NCA interlinkages and NCA-ES linkages to allow for better identification of trade-offs between ESs and reduce the occurrence of ecological surprises (Bennett, 2017; Burkhard et al., 2012; Manning et al., 2019; Norton et al., 2016; Renard et al., 2015). For example, if a decision-maker were to identify evidence for the NCA-NCA interlinkages and NCA-ES linkages as shown by the highlighted arrows in Figure 15b they would discover that NCA 2 positively effects the provision of both ES 1 and ES 2. Based on this evidence alone, the decision-maker may believe that NCA 2 is the best NCA to promote for multiple ES delivery. However, Figure 15b also shows evidence that NCA 2 is positively related to NCA 3, and that NCA 3 negatively effects the provision of ES 1. Based on this additional evidence, if the provision of ES 1 is a key priority for the decision-maker they may consider promoting NCA 1, rather than NCA 2, as it does not lead to negative NCA-ES linkages with ES 1.

Evidence for NCA-NCA interlinkages and NCA-ES linkages is scarce and where it does exist it is highly fragmented and context-specific (Adhikari and Hartemink, 2016; Gutierrez-Arellano and Mulligan, 2018; Harrison et al., 2014; Linney et al., 2020; Maskell et al., 2013). Previous studies have assessed NCA-NCA interlinkages using monitoring or remotely sensed data. However, the primary goal of these studies was to assess trade-offs and synergies between ES, with NCAs being used as proxies to represent ESs (e.g. Le Clec'h et al., 2018; Maskell et al., 2013; Qin et al., 2015; Sylla et al., 2020)

Several studies have assessed NCA-ES relationships showing that one NCA can underpin multiple ESs and one ES can be influenced by multiple NCAs (de Bello et al., 2010; Harrison et al., 2014; Smith et al., 2017). For example, the NCA tree species richness in production forests shows positive to positively hump-shaped relationships with multiple ESs, including production of tree biomass, soil carbon storage, berry production and game production potential (Gamfeldt et al., 2013). Evidence for links between an NCA and multiple ESs can also be obtained by methods such as literature synthesis (e.g. Balvanera et al., 2006; Hevia et al., 2017a; Ricketts et al., 2016; Schwarz et al., 2017; Smith et al., 2017). However, literature synthesis studies investigating NCA-ES linkages do not typically account for NCA-NCA interlinkages.

Both NCA-NCA interlinkages and NCA-ES linkages can vary with context (Andersson et al., 2015; Linney et al., 2020; Manning et al., 2019; Maskell et al., 2013; Norton et al., 2016). This is important for decision-makers as NCA-NCA interlinkages and NCA-ES linkages observed under a specific context may not be present under the decision-maker's context. For example, fruit production in an orchard depends on the presence of pollinators, but the pollinators themselves are influenced by factors located outside the orchard such as additional habitat resources, potential for meta-population dynamics, and additional nectar sources (Andersson et al., 2015).

Gaining access to available evidence for NCA-NCA interlinkages and NCA-ES linkages can be difficult for decision makers due to the costly nature of literature reviews and monitoring programs (Linney et al., 2020). Therefore, there is a strong need for the operationalisation of the identification of evidence for the NCA-NCA interlinkages and NCA-ES linkages (as represented by Figure 15b) using data that is accessible and available to decision makers.

In this study we attempt to further understand the evidence supporting both NCA-NCA interlinkages and NCA-ES linkages. We do this by joining together evidence from monitoring data on NCA-NCA interlinkages with evidence from literature synthesis on NCA-ES linkages. NCA-NCA interlinkages are evidenced using the Countryside Survey, a unique dataset that monitors ecological and land use change over Great Britain from 1978 to 2007 (Maskell et al., 2013; Norton et al., 2016). NCA-ES linkages are evidenced using a large literature synthesis from the operationalisation of natural capitaland ES (OpenNESS) database as it provides a recent, accessible and substantial evidence base pertaining to a wide range of ESs (13) and NCAs (42) (Pérez Soba et al., 2017). Firstly, we show how accounting for NCA-NCA interlinkages may reveal new NCA-ES linkages. Then we investigate the location context dependency of the NCA-NCA interlinkages and how this context can change the NCA-ES linkages identified. Finally we apply our approach to the hypothetical use case scenario of an upland farmer and discuss its utility.

4.3 Materials and methods

4.3.1 Evidence for NCA-NCA interlinkages

NCA-NCA interlinkages were identified using the Countryside Survey 2007 (CS2007) (http://www.countrysidesurvey.org.uk) monitoring dataset. CS2007 is a large-scale but fine-grained monitoring data set of co-located biophysical measurements across Great Britain (Maskell et al., 2013; Norton et al., 2012). CS2007 comprises 591 stratified, randomly selected 1km^2 squares. The stratification was based on a classification of all 1km^2 squares across Great Britain using their topographic, climatic and geological attributes (Reynolds et al., 2013; Smart et al., 2003). Within each 1km^2 square, five 200m^2 vegetation plots were located using a restricted random sample of fields and unenclosed land (Smart et al., 2003). Biophysical vegetation metrics were recorded, as well as soil samples taken from the top 15cm of soil in each vegetation plot (Smart et al., 2010). Some biophysical measurements were limited to the 1km^2 square scale such as broad habitat area, landscape metrics and freshwater samples (Maskell et al., 2013; Smart et al., 2003). CS2007 was chosen to evidence the NCA-NCA interlinkages as it uses a consistent methodology across the whole of Great Britain to record biophysical measurements (Norton et al., 2012) that can represent NCAs. Further, it was preferred over using remotely sensed data for identification of NCA-NCA interlinkages as remotely sensed data will not capture fine scale NCA-NCA interlinkages (Eigenbrod et al., 2010) such as the effect of small biotypes (Maskell et al., 2013). In addition to this, NCAs such as grassland and heath habitat area and vegetation structure can be difficult to classify using remotely sensed data (Maskell et al., 2019).

4.3.2 Evidence for NCA-ES linkages

NCA-ES linkages were evidenced using the Linking NCA Groups to ESs (LiNCAGES) platform (Linney et al., 2020). LiNCAGES presents literature-based evidence for NCA-ES linkages between 42 NCAs and 13 ESs from 780 peer-reviewed papers published in the English language from the Operationalisation of natural capital and ESs (OpenNESS) database (Pérez Soba et al., 2017). The ESs included are four provisioning ESs (food production (crops), freshwater fishing, timber production and water supply); seven regulating ESs (air quality regulation, atmospheric regulation (carbon sequestration), mass flow regulation (erosion protection), water quality regulation (water purification), water flow regulation (flood protection), pollination and pest regulation); and two cultural ESs (species-based recreation and aesthetic landscapes). The literature evidence is gathered from a vote counting approach where the literature reviewer recorded the direction (positive, negative or unclear) of evidence for NCA-ES linkages mentioned in each paper. To maximise the amount of evidence available for NCA-ES linkages the literature base was not filtered by any context for this analysis. See Linney et al. (2020) for investigation of the context dependency of NCA-ES linkages through the creation of the LiNCAGES platform.

4.3.3 Connecting CS2007 biophysical measurements to LiNCAGES NCAs

To combine evidence on NCA-NCA interlinkages and NCA-ES linkages, we identified as many biophysical measurements from CS2007 that could represent NCAs in LiNCAGES. Some of the

NCAs in LiNCAGES such as "presence of a specific species type" were too broad to be represented by CS2007 biophysical measurements; others such as "stem density" were not recorded in CS2007. Therefore, this analysis only includes NCAs from LiNCAGES that had the following criterion: specific enough to be effectively represented by a single CS2007 biophysical metric, which has data available for at least 50% of the CS2007 squares.

Six NCAs in the LiNCAGES platform could be directly represented by CS2007 biophysical measurements (shown in bold in Table 4). See Table C1 for all the NCAs available in the LiNCAGES platform and whether they met the criteria to be included in this analysis. Three NCAs (community/habitat area, presence of a specific functional group and soil) were broken down into more specific NCAs. See Table C2 for this breakdown shown in tabular format. The community/habitat area NCA was broken down through filtering the NCA-ES linkages between community habitat area and all 13 ESs by ecosystem type (classified in the OpenNESS database). However, as some studies span multiple ecosystems this did lead to double counting of NCA-ES linkages with some habitat area attributes. The NCAs "presence of a specific functional group" and "soil" were broken down through further interrogation of the literature-based evidence behind the LiNCAGES platform. This information was extracted through interpretation of the reviewer's comments associated with the evidenced NCA-ES linkages in the OpenNESS database. See Table C3 for an example of how the literature-based evidence underlying the LiNCAGES platform was interrogated to break down a broad NCA (soil). Soil pH was broken down into the separate NCA pH (low) and pH (high) (see Table 4) so that a NCA-NCA interlinkages with acidic and alkali soils could be identified. The breakdown of these broad NCAs gave a total of 26 NCAs that could be represented by CS2007 biophysical measurements (shown in Table 4). See Table C4 for an expanded version of Table 10 including detailed description of the CS2007 data used to represent the NCAs.

Table 4: The CS2007 data representing the NCAs including the spatial scale at which the data is measured and the number of linkages of positive and negative that the NCA has with all 13 ESs. See Table C4 for a more detailed version of Table 1. [1]

NCA	Number of positive and negative linkages	CS2007 data description	Unit	Spatial scale (km^2)	Evidence for suitability of representation (references)
Landscape diversity	26	Shannon index of diversity for habitats surveyed.	-	1	(Nagendra, 2002)
Primary productivity	23	Cover-weighted specific leaf area.	$gm^{-2}d^{-1}$ (Grams per meter squared per day)	0.2	(Garnier et al., 2004)
Slope	34	Slope of the surveyed plot.	Ordinal: 1 = flat, 2 = slight, 3 = moderate and 4 = steep	0.2	-
Species richness	182	Plant species richness.	Species/area	0.2	(Smart et al., 2003)
Successional stage	31	A detrended correspondence analysis axis correlated with light availability and in turn correlated with successional stage and disturbance.	Detrended correspondence analysis axis 2 score	0.2	(Smart et al., 2003)
Water quality	18	Average Score per Taxon for macroinvertebrates in headwater streams.	-	1	(Dunbar et al., 2010)
Cropland habitat area	76	Broad habitat area of arable.	% cover	1	(Smart et al., 2003)
Grassland habitat area	72	Broad habitat area of grassland (acid, improved, neutral and calcareous).	% cover	1	(Smart et al., 2003)
Heathland and shrub habitat area	32	Broad habitat area of shrub heath and bracken.	% cover	1	(Smart et al., 2003)
Rivers and lakes habitat area	31	Broad habitat area of open water and river.	% cover	1	(Smart et al., 2003)
Sparsely vegetated land habitat area	22	Broad habitat area of montane and inland rock.	% cover	1	(Smart et al., 2003)
Wetlands habitat area	38	Broad habitat area of bog and fen.	% cover	1	(Smart et al., 2003)
Woodland	137	Broad habitat area of conifer and	% cover	1	(Smart et al.,

and forest habitat area		broadleaf.			2003)
Birds	4	Total cover of plant species surveyed that are important for the diet of birds.	% cover*	0.2	(Smart et al., 2000)
Broadleaf trees	10	Broadleaf broad habitat area.	% cover	1	(Smart et al., 2003)
Canopy cover	10	Total cover of all plant species surveyed that have a low preference for light (have Ellenberg L score of between 5 and 1).	% cover*	0.2	(Hill et al., 2000)
Coniferous trees	12	Broad habitat area of conifer.	% cover	1	(Smart et al., 2003)
Herbaceous plants and grass	15	Total cover of all plant species surveyed that are forbs or grasses.	% cover*	0.2	-
Nitrogen fixers	24	Total cover of all plant species surveyed that are nitrogen fixers.	% cover*	0.2	-
Pollinators	32	Total cover of all plant species surveyed that were classified as nectar plants for bees.	% cover*	0.2	(Smart et al., 2010)
Soil organic matter content	8	Loss on ignition.	g/kg	0.2	(Reynolds et al., 2013)
Soil carbon concentration	6	Soil carbon concentration of loss on ignition.	g/kg	0.2	(Reynolds et al., 2013)
Soil bulk density	5	The bulk density of the soil sample.	g/cm^3	0.2	(Reynolds et al., 2013)
Soil moisture content	13	The moisture loss at each stage of the process for estimating bulk density was used to estimate the initial moisture content of the soil and from that the initial dry weight of the soil.	%	0.2	(Reynolds et al., 2013)
Soil pH (high)	3	Extent that pH of fresh soil (0-15cm) is greater than neutral (7) (soil pH - 7) if soil pH >7.	-	0.2	(Reynolds et al., 2013)
Soil pH (low)	3	Extent that pH of fresh soil (0-15cm) is less than neutral (7) (7 - soil pH) if soil pH <7.	-	0.2	(Reynolds et al., 2013)

4.3.4 Analysis and clustering

We identify NCA-NCA interlinkages by clustering those NCA that are positively correlated together making the assumption that all NCAs within each cluster have synergistic NCA-NCA interlinkages. This could lead to the identification of NCA-NCA interlinkages that are not causally linked. However, as the aim of this study was to identify all possible NCA-NCA interlinkages, we prioritised the inclusion of a wide variety of NCAs, recognising that further research may be needed to test whether some positive correlations are causal relationships.

To accomplish this, for each NCA we first normalised the data with respect to the maximum and then used a multivariate analysis of the spatial relationships between the NCAs represented in Table 4, where the NCAs are variables and the CS2007 squares are samples. An unconstrained ordination was carried out using Detrended Correspondence Analysis (DCA) (Hill and Gauch, 1980) (decorana function from the R package Vegan (Oksanen et al., 2019)). We used a DCA for this kind of analysis as it makes fewer assumptions and is less restrictive than other multivariate techniques such as a Principle Component Analysis. DCA also allows for the inclusion of ordinal variables such as the NCA of slope (Table 4). The "species scores" the DCA assigns to each of the NCAs were then clustered into mutually exclusive groups using the k-means clustering algorithm (core R function (R Core Team, 2020)). The NCAs were clustered relative to four DCA axis (as the decorana function is limited to four DCA axis). See Box C1 for how we accounted for the random number dependence of the k-means algorithm in generating cluster centres.

We first completed the DCA and clustering for all the NCAs in Table 4 at 1km^2 spatial scale. For NCAs recorded at the 200m^2 plot scale (see scale column of Table 4), all plots in the 1km^2 square were averaged to give a single value for each 1km^2 square. To be included in the DCA, each square must have data for all the NCAs shown in Table 4. Nine of the 591 CS2007 squares could not be used for this reason and are not included in this analysis (shown in red in Figure 16). As a result, this analysis uses data from 582 CS2007 squares.

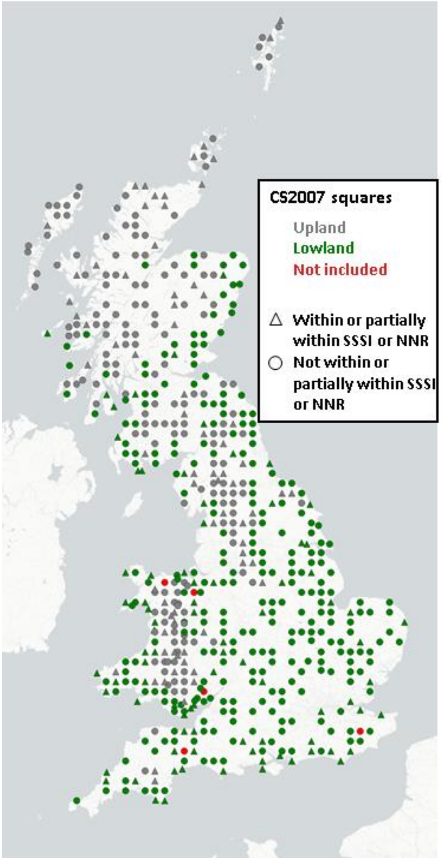


Figure 16: Map showing the location and context of CS2007 1km2 squares as points and whether they are included in this analysis. Upland squares are shown in grey (233), lowland in green (349) and not included in red (9). Triangular points are located within or partially within an SSSI or NNR (154) and circular points are not. [2]

We identified seven clusters of NCAs as being most appropriate for this analysis through visual assessment of the DCA using a Grand Tour (Cook et al., 1995) as shown by (Figure C1) created using the R package Tourr (Wickham et al., 2011). A Grand Tour shows a smooth sequence of projections of high-dimensional data and was appropriate for this analysis as it assists in clustering data when clusters are oddly shaped and in finding general low-dimensional structure in high-dimensional, and in particular, sparse data (Wickham et al., 2011). The Grand Tour allowed for the viewing of all four DCA axes interactively. This was important as the first two DCA axes were found to only represent 32.8% of the variance. Seven clusters were also supported by the generation of an elbow plot, showing how the total within cluster sum of squares changes with number of clusters (Figure C2). A Shiny app was created for exploring this clustering, further details are given in Box C2. The clusters of synergistic NCAs were then linked to 13 ESs using literature-based evidence from the LiNCAGES platform. As the focus of this research was on NCA-NCA interlinkages, if a cluster contained only one NCA this cluster was not investigated further as it does not show any NCA-NCA interlinkages. However as we wanted to represent as many NCA as possible these NCA were kept in the analysis; due to the nature of multivariate analysis their presence will impact the clustering of the other NCA.

The effect of location context on the clustering of NCA-NCA linkages was investigated by filtering CS2007 squares according to the following scenarios: (a) $1km^2$ spatial scale for all CS2007 squares (used as the control scenario), (b) $1km^2$ spatial scale for upland squares, (c) $1km^2$ spatial scale for lowland squares and (d) $1km^2$ squares that are within or partially within protected areas (Site of Special Scientific Interest (SSSI) or National Nature Reserve (NNR)). The location context of the CS2007 $1km^2$ squares is shown in Figure 16. See Table C5 for further details on how these contexts were derived. We then repeated the DCA and k-means clustering to seven clusters on the data from the filtered CS2007 squares for each context.

4.4 Results

The NCA-NCA interlinkages identified through clustering the NCAs at $1km^2$ scale into seven clusters are shown in Table 5. Table 5 also shows the number of different ES positively or negatively linked with the NCA (when not accounting for NCA-NCA interlinkages) compared to the number of different ES positively or negatively with each cluster of NCAs (when accounting for NCA-NCA interlinkages).

Table 5: Clustering of the NCAs across the four DCA axes representing 45.7% of the variation. The number of different ES positively or negatively linked to each NCA and each cluster of NCA is shown. One cluster contained only the NCA water quality so water quality was not investigated further, therefore only six clusters are shown Table 5. The NCAs of interest explored in Figure 17 are shown in bold italic. [2]

Cluster	Cluster name	NCA	Number of different ES (positively or negatively) linked to: NCA	Cluster of NCA
1	Wet soil	***Soil carbon concentration***	2	11
		Soil moisture content	2	
		Soil organic matter content	5	
		Soil pH (low)	2	
		Wetlands habitat area	8	
2	Diversity	Birds	2	13
		Grassland habitat area	11	
		Herbaceous plants and grass	3	
		Nitrogen fixers	4	
		Landscape diversity	5	
		Primary productivity	7	
		Slope	9	
		Successional stage	9	
		Species richness	12	
3	Upland	***Heathland and shrub habitat area***	10	10
		Sparsely vegetated land habitat area	3	
4	Agricultural production	Soil bulk density	4	13
		Cropland habitat area	12	
		Soil pH (high)	2	
5	Woodlands and water	Pollinators	1	12
		Broadleaf trees	6	
		Canopy cover	5	
		Rivers and lakes habitat area	6	
6	Coniferous woodland	Coniferous trees	4	10
		Woodland and forest habitat area	10	

4.4.1 Identified NCA-NCA interlinkages

Table 5 shows that many of the soil NCAs cluster together with wetlands habitat area (cluster 1). The clustering of soil moisture content with wetlands habitat area is anticipated. Relatively low pH benefits the accumulation of organic matter (Zhou et al., 2019), supporting the clustering of soil pH (low) with soil organic matter content. Cluster 2 shows that there are typically many interlinkages between NCAs that are associated with high biodiversity such as successional stage, species richness and landscape diversity. Cluster 3 features the similar habitat areas heathland and shrub and sparsely vegetated land which are commonly associated or interlinked (Zaghi, 2008). Cluster 4 contains NCAs associated with agricultural production, and Cluster 5 links woodlands to pollinators. Alison et al. (2021) found woodlands and bee pollinators to be associated but not causally linked. Cluster 6 is a smaller cluster of the NCAs woodland and forest habitat area and presence of coniferous woodland. This cluster may have arisen from double counting; Table 4 shows that woodland and forest habitat area is represented by areas of both coniferous broad habitat and broadleaf broad habitat. Coniferous trees are also represented by coniferous trees broad habitat area. Water quality is in a cluster of its own. This could be due to the lack of water bodies from which water quality measurements could be assessed; 463 out of 582 squares did not contain water bodies. Water quality was assigned a value of zero for CS2007 squares that did not feature a water body to enable the inclusion of the attribute within the multivariate analysis.

4.4.2 New NCA-ES linkages identified from NCA-NCA interlinkages

We demonstrate the importance of accounting for NCA-NCA interlinkages through exploring four NCAs: soil pH (high), heathland and shrub, landscape diversity and soil carbon concentration. For each NCA, we compare the NCA-ES linkages for only the NCA of interest to the NCA-ES linkages between all of the NCAs that share a cluster with the NCA of interest (Figure 17). For this comparison we use radar plots of differing scales as we are more interested in the presence of an NCA-ES linkage than the number of studies that identify an NCA-ES linkage. This is due to literature bias in NCA-ES linkages as some NCAs, e.g. species richness, are studied more than others (Linney et al., 2020). See Figure C3 for the radar plots in Figure 17 displayed on the same scale by NCA.

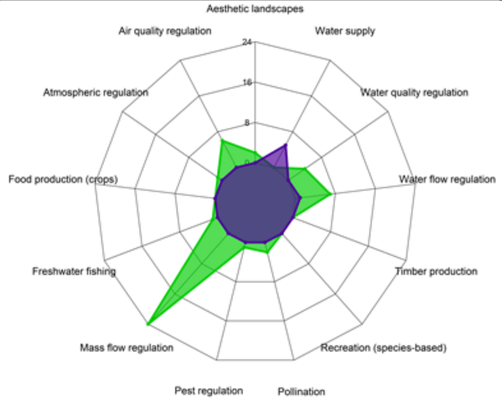
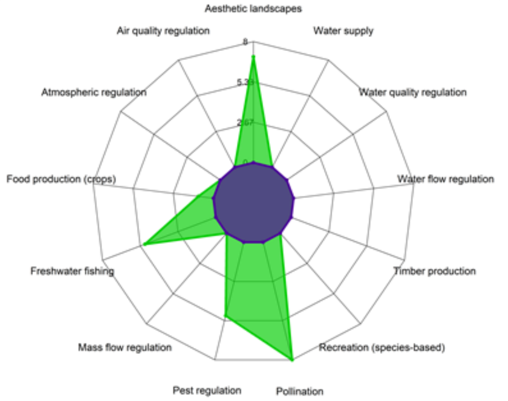
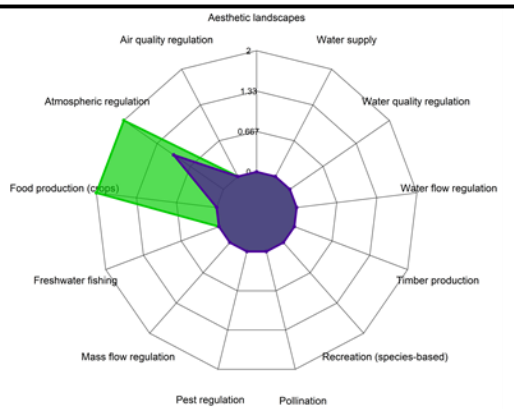
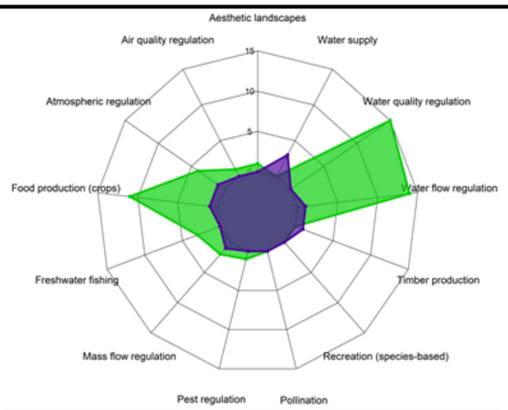
<table>
<tr><th>NCA</th><th>Linkages between the NCA of interest and all ESs</th><th>Linkages between all NCA clustered with the NCA of interest and all ESs</th></tr>
<tr><td>Heathland and shrub habitat area</td><td>
</td><td>
</td></tr>
<tr><td>Soil pH (high)</td><td>
</td><td>
</td></tr>
<tr><td>Landscape diversity</td><td>
</td><td>
</td></tr>
<tr><td>Soil carbon conc.</td><td>
</td><td>
</td></tr>
</table>

Figure 17: Radar plots comparing the evidence for NCA-ES linkages with only the NCA of interest to the NCA-ES linkages for all of the NCAs that share a cluster with the NCA of interest. The radar plot scale is

the number of studies evidencing a NCA-ES linkage. The direction of the NCA-ES linkages is coloured in the plot: positive (green) and negative (purple). All radar plots are scaled to their maximum number of studies evidencing a NCA-ES as we are more interested in the presence of a NCA-ES linkage rather than the number of studies evidencing it. For the radar plots in Figure 17 scaled consistently by NCA see Figure C3. [3]

Figure 17 shows that accounting for NCA-NCA interlinkages for the NCA of heathland and shrub habitat area does not reveal any new NCA-ES linkages, just greater evidence for the same NCA-ES linkages. This suggests that both NCAs in cluster 3 (heathland and shrub habitat area and sparsely vegetated land habitat area) have very similar NCA-ES linkages.

However, this is not the case for the other NCAs in Figure 17. Taking account of clustered NCA-NCA linkages for soil pH (high) reveals positive linkages with 12 additional ESs. These positive NCA-ES linkages are a result of NCA-NCA interlinkages with cropland habitat area, as are the negative NCA-ES linkages with water quality regulation. This is surprising as agroecosystems can have lower soil pH due to nitrogen fertiliser input (Dai et al., 2020).

Landscape diversity shows positive linkages to four ESs when NCA-NCA interlinkages are not considered, yet positive NCA-ES linkages to all 13 ESs when NCA-NCA linkages are considered. This is mostly due to the inclusion of species richness in the cluster, which has well established positive NCA-ES linkages to all ESs (Mace et al., 2012). Many studies have found evidence for positive associations between landscape diversity and species richness (e.g. Hall et al., 2020; Heidrich et al., 2020; Stein et al., 2014). The positive NCA-ES linkages with water flow regulation arise from clustering with grassland habitat area, also observed by Bengtsson et al. (2019). The negative NCA-ES linkages within the overall positive NCA-ES linkage with timber production (Figure 17) are due to uncertainty in the literature as to whether mixed forests or monocultures are best for the provision of timber production (Linney et al., 2020).

Accounting for NCA-NCA interlinkages with soil carbon concentration reveals new positive NCA-ES linkages with water quality regulation, water flow regulation and freshwater fishing due to NCA-ES linkages with wetlands habitat area. Mitsch et al. (2015) also identified NCA-ES linkages between soil carbon concentration and these three ecosystem services. However, we also find that the NCA-NCA interlinkages with wetlands are responsible for negative NCA-ES linkages with water supply.

4.4.3 Location context dependency of NCA-NCA interlinkages

We demonstrate how changing location context can influence the NCA-NCA interlinkages using the example NCA of soil carbon concentration (Table 6). See Table C6, Figure C4 and Box C3 for an alternative example following the NCA of landscape diversity. A decision-maker, e.g. an upland farmer, may be interested in the likelihood of the NCA-NCA interlinkages with soil carbon concentration identified in Table 6 occurring under their context. Table 6 shows how the clustering of the NCAs with soil carbon concentration changes for the location contexts: upland, lowland and SSSI/NNR CS2007 squares.

Table 6: NCA clusters containing soil carbon concentration under each location context. Clustering across four DCA axes representing 45.7%, 37.4%, 35.3% and 43.3 % of the variation for contexts a, b, c and d respectively. [3]

NCAs clustered with soil carbon concentration under context:			
(a) Ignoring context dependency	**(b) Upland**	**(c) Lowland**	**(d) SSSI/NNR**
Soil carbon concentration	Soil carbon concentration	Soil carbon concentration	Soil carbon concentration
Soil moisture content	Soil moisture content	Soil moisture content	Soil moisture content
Soil organic matter content	Soil organic matter content	Soil organic matter content	Soil organic matter content
Soil pH (low)	Soil pH (low)	Herbaceous plants and grass	Soil pH (low)
Wetlands habitat area	Woodland and forest habitat area	Nitrogen fixers	Wetlands habitat area
	Coniferous trees	Pollinators	
		Birds	
		Grassland habitat area	
		Species richness	

When considering only CS2007 squares in upland areas (context (b)), soil carbon concentration has NCA-NCA interlinkages with woodland and forest habitat area. This may be due to the afforestation of peat, which is now actively discouraged as soil carbon losses due to drainage often exceed gains in tree biomass, producing a net increase in carbon emissions (Reed et al., 2009). When considering only lowland CS2007 squares (context (c)) NCA-NCA interlinkages between soil carbon concentration and species richness are observed. Chen et al. (2018) found that higher plant species richness leads to greater aboveground net primary productivity and belowground biomass, resulting in consistent positive effects on soil organic carbon storage. Furthermore, Anacker et al. (2021) and Cole et al. (2019) found positive interlinkages between soil carbon concentration and plant species richness in grasslands (which are typically present in lowlands).

Clustering when using data from CS2007 squares located within or partially within SSSI/NNR is identical to the clustering when ignoring context dependency, suggesting that the NCA-NCA interlinkages with soil carbon concentration identified in CS2007 squares within or partially within SSSI or NNR represent the NCA-NCA interlinkages with soil carbon concentration for the whole of Great Britain. Consequently, Figure 18a is identical to Figure 18d.

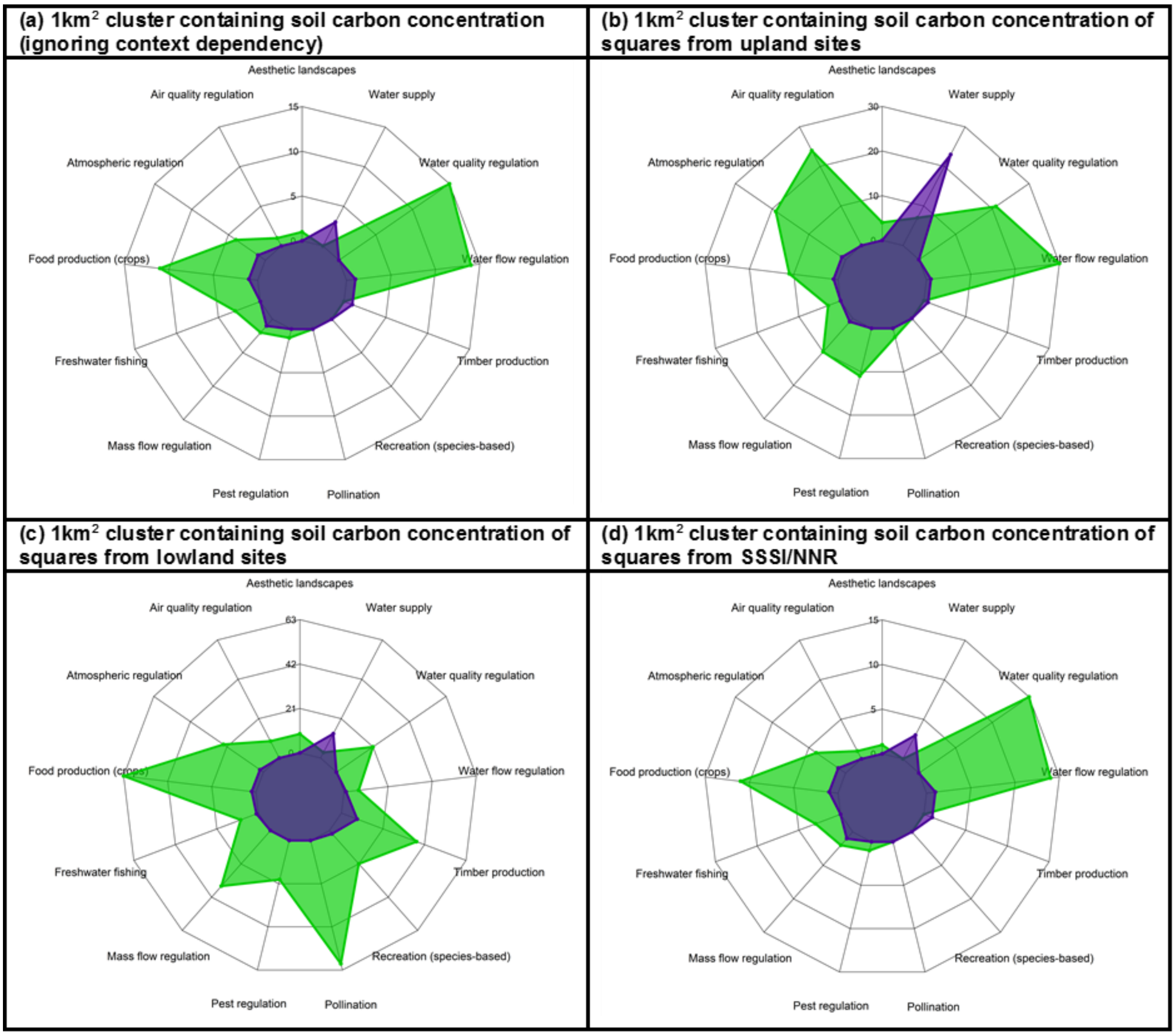


Figure 18: Radar plots showing how the evidence for NCA-ES linkages changes for each cluster containing the NCA of soil carbon concentration under the three location contexts. The radar plot scale is the number of studies evidencing a NCA-ES linkage. The direction of the NCA-ES linkages is coloured in the plot: positive (green) and negative (purple). All radar plots are scaled to their maximum number of studies evidencing a NCA-ES as we are more interested in the presence of a NCA-ES linkage rather than the number of studies evidencing it. For the radar plots in Figure 18 scaled to the same scale see Figure C5. [4]

Upland context reveals new negative NCA-ES linkages with water supply and new positive NCA-ES linkages with air quality, atmospheric regulation, food production, mass flow regulation and pest regulation (Figure 18b). All new NCA-ES linkages are due to NCA-NCA interlinkages with woodland and forest habitat area and coniferous trees (shown in Table 6). Air quality, atmospheric regulation and mass flow regulation are well known ESs provided by forest habitat area and coniferous trees, as are the negative linkages with water supply (Bullock et al., 2014). The NCA-ES linkages between woodland habitat area and pest regulation are less common but arise in studies such as Boccaccio & Petacchi (2009) who found woodland plays a role in enhancing parasitoid activity on the olive fruit fly. Lowland context results in new positive linkages with all 13 ESs (as shown by Figure 18c). This is mostly due to the inclusion of species richness in the cluster (shown in Table 6).

4.5 Discussion

This study has brought together evidence for NCA-NCA interlinkages and NCA-ES linkages to enable a more comprehensive understanding of the relationships between NC and ESs. Our results show that combining a large-scale but fine-grained monitoring data set (CS2007) to evidence NCA-NCA interlinkages with a large literature synthesis (LiNCAGES) helps to reveal potential new NCA-ES linkages. We identified possible new positive and negative NCA-ES linkages for three out of the four NCAs explored (soil pH (high), landscape diversity and soil carbon concentration). Whether these potential new NCA-ES linkages reflect causal relationships requires further study, underscoring the need for further monitoring or experimental studies on these NCA-ES linkages. Nevertheless, the illustrations highlight the importance of considering both NCA-NCA interlinkages and NCA-ES linkages concurrently in decision-making related to sustainable ES delivery.

To demonstrate the additional understanding gained by combining evidence for NCA-NCA interlinkages and NCA-ES linkages, the hypothetical schematic from Figure 15 has been populated with the evidence discovered in this study. The specific scenario here is that of an upland farmer interested in understanding the consequences of managing for soil carbon concentration on the provision of the ESs they may value (Figure 19).

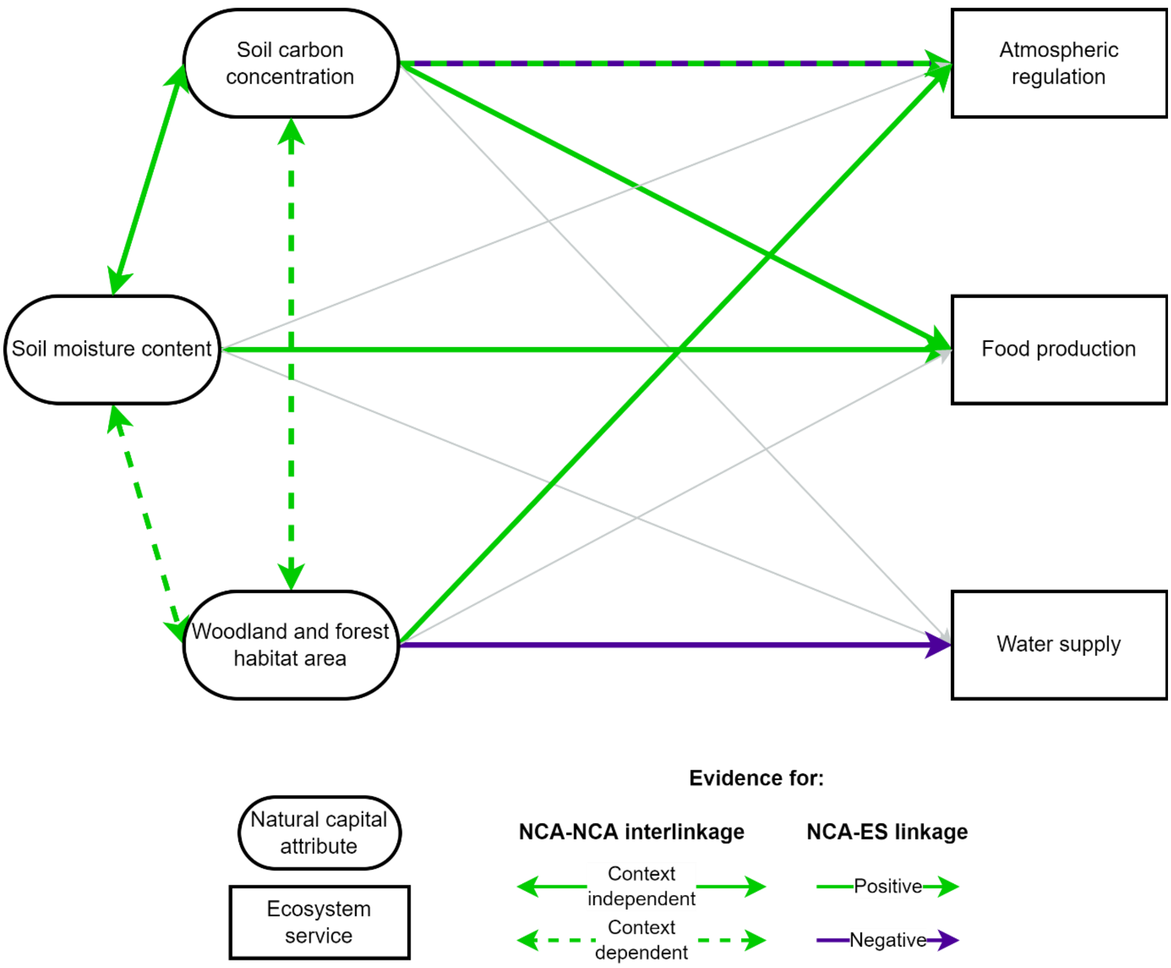


Figure 19: Hypothetical schematic from Figure 15 populated with evidence for NCA-NCA interlinkages and NCA-ES linkages discovered in this study. Dashed arrows show context dependent NCA-NCA interlinkages (in this case the dashed NCA-NCA interlinkages only occur in upland context). Green and purple dashed arrows show NCA-ES linkages that have evidence for both positive and negative linkages. Thin grey arrows show possible NCA-NCA interlinkages and NCA-ES linkages that we did not find evidence for. All 26 NCA and all 13 ESs are not included in this Figure for clarity. See Figs. S6a–h for the inclusion of all 26 NCAs and 13 ESs. [5]

Figure 19 shows that when considering only NCA-ES linkages with soil carbon concentration, evidence for positive NCA-ES linkages with food production and positive and negative NCA-ES linkages with atmospheric regulation are observed. See radar plot in Figure 17 for the conflicting evidence for the direction of the NCA-ES linkage between soil carbon concentration and atmospheric regulation. Figure 19 also shows that soil carbon concentration has evidence for NCA-NCA interlinkages with soil moisture content and woodland and forest habitat area, thus managing for soil carbon concentration could involve increasing soil moisture and expanding woodland and forest habitat area. Figure 19 shows that increasing soil moisture could lead to synergies with food production, however expanding woodland and forest habitat area may lead to trade-offs with water supply, yet synergies with atmospheric regulation. Using this approach provides additional evidence that the upland farmer may wish to consider to direct further investigations in a cost-efficient manner.

4.5.1 Context dependency of NCA-NCA interlinkages

We found context dependency in NCA-NCA interlinkages with soil carbon concentration in uplands and lowlands but not for SSSI/NNR locations (Table 6). This is surprising as it suggests that managing ecosystems with high soil carbon concentration for conservation has little effect, as the same NCA-NCA interlinkages are observed in the wider countryside. A possible explanation for this may be due the inclusion of SSSIs and NNRs across uplands and lowlands, resulting in the differences in NCA-NCA interlinkages balancing out.

Figure 5 shows that the NCA-NCA interlinkages with woodland and forest habitat area are context dependent as they only occur in uplands context (Table 6), and thus the trade-off with water supply shown in Figure 5 may only exist in upland areas. This shows the importance of investigating context in the NCA-NCA interlinkages as this trade-off may be of concern to the upland farmer. The NCA-NCA interlinkage between soil moisture content and soil carbon concentration shown in Figure 5 was found to be present under all location contexts (Table 6) and the two NCAs were also recorded at the same spatial scale (Table 4). Decision-makers may wish to start by investigating the NCA-NCA interlinkages that are present under a range of contexts and recorded at the same spatial scale as they are most likely to be causal. For example the NCA-NCA interlinkage between soil moisture content and soil carbon concentration shown in Figure 19 is well established (Falloon et al., 2011; Rawls et al., 2003; Wang et al., 2016). Soil organic matter content was also found to be clustered with soil moisture content and soil carbon content under all contexts, and slope and successional stage were also clustered together. Osman and Barakbah (2011) identified a strong positive linkage between natural succession and the stability of slopes. Birds, grassland habitat area, herbaceous plants and grasses and nitrogen fixers clustered across all location scales, as did landscape diversity, slope, primary productivity, and successional stage. However, some of these NCA were measured at different spatial scales and therefore are less likely to be causally linked. This results in evidence for 16 NCA-NCA interlinkages present across all location contexts. Interestingly, the NCAs pollinators and species richness clustered together (also found by Ebeling et al. (2008)), as did the NCAs primary productivity and soil bulk density in all contexts considered, yet did not cluster together when context was ignored. See Table C7 for the clustering of all the NCAs in Table 4 across all location contexts.

4.5.2 Limitations and further work

The most prominent limitation with our approach to combining evidence for the NCA-NCA interlinkages and NCA-ES linkages is the assumption that NCAs clustered together from the DCA and k-means have synergistic NCA-NCA interlinkages. Furthermore, some of the observed unexpected clustering of NCA such as the lack of NCA-NCA interlinkages identified for water quality may be due to the limitations of multivariate techniques such as DCA when dealing with missing data. For example, it was necessary to assign a value of zero to water quality for CS2007 squares that did not feature a water body to enable the inclusion of the attribute within the DCA. These methodological limitations could explain the lack of support in the literature for some of the NCA-NCA interlinkages identified in this study. However, this could also be due to the need for further studies to investigate the plausibility of the NCA-NCA interlinkage. Our approach for the investigation of NCA-NCA interlinkages deliberately prioritised the inclusion of as many NCAs and ESs as possible, as we believe that the potential over identification of meaningful NCA-NCA interlinkages by assuming a NCA-NCA interlinkage from positive correlation is less of a limitation than potentially missing important NCA-NCA interlinkages and NCA-ES linkages.

The varying spatial scales of the biophysical measurements used to represent the NCA may also impact the clustering. In this study we aggregated the 13 NCA recorded at $200m^2$ spatial scale (Table 4) by averaging over the $1km^2$ square similar to Eigenbrod et al. (2010). Data for different NCAs is collected at a variety of scales and so aggregation was required to ensure that NCA-

NCA interlinkages could be compared at the same spatial scale (Maskell et al., 2013), for example, some NCAs such as landscape diversity and habitat area cannot be observed at smaller spatial scales by definition (Manning et al., 2018). Other NCAs were limited by the sampling in the CS2007 dataset, e.g., water quality. Aggregation is commonly use in integrated models such as InVEST (Nelson et al., 2009). However, it is important to disaggregate data behind the NCAs to determine the stability of NCA-NCA interlinkages at different scales (Maskell et al., 2013) as spatial interactions between organisms and landscape features, or spill over effects, strongly influence some ecosystem functions (Blitzer et al., 2012; Brudvig et al., 2009; Manning et al., 2019, 2018). These spatial interactions can be lost during aggregation. As a result, more caution should be taken when investigating NCA-NCA interlinkages between NCAs recorded at different spatial scales. This study has revealed data scarcity for some of the NCAs at smaller spatial scales and we hope that the transparency of our analysis will help with the investigation of the NCAs recorded at different spatial scales.

This study only shows presence of NCA-NCA interlinkages and NCA-ES linkages and does not review their strength of evidence. Due to the inclusion of varying types of data, NCAs and ESs, quantifying strength of evidence was not appropriate for this study as it has high potential to mislead the decision-maker. It is also important to remember that this approach will not identify all evidence for NCA-NCA interlinkages and NCA-ES linkages. Many of the NCAs in the LiNCAGES platform could not be effectively represented with CS2007 biophysical measurements, therefore some NCA-NCA interlinkages will be missing from this approach. Further work is required to obtain monitoring data for these NCAs as inclusion of additional NCAs will potentially reveal further NCA-NCA interlinkages and thus further NCA-ES linkages. Moreover, the LiNCAGES platform does not represent all NCAs and ESs, for example coastal and marine ESs are severely lacking (Pérez Soba et al., 2017). For these reasons we would advise a decision maker to always keep Figure 15a in mind when using this approach for identifying NCA-NCA interlinkages and NCA-ES linkages.

Although the aforementioned methodological assumptions are a large limitation of the study, the aim of this approach is to provide an evidence-based starting point for decision-makers on which NCA-NCA interlinkages and NCA-ES linkages they may wish to investigate further in relation to their own context. From further investigation of the NCA-NCA interlinkages or NCA-ES linkages the decision-maker can then identify if there is evidence for a causal link under their context. The LiNCAGES platform (Linney et al., 2020) can help with the investigation of NCA-ES linkages.

4.6 Conclusion

This study presents a novel attempt to more comprehensively take account of evidence from different sources on NCA-NCA interlinkages and NCA-ES linkages in decisions related to sustainable ES delivery. We use a combination of both monitoring and literature-based data to show that accounting for NCA-NCA interlinkages reveals evidence for new potential indirect positive and negative NCA-ES linkages, identifying new potential indirect trade-offs and synergies between ESs that decision-makers may not be aware of. We show how context dependencies identified in NCA-NCA interlinkages may propagate down the chain to influence which NCA-ES linkages are revealed to a decision-maker depending on their contextual need. It represents an important starting point for decision-makers and academics, highlighting where further literature research or data collection may be needed to confirm the identified NCA-ES linkages and NCA-NCA interlinkages for their own context. The approach further raises awareness and knowledge of the complexity of natural capital relationships that underlie ES supply.

5 Vive la différence: evidence matters to ecosystem service mapping

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This chapter is a replication of a paper that is under review with the journal *People and Nature.*

5.1 Abstract

1. Mapping the provision of ecosystem services is important to inform management and policy decisions to help avoid further loss of our vital ecosystem services. A wide range of evidence types can be used to map ecosystem service provision. However, we do not fully understand the impact of using different evidence types upon the ecosystem service maps produced in different contexts, or the implications for management decisions when these maps are used to inform short-term or longer-term decisions.

2. We created ecosystem service provision maps for Europe using evidence from an integrated modelling platform, expert opinion and literature synthesis, for the ecosystem services timber production, carbon sequestration and aesthetic landscapes. These maps were then compared to identify similarities and differences for current conditions.

3. We then created future ecosystem service provision maps for different climate change and socioeconomic scenario combinations using each evidence type and investigated how they varied depending on the future scenario mapped.

4. We found that the variations between ecosystem service maps derived from different evidence types changed according to region, ecosystem service and the future scenario investigated, to the extent that the different evidence types can give a different overall direction of change in ecosystem service provision for the same scenario.

5. This study highlights that the type of evidence underpinning ecosystem service maps may have strong impacts on decision making, emphasizing the importance of understanding the strengths and limitations of the underlying evidence.

5.2 Introduction

Ecosystems provide services that are vital for human wellbeing and good quality of life (Burkhard and Maes, 2017; IPBES et al., 2019; Shen et al., 2021). However the complex processes and functions of ecosystems and their interdependencies lead to challenges for the implementation of the ecosystem services approach into management and policy (Vorstius and Spray, 2015). Ecosystem service provision maps have the potential to help overcome these challenges (Vorstius and Spray, 2015), due to their ability to efficiently communicate complex spatial information in a way that people are familiar with (Burkhard et al., 2012; Burkhard and Maes, 2017; Le Clec'h et al., 2016; Vorstius and Spray, 2015). Therefore, ecosystem service provision maps have significant potential for incorporation into policy frameworks (Czúcz et al., 2020; Dunford et al., 2018; Le Clec'h et al., 2016; Vorstius and Spray, 2015), landscape suitability assessments (Burkhard et al., 2012; Crossman et al., 2012) and strategic resource planning (Crossman et al., 2012).

Ecosystem service provision maps are also created for possible future scenarios to inform longer-term decision-making in terms of the robustness of actions and policies, and the resilience of ecosystem service outcomes, to key uncertainties. Such maps can be created using modelling approaches applied to scenarios (Dunford et al., 2015), such as the Shared Socio-economic Pathways (SSPs) and Representative Concentration Pathways (RCPs). The SSPs describe a set of alternative plausible trajectories of future societal development determined by societal challenges to climate change mitigation and adaptation (O'Neill et al., 2017) and have been downscaled and extended for Europe by Kok et al. (2019). The RCPs are emission pathways characterised by the radiative forcing that are used to drive global climate models (van Vuuren et al., 2011).

A wide range of methods and tools exist for creating ecosystem service provision maps (Harrison et al., 2018; Vorstius and Spray, 2015), each with a variety of different evidence types underlying them (Dunford et al., 2018). Combining evidence types allows for the resulting ecosystem service provision map to share beneficial attributes of each evidence type (Dunford et al., 2018). Yet, this also leads to combined evidence maps sharing the limitations of its comprising evidence types.

Harrison et al. (2018) provide a comprehensive description of the range of ecosystem service assessment methods available, and Vorstius and Spray (2015) present a comparison of three ecosystem service mapping tools. Due to the abundance of choice in methods and tools for creating ecosystem service provision maps and the complexity of some of these approaches, the strengths and limitations of their underlying evidence can be overlooked. For example, Schulp et al. (2014) found only a small fraction of ecosystem system service mapping studies addressed uncertainty in a quantitative way.

This study focuses on the evidence underlying ecosystem service provision maps, which we broadly group into: *modelled* (qualitative or quantitative representations of key components of a system and the relationships between those components (Jackson et al., 2000)), *expert* (linking spatially explicit biophysical landscape units to ecosystem service supply and demand using expert judgements (Burkhard et al., 2012)) and *literature* (linking spatially explicit biophysical landscape units to systematic literature review data describing the relationship between natural capital attributes and ecosystem services (Smith et al., 2017)).

These evidence types have different strengths and limitations for mapping ecosystem service provision, which can become more apparent for certain ecosystem services and contexts. For example, modelled data allows for the investigation of future scenarios (Harrison et al., 2018), but can perform poorly for cultural ecosystem services as their complexity makes them much harder to represent with quantitative information (Shoyama et al., 2017). Expert-based evidence allows the creation of maps that are flexible and easy to communicate and understand (Burkhard et al., 2014; Campagne and Roche, 2018), but can be subjective and biased to the expert's experiences (Burkhard et al., 2012). Literature evidence is more comprehensive in its coverage as knowledge is derived from a wide range of studies using different evidence types and approaches, yet is less flexible, and bias in the literature is still observed (Linney et al., 2020), for example ecosystem service studies in marine environments are underrepresented in existing ecosystem service literature (Pérez Soba et al., 2017). Consequently, the evidence types used to create ecosystem service provision maps have the potential to influence their output and thus the decisions informed by the map.

Using Europe as a case study we investigate the research question: how do ecosystem service maps vary depending on the evidence that underpins them? We focus on three types of evidence (modelling, expert and literature) and three ecosystem services (timber production, carbon sequestration and aesthetic landscapes), chosen to be illustrative of provisioning, regulating and cultural services. The research question is first investigated for maps produced under current conditions, where maps are sense checked against empirical data where available. We then combine modelled and expert evidence, and modelled and literature evidence, to create ecosystem service provision maps for future scenarios. The reasons for variations between maps in relation to the strengths and limitations of each evidence type and the implications for the use of ecosystem services maps in planning and decision-making is discussed.

5.3 Materials and methods

5.3.1 Creating ecosystem service provision maps from the different evidence types

5.3.1.1 Present-day ecosystem service provision maps

We first created present-day ecosystem service provision maps for Europe using information that was appropriate to represent each of the three evidence types. For each evidence type we used the same approach to map the three ecosystem services for consistency and to ensure that the approach was openly available to decision makers. Table 7 shows the approach used to represent each of the evidence types for each of the ecosystem services. Table D1 provides a more detailed version of Table 7, including the definitions of the ecosystem services used by each approach.

Table 7: The characteristics of the information sources used to represent each evidence type for each ecosystem service.

Evidence type	Ecosystem service	Approach	Spatial resolution	Indicator name	Units	Reference
Modelled	Timber production	The IMPRESSIONS Integrated Assessment Platform 2 (IAP2) is a suite of ten sectoral meta models which outputs eight land uses and various metrics that can represent ecosystem services.	10` × 10`	Timber production	Mt	(Harrison et al., 2013)
	Carbon sequestration			Potential carbon stock (stored in forests)	t/ha	
	Aesthetic landscapes			Land use diversity	Score 0 - 1	
Expert	Timber production	Matrix linking land cover to 31 ecosystem services using expert derived scores from Burkhard et al. (2014) adapted for Europe by Stoll et al. (2015).	Spatially independent*	Timber	Score 0 – 5 (where 0 = no relevant potential and 5 is maximum relevant potential)	(Burkhard et al., 2014)
	Carbon sequestration			Global climate regulation		
	Aesthetic landscapes			Recreation and aesthetic values		
Literature	Timber production	LiNCAGES platform. An interactive platform linking 42 natural capital attributes to 13 ecosystem services using literature synthesis using a vote counting approach.	Spatially independent*	Timber production	Number of studies evidencing a positive linkage with the ecosystem service.	(Linney et al., 2020)
	Carbon sequestration			Atmospheric regulation		
	Aesthetic landscapes			Aesthetic landscapes		

*Spatially independent; requires a land cover or land use map as input data to create an ecosystem service provision map. The resulting ecosystem service provision map will be same spatial resolution as the input data.

Ecosystem service provision maps based on modelled evidence were created using the IMPRESSIONS Integrated Assessment Platform 2 (IAP2), which combines ten sectoral models and simulates cross-sectoral interactions between agriculture, forestry, biodiversity, water resources and flooding under varying climate and socio-economic conditions (Harrison et al., 2019). The IAP2 splits Europe into 10` x 10` cells and simulates values for ecosystem service indicators for each cell.

The approaches based on expert and literature evidence are spatially independent and thus require joining to a land cover map to produce ecosystem service provision maps. We used the CORINE 2018 land cover map to represent current land cover.

Expert evidence was represented by a land cover to ecosystem service matrix developed by Burkhard et al. (2014) and adapted by Stoll et al. (2015), which links the CORINE land classes to 31 ecosystem services using expert derived scores (Table D2). Stoll et al. (2015) adjusted these scores for Europe through consultation with the site expert teams that worked on the European Long-Term Ecological Research network (LTER-Europe) sites. A total of 28 LTER sites from 11 countries participated in their study (Stoll et al., 2015).

Literature evidence was represented by the LiNCAGES platform (Linney et al., 2020), which links 42 natural capital attributes to 13 ecosystem services using evidence from a systematic review of 780 relevant journal articles. To create a land cover to ecosystem service matrix using the LiNCAGES platform we associated the ecosystem type (classified under Action 5 of the EU biodiversity strategy to 2020 (Maes et al., 2016)) within which each study was located with the relevant CORINE land cover class (see Table D2). The number of positive linkages between natural capital attributes and the three ecosystem services of interest that were recorded within each ecosystem was extracted from the database. Studies that span multiple ecosystems were removed from this analysis to avoid natural capital to ecosystem service linkages not present in a particular ecosystem contributing to the overall score for that ecosystem. 541 of the 780 studies met this criterion. Furthermore, we filtered the literature evidence so that only studies that were located in the continents of Europe and North America were included, leaving 289 studies. Studies located in North America (148) were included to supplement the limited number of studies completed in Europe (141). This was deemed appropriate due to similarities in climatic and socioeconomic conditions of the countries located in these continents. We condensed the two ecosystems "heathland and shrub" and "sparsely vegetated land" into the one ecosystem class of "upland" through summation of the linkages. This was due to the scarcity of literature data for these two ecosystems as many studies featuring these ecosystems span multiple ecosystems and therefore were filtered out of this analysis. This resulted in the final LiNCAGES matrix matched to the CORINE land cover classes (Table D2).

To aid comparison of the maps produced by different evidence sources, the expert and literature evidence ecosystem service maps were scaled from their original 100 x 100m resolution up to the 10` x 10` resolution of the modelled ecosystem service provision maps. This was completed using the Zonal Histogram function in QGIS (QGIS.org, 2022), giving the percentage cover of each

CORINE land class within the 10` x 10` cells. The CORINE land cover percentages were multiplied by their associated ecosystem service provision score for the ecosystem service of interest in the expert matrix (Table D2) (to create the expert ecosystem service provision map) and by their associated number of positive linkage studies in the literature matrix (Table D2) for the ecosystem service of interest (to create the literature ecosystem service provision map). The process for creating modelled, expert and literature evidence type ecosystem service provision maps is represented visually by the workflow in Figure 20.

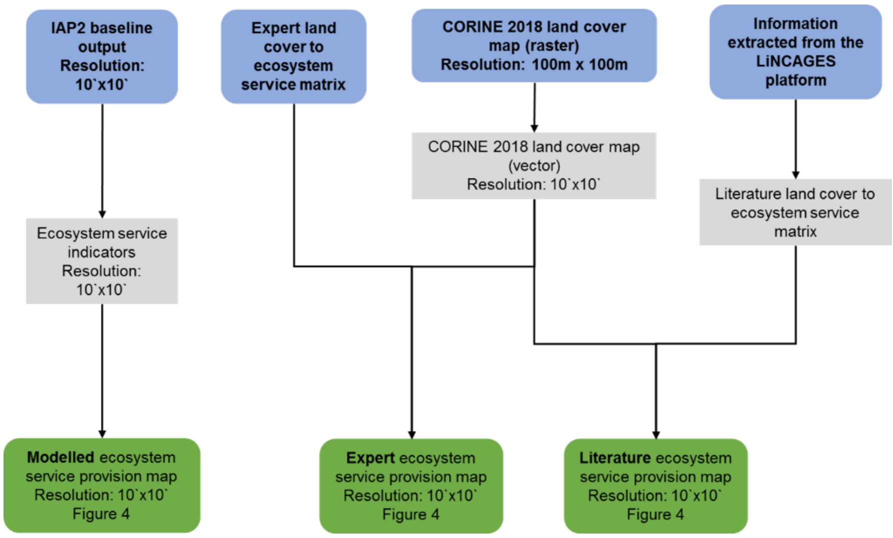

Figure 20: Workflow for creating the ecosystem service provision maps using modelled, expert and literature evidence types. Blue boxes are data sources, grey boxes are intermediate processes and green boxes are the map outputs.

5.3.1.2 Ecosystem service provision maps for future scenarios

The IAP2 includes a range of climate change and socioeconomic scenarios based on the SSPs and RCPs. In this study we investigate future scenarios for three combinations of SSPs and RCPs: (i) SSP1 (Sustainability) x RCP2.6 (EC-EARTH_RCA4 model), (ii) SSP3 (Regional Rivalry) x RCP8.5 (HadGEM2-ES_RCA4 model) and (iii) SSP4 (Inequality) x RCP4.5 (HadGEM2-ES_RCA4 model). The three scenario combinations were chosen to illustrate a varied range of plausible futures likely to lead to different management decisions, rather than fully representing the range of scenario uncertainty. See Box D1 for full details on the parameters for each scenario. For each scenario the IAP2 simulates ecosystem service indicators and the percentage of eight land uses allocations for each 10` x 10` cell (Harrison et al., 2015). The ecosystem service indicators were used to represent the modelled ecosystem service provision maps as shown by Figure 21.

To create similar maps based on expert and literature evidence for the future scenarios, the eight land use allocations (intensive arable, intensive grassland, extensive grassland, very extensive grassland, unmanaged land, managed forest, unmanaged forest and urban) generated by the IAP2 model were further refined to match with 22 of the CORINE land cover classes. The refining of the land use allocations was informed by the CORINE 2018 land cover map and other outputs from the IAP2 such as irrigation and climate suitability (Figure 21). See Box D2 for a more detailed description of how the land use classes were refined. The IAP2 land use allocations for the arable, intensive grassland and forest land allocations were not changed. The extensive grassland and unmanaged land classes were broken down using the CORINE class that underlies them at baseline providing they were climatically suitable. Table D3 shows the CORINE land cover classes that the IAP2 land use allocations were refined to.

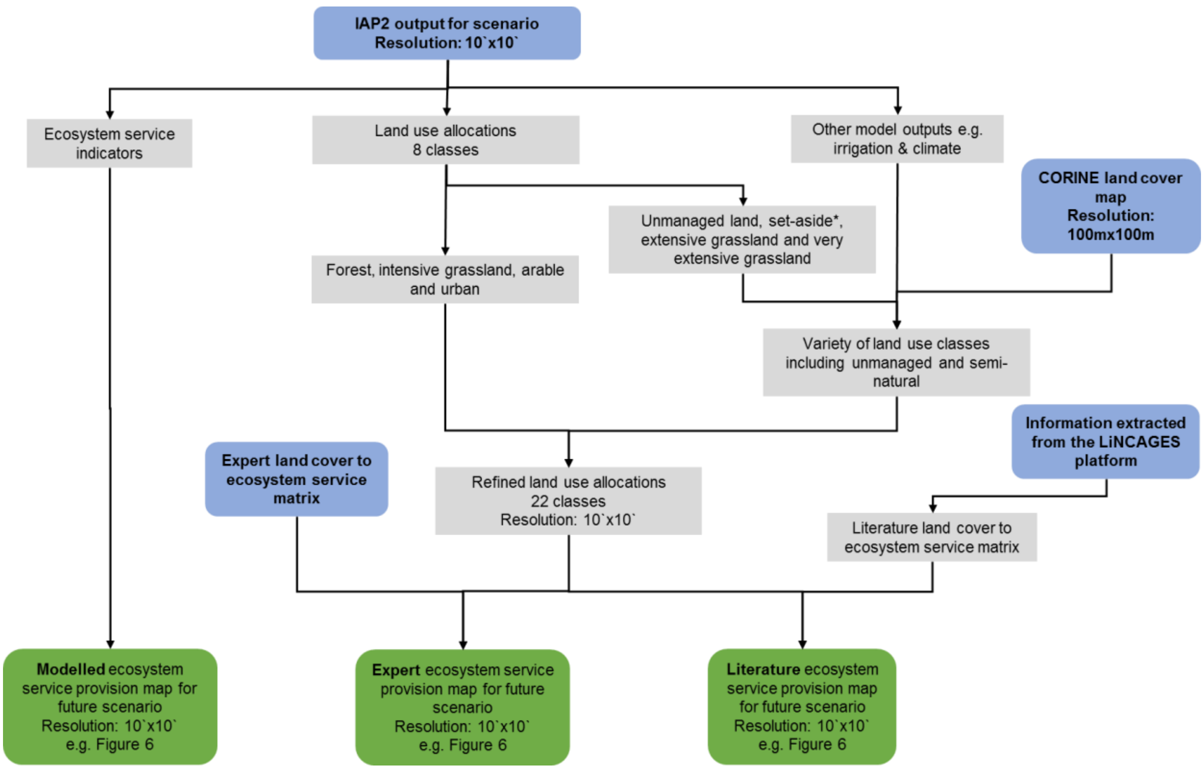

*Figure 21: Method for creating ecosystem service provision maps for future climate and socioeconomic scenarios using modelled, expert and literature evidence. Blue boxes are data sources, grey boxes are intermediate processes and green boxes are maps. *Set aside is calculated as:*

$$\frac{(Arable\ land\ use\ allocation \times Arable\ conservation\ land\ scalar\ slider)}{100}$$

As before we then multiplied these future land cover percentages by their associated ecosystem service provision score for the ecosystem service of interest in the expert matrix (Table D3) (to create the expert ecosystem service provision map) and by their associated number of positive studies in the literature matrix for the ecosystem service of interest (Table D3) (to create the literature ecosystem service provision map).

5.3.2 Comparing ecosystem service provision maps

The ecosystem service provision maps of Europe created using the different evidence types were evaluated visually and by generating a Spearman correlation coefficient to quantify the correspondence between pairs of maps for a particular service derived from two different evidence types (modelled and literature, modelled and expert, expert and literature). Comparisons were completed for the whole map (Europe) and by the regions shown in Figure 22. These regions were developed by Kovats et al. (2014) derived by aggregating the climate zones developed by Metzger et al. (2005) and therefore represent geographical and ecological zones rather than political boundaries. Spearman correlation coefficients were appropriate for comparing the maps due to the varying units of the approaches that represent the evidence types and the skewed nature of the ecosystem service indicators from the IAP2 (Figure D1). This analysis was undertaken in R using the “cor.test” function (R Core Team, 2021).

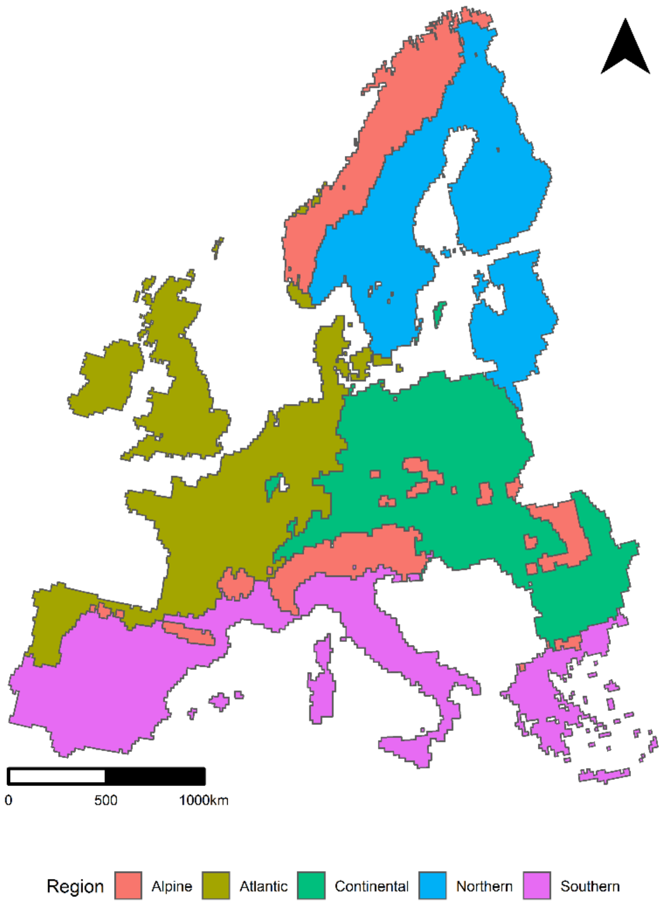

Figure 22: The regions considered in this analysis based on those defined by Metzger et al. (Metzger et al., 2005).

Validation of the present-day ecosystem service provision maps was undertaken where possible through comparison to empirical evidence. This was not intended to be a full quantitative validation due to the limited data available at a relevant spatial resolution, but rather as a sense check of the ranges and broad spatial pattern of the ecosystem service maps. Empirical evidence for timber production (m^3) was represented by empirical data from the European Commission (Eurostat, 2020). Empirical evidence for carbon sequestration (tonnes) was represented by data from the United Nations (FAO, 2020). Due to the subjective nature of the ecosystem service aesthetic landscapes (Dunford et al., 2018; Schulp et al., 2014), we could not find an empirical data source that was representative of the indicator and hence, omitted this from the empirical comparison. Empirical evidence was only available at the national resolution; therefore, we scaled the modelled, expert and literature data to the country level by summing the ecosystem service provision values of the cells within each country. To validate the maps, we calculated Spearman's rank correlation coefficients for the country provision of each evidence type (modelled, expert and literature) compared to the empirical evidence.

5.4 Results

5.4.1 Comparing the present-day ecosystem service provision maps

Comparing the present-day ecosystem service provision maps created using different evidence types for Europe reveals strong disagreement between the modelled maps and the other evidence type maps (Table 8); timber production maps show the most agreement followed by carbon sequestration, and the aesthetic landscape maps show the least agreement. Overall, the expert and literature maps have the greatest agreement for all ecosystem services (Table 8), with timber production showing the most agreement and carbon sequestration and aesthetic landscapes maps showing similar levels of agreement (Table 8).

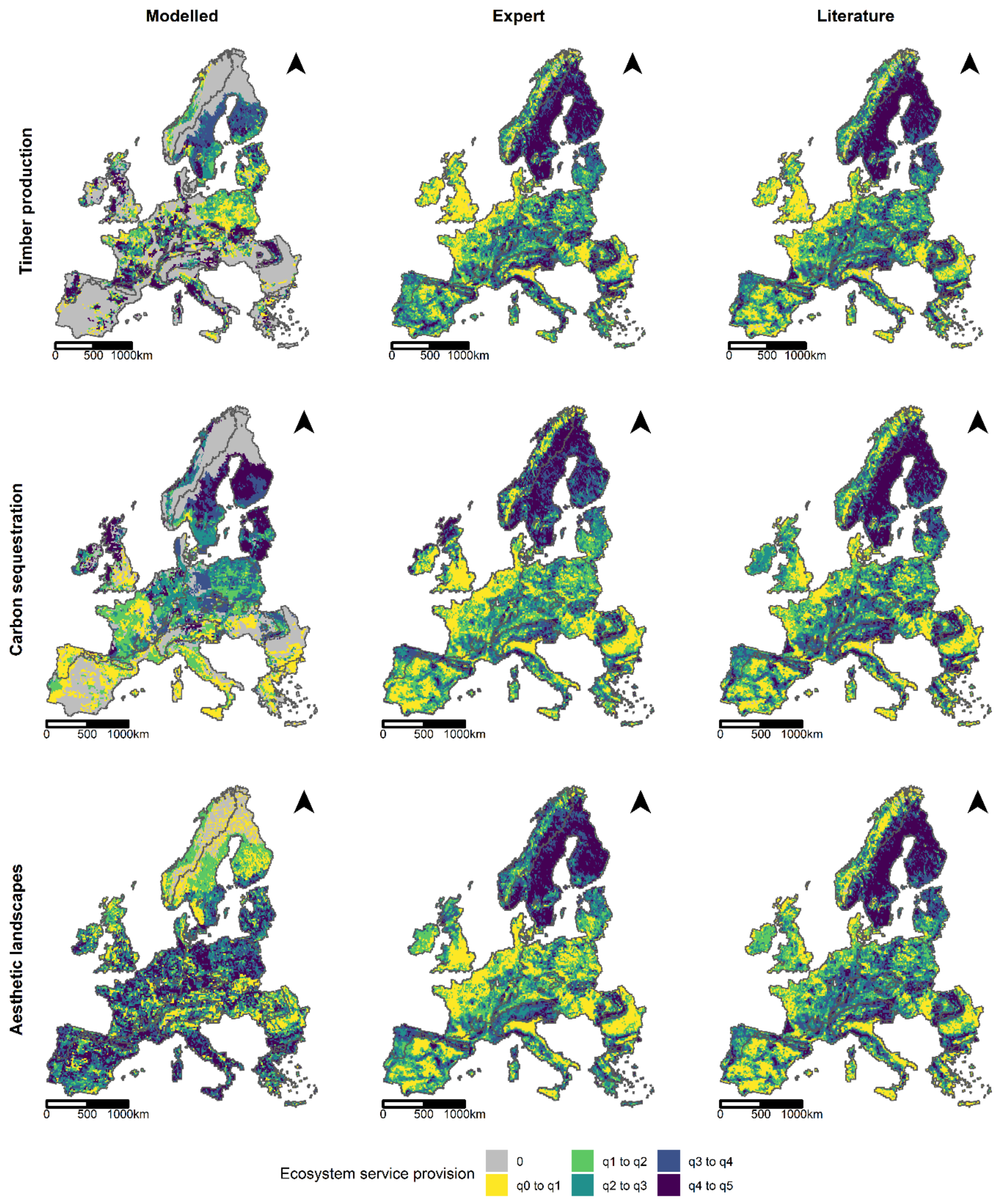

Figure 23: Present-day ecosystem service provision maps for the three ecosystem services created using three evidence types. Ecosystem service provision is coloured by quintiles (q) of the non-zero values of the data due to the diverse and skewed nature of the information sources representing the evidence types. Areas with zero ecosystem service provision are shown in grey. See Figure D1 for locations of the quintile boundaries shown on histograms of each information source behind each map.

Table 8: Spearman correlation coefficient for the comparison of evidence types by region for each ecosystem service. All comparisons were completed at the cell level. A Spearman correlation coefficient of 1 indicates perfect positive correlation and a value of -1 indicates perfect negative correlation. The significance of the correlations are denoted by "" where: *** is p <0.001, ** is p<0.01 and * is p<0.05.*

Evidence compared	Ecosystem service	Europe	Northern	Alpine	Atlantic	Continental	Southern
Modelled and literature	Timber production	**0.28*** **	0.17***	0.3***	0.26***	0.31***	0.22***
	Carbon sequestration	**0.23*** **	-0.08***	0.22***	0.15***	0.21***	0.23***
	Aesthetic landscapes	**-0.07*** **	-0.26***	0.39***	0.16***	0.21***	0.27***
Modelled and expert	Timber production	**0.28*** **	0.29***	0.32***	0.25***	0.32***	0.18***
	Carbon sequestration	**0.17*** **	-0.22***	0.15***	0.25***	0.22***	0.14***
	Aesthetic landscapes	**-0.26*** **	-0.26***	0.17***	0	0.2***	0.2***
Expert and literature	Timber production	**0.96*** **	0.91***	0.99***	0.97***	0.98***	0.8***
	Carbon sequestration	**0.69*** **	0.57***	0.54***	0.41***	0.93***	0.57***
	Aesthetic landscapes	**0.72*** **	0.76***	0.65***	0.43***	0.92***	0.63***

- 0.75 - 1
- 0.5 - 0.75
- 0.25 - 0.5
- 0 - 0.25
- < 0

5.4.1.1 Timber production

The largest disagreement between the different timber production maps is between those based on modelled evidence and those based on other evidence types (Table 8). This discrepancy could be a result of the greater area of zero provision in the modelled timber production maps (Figure 23). Modelled timber production maps use a simulated baseline which takes into consideration demand for both food and timber based on population size and preference. As a result, timber production is only simulated within managed forests that contribute to meeting European timber demand. This leads to the under allocation of forest in the simulated baseline. Furthermore, as the modelled baseline is simulated it may show forest where there may or may not be forest in reality. For example, for areas such as Northern Scandinavia the IAP2 does not identify forest (only unmanaged land) (Figure D4), yet the CORINE land cover map shows areas of coniferous and broadleaved forest (Figure D3).

The modelled timber production maps account for the management status of forests and therefore only show timber production where managed forests are present. Management status is not accounted for in the literature and expert timber production maps, hence high timber production is shown for all forest cover regardless of management status. For example, the IAP2 land use maps show Spain to contain a greater area of unmanaged forest than managed forest (Figure D4). As a result, the modelled timber maps show smaller areas of high timber production in Spain than the expert and literature maps (Figure 23).

The strong agreement between the expert and literature timber production maps is due to both evidence types assigning high values for timber production for the same land classes of broadleaved forest, coniferous forest and mixed forest (Table D2). Differences are present for some minor land cover classes, for example, the expert matrix assigns high scores to the land classes "fruit trees and berries" and "olive groves", whereas the literature matrix does not assign timber provision to this land cover. This leads to small differences between the timber production maps. For example, the slightly lower similarity between the literature and expert timber production maps in the Southern region (Table 8) is due to high timber production scores assigned to the olive groves in Southern Spain (Figure D3) observed in the expert map (Figure 23).

5.4.1.2 Carbon sequestration

The largest disagreement between the carbon sequestration maps is between those based on modelled evidence and those based on other evidence types (Table 8), with the disagreement ranging considerably by region. The high disagreement in the Northern region for the modelled versus other evidence type comparisons is likely due to discrepancies in location of forest area between the evidence types. The IAP2 shows high carbon sequestration in the southern area of the Baltic States (Figure D4), yet the literature and expert do not identify such high values for carbon sequestration (Figure D3). This could be due to the additional aspects of complexity of the biophysical system that the modelled maps consider. The metaGOLTILWA+ model used for the carbon sequestration indicators in IAP2 contains a process-based representation of the carbon cycle and how it responds to changes in temperature, precipitation, effective soil volume, CO_2 concentration, forest management and tree species (Holman and Harrison, 2011), whereas the expert and literature maps are entirely land use based.

The carbon sequestration maps also show the largest disagreement between the expert and literature comparisons of all the ecosystem services mapped. Furthermore, this comparison shows the highest variation in agreement between the regions (0.41 in the Atlantic region to 0.93 in the Continental region). The variation is likely due to the higher area of peat bogs in the Atlantic, Northern and Alpine regions (Figure D3). Peat bogs have a very high score for provision in the expert matrix (5/5), yet a lower value in the literature matrix as peat bog land cover was assigned to the upland ecosystem class in the literature matrix which had a smaller proportion of studies showing positive linkages for carbon sequestration (8/79) (Table D2). Furthermore, along the Northwest coast of Norway, the expert carbon sequestration map shows much higher provision (Figure 23). Figure D3 shows that these areas contain mostly sea and ocean. Sea and ocean have a very high score (5/5) for both carbon sequestration and aesthetic landscapes in the expert matrix (Table D2). The ecosystem sea and ocean was not included in the literature matrix as there were only four studies in the LiNCAGES platform undertaken in this ecosystem and therefore zero provision is given for this ecosystem. Further disagreement (in the Southern and Northern regions) arises from the presence of transitional woodland and shrub. This land cover has very high provision for carbon sequestration in the literature matrix (as it is connected to woodland and forest) yet no provision for carbon sequestration in the expert matrix (Table D2).

5.4.1.3 Aesthetic landscapes

The aesthetic landscapes maps have the largest disagreement of all the ecosystem services we investigated for the modelled evidence versus other evidence type comparisons (Table 8). In this case, the disagreement is most likely related to the different definition of the ecosystem service aesthetic landscapes used by the IAP2 model (Table D1). The modelled evidence uses land use diversity as a proxy for aesthetic landscapes, whereas the expert and literature matrices assign values to specific land covers (Table D2). For example, the low levels of land use diversity in managed woodlands result in low values for aesthetic landscapes in the modelled maps, yet high values are shown in these areas in the expert and literature maps as woodland land cover is assigned high values for aesthetic landscape provision in the expert and literature matrices (Table D2). This disagreement is exemplified by the low Spearman rank correlation coefficient for the modelled versus other evidence type comparisons in the Northern Region (Figure 23) which contains large areas of forest (Figure D3).

Comparing the expert and literature maps for aesthetic landscapes reveals good agreement overall (Table 8), with the least agreement in the Atlantic, Alpine and Southern regions which contain large areas of grassland and upland land covers. These land covers have a lower provision of aesthetic landscapes in the literature matrix than the expert matrix (Table D2).

5.4.1.4 Sense checking the maps against empirical data

All evidence types show good agreement with the empirical evidence; the lowest Spearman correlation coefficient is 0.69 for the comparison between the modelled and empirical evidence for carbon sequestration (Table 9). However, the coarser resolution (country level) of this comparison is likely responsible for the high agreement due to the underestimation of the cell level discrepancies identified in Figure 23. To further support this theory, when we make comparisons between each evidence type at the country level there is much greater agreement, with Spearman correlation coefficients ranging from 0.79*** to 0.99*** for timber production and 0.79*** to 0.97*** for carbon sequestration (Table D7).

Table 9: Spearman correlation coefficient for the evidence comparisons of the ranked total ecosystem service provision for Europe. All comparisons were completed at the country level. A Spearman correlation coefficient of 1 indicates perfect positive correlation and a value of -1 indicates perfect negative correlation. Aesthetic landscapes is not included in this comparison due to lack of empirical data.

	Evidence comparison		
Ecosystem service	Modelled and empirical	Expert and empirical	Literature and empirical
Timber production	0.84***	0.89***	0.9***
Carbon sequestration	0.69***	0.78***	0.83***

0.75 - 1
0.5 - 0.75
0.25 - 0.5
0 - 0.25
< 0

Out of the 26 countries for which empirical data was available 9/26 show good agreement between the empirical data and all evidence types for carbon sequestration and 14/26 countries show good agreement between empirical data and all evidence types for timber production (Figure D14). All evidence types overestimate carbon sequestration in the northern region countries when compared to empirical data and for Greece both carbon sequestration and timber production are significantly overestimated. Bulgaria has good agreement between the expert and literature evidence and the empirical data, yet the modelled evidence significantly underestimates both timber production and carbon sequestration.

5.4.2 Future ecosystem service provision maps

To explore the extent that different evidence types can affect the creation of future ecosystem service provision maps we first investigated whether the maps agreed on the direction of change in provision for Europe for each possible future scenario (Figure 24).

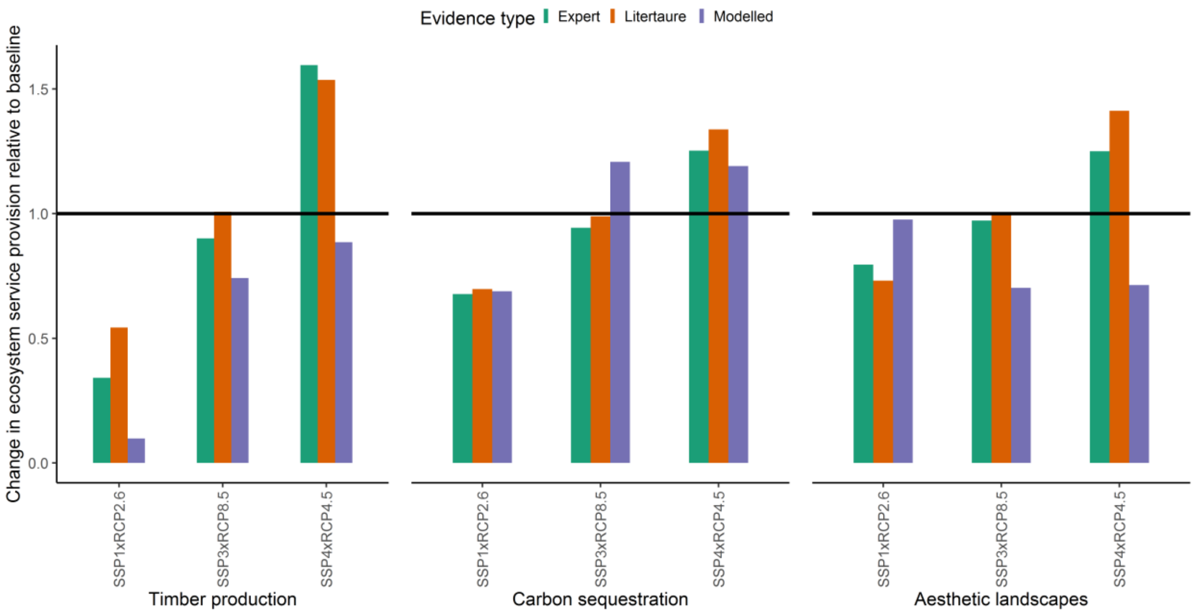

Figure 24: Direction of change in ecosystem service provision relative to baseline for Europe (calculated as: $\frac{scenario\ value}{baseline\ value}$) for each future scenario and evidence type. The y-intercept shows a relative change of 1 (no change). Bars ending below the y-intercept show a decrease in ecosystem service provision; bars ending above the y-intercept show an increase in ecosystem service provision.

Figure 24 shows that the overall direction of change in ecosystem service provision for Europe can vary for the same scenario depending on the evidence type underlying the map. For the SSP4xRCP4.5 scenario both the timber production and aesthetic landscapes maps show an increase in overall ecosystem service provision for Europe based on expert and literature evidence, yet a decrease in ecosystem service provision based on modelled evidence. The carbon sequestration maps for the SSP3xRCP8.5 scenario display the opposite, with the modelled evidence showing an increase in provision and the expert and literature evidence showing a decrease in overall provision for Europe. All the ecosystem service maps for the SSP1xRCP2.6 scenario agree on the overall direction of ecosystem service provision, yet large disagreements in the magnitude of change in the provision of timber production are observed. Figure 24 shows that all three of the ecosystem services have at least one disagreement in direction in change in ecosystem service provision for a possible future scenario.

Comparisons at the cell level help to reveal why the use of different evidence types in the creation of future ecosystem service provision maps results in disagreements in the direction and magnitude of change in ecosystem service provision for Europe. In general, the maps for timber production and carbon sequestration show a greater level of agreement between the modelled and other evidence type comparisons under the future scenarios (Tables D5 and D6) than for the present-day comparison (Table 8). This is expected as the expert and literature future ecosystem service provision maps use the modelled land use output. However, the maps for aesthetic landscapes only show very slightly more agreement across the evidence types for the future scenarios compared to present-day (Table 10).

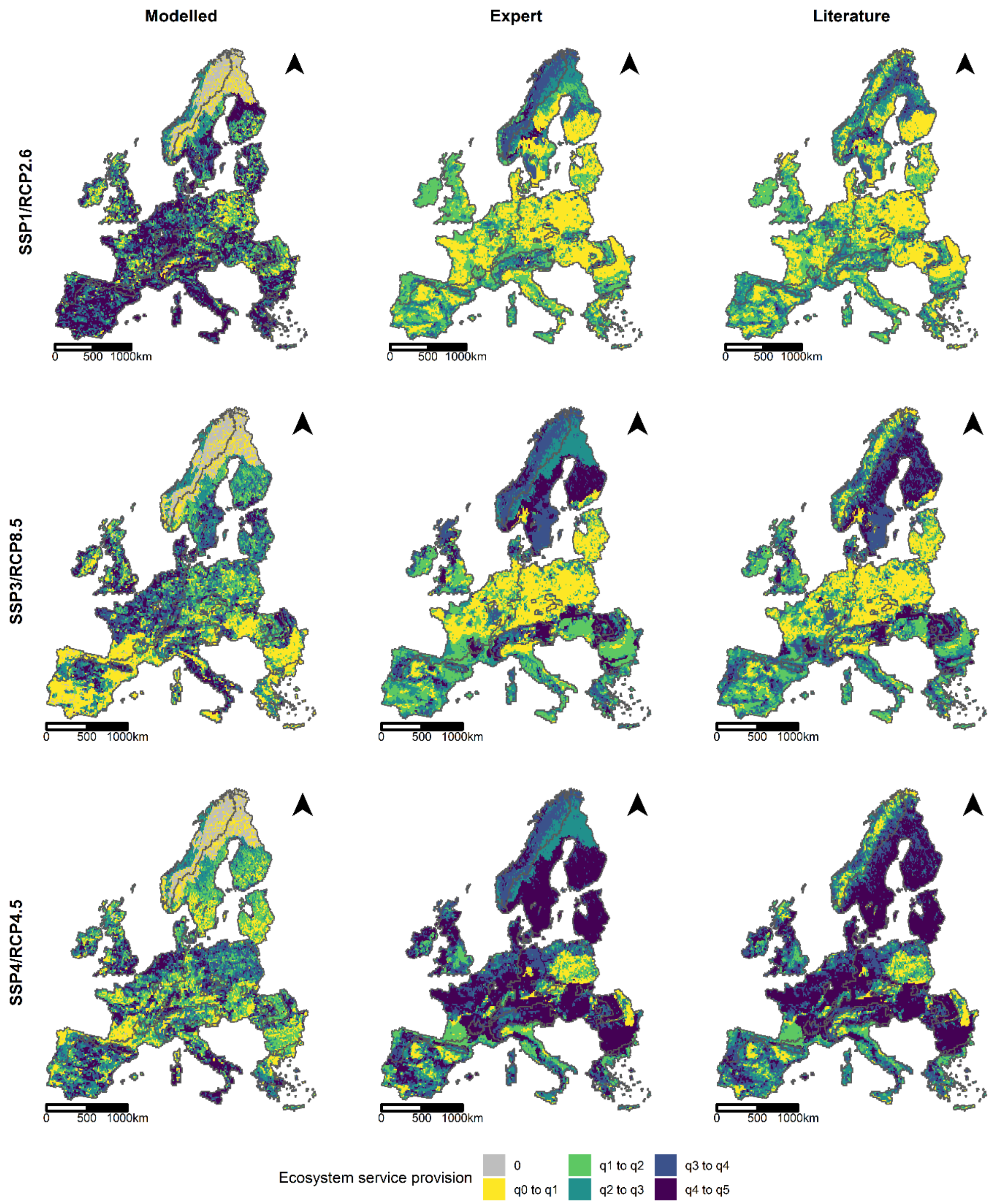

Figure 25: Aesthetic landscapes future ecosystem service provision maps for three possible future scenarios created using the three evidence types. Ecosystem service provision is coloured by quintiles (q) of the non-zero values of the data due to the diverse and skewed nature of the information sources representing the evidence types. Areas with zero ecosystem service provision are shown in grey. See Figure D2 for locations of the quintile boundaries shown on histograms of each information source behind each map.

5.4.2.1.1 SSP1xRCP2.6

The SSP1xRCP2.6 scenario is characterised by an increase in population, a decline in food imports, greater agricultural mechanisation, reduced dietary preferences for meat, and a focus on environmental sustainability reflected by a more extensive approach to agriculture with lower crop yields (Harrison et al., 2019). The need to meet increasing food demands due to the increasing population and reducing imports leads to an agricultural expansion at the expense of forest (Harrison et al., 2019). The larger area of arable land and grassland in the SSP1xRCP2.6 scenario leads to disagreement between the carbon sequestration maps as grassland has a high value (31/79) in the literature matrix yet a low score (1/5) in the expert matrix. However, for the aesthetic landscapes maps, SSP1xRCP2.6 shows the greatest agreement between all of the evidence types, particularly in the Continental region (Table 10). This is because arable land has low values for the provision of aesthetic landscapes in the expert and literature matrices (Table D3) and is located in areas of low land use diversity (Figure D5), therefore areas of arable land will also show low areas of provision in the modelled maps (Figure 25).

Table 10: Spearman correlation coefficient for the comparison of evidence types by region for each scenario for aesthetic landscapes. All comparisons were completed at the cell level. A Spearman correlation coefficient of 1 indicates perfect positive correlation and a value of -1 indicates perfect negative correlation. The significance of the correlations are denoted by "" where: *** is p <0.001, ** is p<0.01 and * is p<0.05.*

Evidence compared	Scenario	Europe	Northern	Alpine	Atlantic	Continental	Southern
Modelled and literature	SSP1xRCP2.6	**0.22*** **	-0.02	0.27***	0.34***	0.79***	0.42***
	SSP3xRCP8.5	**-0.04*** **	-0.31***	0.35***	0	0.26***	-0.02
	SSP4xRCP4.5	**-0.23*** **	-0.03*	0.4***	-0.33***	-0.56***	-0.07***
Modelled and expert	SSP1xRCP2.6	**0.06*** **	0.02	-0.23***	0.29***	0.81***	0.37***
	SSP3xRCP8.5	**-0.07*** **	0.05***	0.24***	0.01	0.27***	0
	SSP4xRCP4.5	**-0.2*** **	0.26***	0.46***	-0.41***	-0.58***	-0.09***
Expert and literature	SSP1xRCP2.6	**0.79*** **	0.86***	0.1***	0.78***	0.95***	0.81***
	SSP3xRCP8.5	**0.81*** **	0.6***	0.3***	0.89***	0.97***	0.72***
	SSP4xRCP4.5	**0.84*** **	0.57***	0.46***	0.93***	0.98***	0.91***

0.75 - 1
0.5 - 0.75
0.25 - 0.5
0 - 0.25
< 0

5.4.2.1.2 SSP3xRCP8.5

The SSP3xRCP8.5 scenario is characterised by inequality and regional conflict with many countries struggling to maintain living standards. This leads to a large decrease in population, lower food imports and lower arable crop yields (Harrison et al., 2019). To compensate for low food imports and yields, arable land increases at the expense of woodland, yet arable land does not increase to the same extent as it does for SSP1xRCP2.6 due to the lower population and thus food demand. Table 10 shows relatively large disagreement between the expert and literature maps for the SSP3xRCP8.5 in the Southern region. This is due to the large areas of agriculture and natural vegetation (Figure D11), which have a moderately high score for timber production in the expert matrix (2/5) but no value in the literature matrix (Table D3). The dissimilarities in the Northern region are due to the presence of transition woodland and shrub (Figure D10), which has no score for provision of timber production in the expert matrix yet is connected to woodland and forest in the literature matrix and hence assigned a high score (Table D3). Transitional woodland also has very high provision for carbon sequestration in the literature matrix as it is connected to woodland and forest but has no provision for carbon sequestration in the expert matrix (Table D3). This again leads to disagreement between the maps for SSP3xRCP8.5, with the expert and modelled carbon sequestration maps showing very low or no provision of carbon sequestration in Northern Scandinavia whereas the literature maps show high provision (Figure 25).

5.4.2.1.3 SSP4xRCP4.5

The SSP4xRCP4.5 scenario is characterised by a low focus on environmental protection, a willingness to increase dependence on food imports and a decreasing population. This removes considerable pressure on the food demand that needs to be met from Europe's own agricultural system. Furthermore, high-tech and intensive land management means that greater yields are possible from the same unit of land resulting in significant decline of agricultural area. Consequently, forest area expands considerably (Harrison et al., 2019). The increase in forest area in this scenario exaggerates the disagreement between the modelled evidence maps and the other evidence types. As shown by the present-day timber maps (Figure 23), the literature and expert maps assign timber production to all forests regardless of management status, whereas the modelled evidence only assigns timber to managed forests when it is needed to meet demand. The increased area of unmanaged forest present in Europe under the SSP4xRCP4.5 scenario (Figure D7) exaggerates this discrepancy between the evidence types resulting in very good agreement between the literature and expert maps and relatively poor agreement for the modelled evidence comparisons (Table D5). For the carbon sequestration maps the presence of more established forest land cover in SSP4xRCP4.5 and less transitional woodland, likely contributes to SSP4xRCP4.5 having the greatest agreement across evidence types. However, the increased forest area of SSP4xRCP4.5 exaggerates the difference in definitions of aesthetic landscapes between the modelled and other evidence types identified in the present-day maps (Figure 25). The land use in SSP4xRCP4.5 is less diverse due to the increased forest cover and as such land use diversity is lower. Continental and Atlantic regions (Table 10) feature the lowest agreement as they contain large areas of forest under SSP4xRCP4.5 (Figure D7).

5.5 Discussion

5.5.1 Do ecosystem service provision maps vary depending on their underlying evidence?

This study finds that ecosystem service provision maps do vary depending on the evidence that underpins them. When comparing maps for present-day ecosystem service provision, we found those based on modelled evidence show the most differences when compared to the other evidence types. The maps for the provisioning service of timber production showed the most agreement between the evidence types and the maps for the cultural service of aesthetic landscapes showed the least agreement between the evidence types. Furthermore, we found the level of agreement could vary considerably with the region of Europe mapped.

When combining evidence types to investigate ecosystem service provision for possible future scenarios we found disagreement in the evidence type maps for the same future scenario to the extent that they showed opposing overall direction in change of ecosystem service provision for Europe. When investigating the future ecosystem service provision maps in detail we found that the underlying reasons for the differences between evidence types identified in the present-day maps (e.g. level of complexity, sensitivity of certain land use classes, variation in definitions or proxy indicators), could be exaggerated by some possible future scenarios and minimised by others.

5.5.2 Importance to decision makers

Variation in ecosystem service provision maps depending on their underlying evidence type is a major concern if such maps are to be used in informing important environmental governance decisions (Le Clec'h et al., 2016; Schulp et al., 2014). Significant challenges in sense checking the ecosystem service provision maps (Kopperoinen et al., 2014; Schulp et al., 2014) only exacerbates this problem. Many ecosystem services cannot be measured directly (Schulp et al., 2014), which is particularly apparent for some regulating and cultural services (Kopperoinen et al., 2014). For the services that can be measured directly empirical data is scarce and of low resolution (Dick et al., 2014). In this study we were only able sense check the maps using data at the national scale. Like Schulp et al. (2014) we found comparisons at such coarse resolution to underestimate the spatial variation in ecosystem service provision, resulting in higher agreement between the maps. This reveals the importance of spatial scale when comparing ecosystem service provision maps (Schulp et al., 2014). Obtaining detailed quantitative assessments of all the individual ecosystem services is a time-consuming and expensive task and therefore usually unfeasible in a tightly scheduled land use planning process (Kopperoinen et al., 2014). As a result the use of other evidence types is required (Kopperoinen et al., 2014), therefore decision makers must be aware of the different strengths and weaknesses of the evidence types used in ecosystem service provision maps (Le Clec'h et al., 2016).

5.5.3 Strengths and limitations of different evidence types

5.5.3.1 Modelled evidence

Modelled evidence accounts for more of the complexity of the biophysical system (Dunford et al., 2018). Integrated assessment models, such as the IAP2, go a step further and include interdependencies between biophysical and human systems, accounting for cross-sectoral interactions (Harrison et al., 2018). In this study we found that the greater complexity represented in the modelled maps resulted in large disagreements with the literature and expert maps, which use a simple land cover to ecosystem service matrix-based approach. For example, we found the modelled maps only assigned timber production to managed forests whereas the other evidence types assigned it to all forests, potentially resulting in overestimates of timber production.

However, complex models have their own limitations and assumptions and are often seen as a 'black box' that can be difficult to understand by decision makers (Harrison et al., 2018; Shoyama et al., 2017; Vorstius and Spray, 2015). Pragmatically, modelled evidence can be high cost and thus inaccessible to some decision makers due the requirement of data inputs, expert knowledge to run and build models along with the long run time and computing power requirement of the models themselves (Harrison et al., 2018, 2013; Martínez-López et al., 2019).

Ecosystem service maps created using modelled data use proxies to represent ecosystem service provision (due to lack of monitoring data). However, whilst it is recognised that maps based on such proxies are crude estimates, there is little discussion on the magnitude of errors associated with them (Schulp et al., 2014). For example, in this study the proxy used to represent aesthetic landscapes was land use diversity. This proxy differed considerably to how the expert and literature evidence defined aesthetic landscapes, resulting in the aesthetic landscapes maps showing the most disagreement of all the ecosystem service maps compared.

The IAP2 uses a simulated baseline, which will never be a perfect substitute for observed data. The disagreements between the land cover distributions derived from CORINE 2018 monitoring data and the simulated land use data from the IAP2 had a significant influence on the present-day ecosystem service map comparisons. However, many models can run from an observed baseline to overcome such limitations. Furthermore, simulated data is useful for where data is sparse (Willcock et al., 2020) or non-existent, for example, when predicting future ecosystem service provision. This ability to explore possible future ecosystem service provision is a major advantage of modelled evidence (Harrison et al., 2018).

5.5.3.2 Land use matrix-based approach

In contrast to modelled evidence, the expert and literature evidence use a simple matrix based approach which is fast, transparent and easy to use and communicate (Burkhard et al., 2009; Harrison et al., 2018). This makes their strengths and limitations easy to identify and communicate to stakeholders (Harrison et al., 2018). However, this simplicity comes at a cost as this method is entirely land use driven and therefore makes the assumption that the landscape provides all of the services (Burkhard et al., 2009; Le Clec'h et al., 2016; Schulp et al., 2014). Land use alone lacks information regarding important components of ecosystem condition that supports ecosystem service capacities, such as soil type and quality, water availability, geomorphology or overall ecosystem integrity (Campagne et al., 2020). This aggregation of complex information (Burkhard et al., 2012) gives a false impression of completeness (Harrison et al., 2018; Shen et al., 2021). For example, the class of ''water bodies'' provides several ecosystems services but in reality various qualities of water bodies determine the real ecosystem service provision potential (Kopperoinen et al., 2014). More complex inputs can be used to combat this dependency, e.g. Kopperoinen et al. (2014), however this reduces the simplicity, manageability, generality, and comparably of the simple matrix based approach (Campagne et al., 2020). Furthermore, it has been shown that more complex ecosystem service assessment approaches do not necessarily deliver more robust results than those harnessing expert knowledge (Campagne et al., 2020).

5.5.3.3 Expert evidence

Expert derived scores for ecosystem service provision are subjective to the expert who came up with them. However, this does allow for the involvement of stakeholders in the generation of the scores, making the maps more specific to their context (Harrison et al., 2018). For example, in this study we used expert scores that were more relevant to our context by using scores adjusted for Europe through consultation with the site expert teams that worked on the LTER-Europe sites, see Stoll et al. (2015). Due to the human element present in assigning scores for cultural services, it can be argued that such services are better represented by expert evidence, yet the subjectivity of cultural services such as aesthetic landscapes is still present (Dunford et al., 2018).

The inclusion of a large amount of ecosystem services in the expert matrix leads to much quicker and less costly creation of ecosystem service provision maps. For example, expert derived ecosystem service provision maps for up to 31 ecosystem services could have been created for this study.

5.5.3.4 Literature

Literature evidence derived values for ecosystem service provision are less biased than expert derived scores as the evidence comes from a wide range of studies including those based on quantitative data. However, the interpretation of the literature reviewer can affect the literature values (Smith et al., 2017). Furthermore bias is present in the ecosystem service literature (Ricketts et al., 2016; Schwarz et al., 2017). For example, marine ecosystems have been identified as poorly studied in the existing ecosystem service literature (Pérez Soba et al., 2017). In this study we did not assign marine land classes a literature value due to the lack of studies conducted in this ecosystem. This led to disagreements with the expert ecosystem service provision maps.

Literature based ecosystem service provision maps are novel. To the best of the author's knowledge literature synthesis has not been used to create ecosystem service provision maps in this way. However, creating a land use to ecosystem service matrix requires significant aggregation and simplification of literature data through assigning the literature studies to broad ecosystem classes. This pooling of evidence can oversimplify the literature evidence as different studies have varying contexts within the ecosystem they are assigned to (Martnez-Harms and Balvanera, 2012; Ricketts et al., 2016). For example, the upland ecosystem class includes studies in both "heathland and shrub" and "sparsely vegetated land ecosystems. When comparing the literature maps with those of other evidence types, we found some areas of disagreement to be a result of this aggregation and simplification of evidence. For example, as peatland is represented by the broad ecosystem class "upland" the individual value of peatland in the literature carbon sequestration maps is underestimated.

Literature synthesis is costly and time consuming. However, existing literature reviews can be used for generating ecosystem service provision maps. For example, the LiNCAGES platform (Linney et al., 2020) allows for interactive assignment of context dependency to this type of evidence. By using this existing literature data, ecosystem service provision maps can be generated at very little cost. For example, a further 10 ecosystem service maps can be generated using the literature data from the LiNCAGES platform.

5.5.4 Advice when using ecosystem service provision maps

Due to the impact that the strengths and weaknesses of the different evidence types can have on ecosystem service provision maps it is essential that map makers clearly communicate the definitions of the ecosystem services used, the methods, and related uncertainties (Schulp et al., 2014). There is an increasing number of networks, tools and training opportunities to help in selecting and applying new methods (Dunford et al., 2018), but guidance on the choice and consequences of method selection is not widely published in the academic literature (Harrison et al., 2018).

We have shown that the extent of the disagreement between different evidence type maps is context and application dependent. When creating future ecosystem service provision maps we found that disagreements between the evidence types can be exacerbated or diminished depending on the scenario mapped. Therefore, it remains an important task to elaborate which are the most appropriate evidence types for ecosystem service provision mapping for different assessment purposes and contexts (Campagne et al., 2020; Le Clec'h et al., 2016), as depending on the decision maker's context or mapping application, certain strengths and limitations of the evidence types may be more or less apparent in their maps.

Where possible, we recommend that multiple ecosystems service provision maps should be created using different evidence types and compared to identify potential areas of disagreement in ecosystem service provision. This can enhance understanding and build confidence in the results for the stakeholder and allows for targeting areas for further research (Dunford et al., 2018). For example, the high level of agreement between the expert and literature maps gives greater confidence in the more subjective expert derived scores as they are in agreement with literature values based off quantitative studies. However, creating ecosystem service provision maps using different evidence types is time consuming and requires more resources, expertise and interdisciplinary working (Dunford et al., 2018).

5.5.5 Methodological limitations and further work

This study has focussed on the evidence types underlying ecosystem service provision maps. We do not account for methodological differences in creating ecosystem service provision maps which have been shown to cause disagreements between maps (Harrison et al., 2018; Schulp et al., 2014). However, methods get replaced and evolve, yet evidence types stay the same. As a result, we believe that comparing evidence types improves the longevity of our findings.

Due to the differing units of the evidence type maps investigated in this study, quintiles and ranked values were compared. This is less accurate than comparing absolute values (Schulp et al., 2014), however we believe it is the best available way to compare evidence types with diverse units of measure. The scale of the comparison of the maps has also been shown to effect the results (Schulp et al., 2014). We found that comparisons at coarser spatial scales miss smaller level spatial variation. The highest spatial resolution of the maps compared in this study is still fairly low (10` by 10`). Further studies should compare the evidence types at higher spatial resolution to discover if new disagreements are revealed at this scale.

We used land use to ecosystem service matrices that use present-day expert scores and literature evidence to create the expert and literature maps of possible future ecosystem service provision. The assumption that the scores and literature evidence will not change for the future scenarios we considered is a key limitation of these maps. However, the ecosystem service provision maps created in this study are not designed for informing decisions making as in reality evidence types are not used completely independently (Harrison et al., 2018). Instead, the aim of this study was to show how the different characteristic strengths and limitations of the different evidence types that underlie ecosystem service provision maps can significantly impact the final ecosystem service provision map.

5.6 Conclusions

This study has illustrated the dependence of ecosystem service maps on the evidence that underpins them. Differences between present-day maps based on modelled, expert and literature evidence vary according to the region of Europe mapped and the spatial scale of the comparison. Combining evidence types can be useful for exploring ecosystem service provision under a range of plausible futures. However, comparison between maps based on different evidence sources showed that disagreements found in the present-day maps can be exacerbated or diminished depending on the scenario. Such disagreements can result in different mapped directions of change, in addition to varying magnitudes and spatial patterns of change. This is a major concern if ecosystem service provision maps are to be used in informing important environmental governance decisions. Furthermore, the lack of empirical data for many ecosystem services, particularly at large scales, significantly limits the sense checking of ecosystem service provision maps (Layke et al., 2012; Müller and Burkhard, 2012).

Ecosystem service provision maps are powerful tools for supporting the implementation of the ecosystem services approach into management and policy; but with great power comes great responsibility. Map makers need to successfully communicate their ecosystem service definitions, methods, and strengths and limitations of the evidence underlying their maps, and decision makers need to understand how the strengths and limitations of the underlying evidence may become more or less apparent depending on their context or mapping application.

6 Discussion and research outcomes

6.1 Introduction

In this chapter, the research presented in Chapters 3-5 is summarised and discussed in relation to the objectives of the thesis. These were to:

- Holistically investigate the link between natural capital and ecosystem services while accounting for context dependency
- Investigate how accounting for interlinkages across ecosystem service cascades can reveal new indirect natural capital to ecosystem service linkages
- Compare how the evidence type used for natural capital to ecosystem service linkages can influence maps of ecosystem service provision
- Show how different evidence types can be combined to further support environmental decision making based on linkages between natural capital and ecosystem services
- Create a platform to support collation, exploration, and synthesis of evidence on natural capital and ecosystem services and its communication in environmental decision making and research.

Detailed discussion of the results from the work in this thesis is described in each chapter. This discussion chapter presents a synthesis in terms of the overall objectives of the thesis and the wider implications of the findings.

6.2 Synthesis of findings

6.2.1 Holistically investigate the link between natural capital and ecosystem services while accounting for context dependency

Chapter 3 investigates the linkages between natural capital and ecosystem services using a large literature synthesis undertaken by Smith et al. (2017), and discusses the development of the interactive LiNCAGES platform for analysing the context dependency of this evidence base. The inclusion of 42 natural capital attributes and 13 ecosystem services in this review allowed for holistic exploration of the linkages between natural capital and ecosystem services. The analysis undertaken by Smith et al. (2017) focusses on biotic and abiotic attributes, the interactions between ecosystem services and the impact of any human input or management. Although they recognised the importance of context dependency, they did not analyse the effect it had on different natural capital to ecosystem service linkages. This was also the case for other similar literature synthesis (e.g. Balvanera *et al.*, 2006; de Bello *et al.*, 2010; Harrison *et al.*, 2014; Ricketts *et al.*, 2016; Hevia *et al.*, 2017; Schwarz *et al.*, 2017; Smith *et al.*, 2017). Table B1 for a comparison of these systematic reviews including details on the data they recorded on context.

This is due to the literature synthesis studies having their own research aims which effects the type and range of literature synthesised. Although the literature synthesis by Smith et al. (2017) is perhaps the most comprehensive study covering 780 papers, 13 ecosystem services and 42 natural capital attributes, the evidence collected is still specific to the study's own context and research aims. Ideally knowledge from the different reviews needs to be brought together, but this knowledge is difficult to integrate due to the different research aims of the studies, varying database templates, and the data behind many literature syntheses is not open access. The LiNCAGES platform helps to overcome this by allowing researchers to add additional literature. This could include the addition of existing literature syntheses to create a more powerful, and continually growing, database for end-users.

Certain types of evidence may be more useful for specific research or stakeholder questions (Linney et al., 2020). The need to account for this subjectivity in utilising the type of evidence underlying natural capital to ecosystem service linkages (Duncan et al., 2015) led to the creation of the LiNCAGES platform. The LiNCAGES platform enables users who have different context requirements to prioritise evidence from studies that are more relevant. This is achieved through allowing the user to filter and hierarchically weight studies, evidencing natural capital to ecosystem service linkages by context dependent aspects (e.g. spatial scale of study, temporal scale of study, continent that study was located in). Through exploration of a hypothetical use case scenario, Chapter 3 shows that evidence for many natural capital to ecosystem service linkages, and trade-offs and synergies between services, is severely diminished or non-existent under certain contexts, such as larger spatial scales and European study location. For example, when investigating linkages between natural capital attributes and the ecosystem service of timber production, filtering for only studies located in Europe considerably reduces the large amount of evidence for positive linkages between the natural capital attribute "presence of a specific species type" and timber production. This is due to many European studies focussing on natural forests rather than plantations.

In addition, LiNCAGES is not only useful for showing evidence for natural capital to ecosystem service linkages; the linkages that the platform doesn't show provide important information to researchers on key evidence gaps. This supports researchers in working collaboratively to target and collate additional evidence that can strengthen and add value to this evidence base. This will help reduce the reporting bias in the evidence base, where some natural capital and ecosystem services are studied more frequently than others. This is a key limitation of the literature synthesis evidence that underlies the platform (Adhikari and Hartemink, 2016; Balvanera et al., 2006; Hevia et al., 2017).

Although the LiNCAGES platform is holistic by accounting for linkages between 42 natural capital attributes and 13 ecosystem services, this is not an exhaustive list. Further work could involve the addition of further natural capital attributes and ecosystem services, particularly those linked to marine and coastal ecosystems that are underrepresented in the literature synthesis underlying LiNCAGES as well as in the wider literature (Burkhard and Maes, 2017; Pérez Soba et al., 2017).

6.2.2 Investigate how accounting for interlinkages across ecosystem service cascades can reveal new indirect natural capital to ecosystem service linkages

Chapter 4 explores how accounting for relationships between multiple natural capital attributes can reveal new indirect natural capital to ecosystem service linkages. Previous studies have investigated how different natural capital attributes are related to each other separately from how natural capital attributes are related to ecosystem services. Natural capital to ecosystem service linkages are usually explored using literature synthesis as demonstrated in Chapter 3. Alternatively, interlinkages between multiple natural capital attributes are typically investigated using primary monitoring or observed data, where a natural capital attribute often acts as a proxy for an ecosystem service. Synergies and trade-offs between ecosystem services are then investigated using some form of correlation analysis (Spearman or Pearson's). Examples of such studies include Le Clec'h et al. (2018), Maskell et al. (2013), Qin et al. (2015) and Sylla et al. (2020). In Chapter 4 we have brought these two approached together for the first time (to the best of the authors knowledge). This was achieved through the use of primary monitoring data from the Countryside Survey 2007 to evidence interlinkages between multiple natural capital attributes. This was then combined with literature-based information from the LiNCAGES platform (filtered to the context of the Countryside Survey 2007), which provided evidence on the natural capital to ecosystems service linkages. Primary monitoring data schemes such as the Countryside Survey 2007 are expensive (Burkhard and Maes, 2017; Heink et al., 2016) so are not available on the global scale, therefore this analysis was limited to the extent of the Countryside Survey 2007 (i.e. Great Britain).

We found that accounting for interlinkages between multiple natural capital attributes identified new indirect linkages between natural capital and ecosystem services. For example, interlinkages between soil carbon content, soil moisture content, soil organic matter content, low soil pH and wetlands habitat area were identified. When considering the natural capital to ecosystem service linkages with all these interlinked natural capital attributes (rather than only soil carbon content) new positive and negative linkages are revealed with the ecosystem services: air quality regulation, aesthetic landscapes, water supply, timber production, pest regulation and freshwater fishing.

We also found that interlinkages between different natural capital attributes can be context dependent. The presence of some natural capital interlinkages varied depending on whether the monitoring data was from upland or lowland locations. A surprising outcome was that when only using monitoring data from sites located within National Nature reserves (NNRs) or Sites of Special Scientific interest (SSSIs) we observed the same natural capital interlinkages as when we used evidence from all the monitoring sites. A possible explanation for this may be due to the inclusion of SSSIs and NNRs across uplands and lowlands, resulting in the differences in natural capital interlinkages balancing out, however the confirmation of this explanation requires further research.

The interlinkages between natural capital attributes in this study were identified statistically and, hence, may not be causally related. They thus require careful interpretation and possibly additional assessment using more targeted data or literature. Nevertheless, the approach reveals possible new relationships across ecosystem service cascades for consideration by researchers or decision-makers. Even if these represent an overestimation of meaningful natural capital interlinkages, it potentially brings to light linkages that may have been previously unknown for the focus of new research.

The study also highlighted key data limitations in monitoring datasets as many of the natural capital attributes in the LiNCAGES platform could not be effectively represented with biophysical measurements from the Countryside Survey monitoring dataset, which has one of the highest degrees of co-location of different measurements across Europe. Further work is required to obtain monitoring data for these natural capital attributes. Given the high cost of monitoring surveys, such investments should better align the data collected with the needs of the data in ecosystem service decision making, such as co-locating the collection of both biophysical and social and economic measurements across the cascade. Inclusion of additional natural capital attributes will potentially reveal further natural capital interlinkages and thus further natural capital to ecosystem service linkages that are important to decision makers. This study in Chapter 4 was carried out for Great Britain, further studies could repeat this approach using primary monitoring datasets from other countries and assess the context dependency of natural capital attribute interlinkages between countries.

Despite these limitations, Chapter 4 represents an important starting point for decision-makers and academics, highlighting where further literature research or data collection may be needed to confirm the identified natural capital interlinkages and natural capital to ecosystem service linkages for their own context. Chapter 4 raises awareness of the importance of accounting for both natural capital interlinkages and natural capital to ecosystem service linkages concurrently in order to provide a decision-maker with more comprehensive evidence of possible trade-offs and synergies that may be important to their ecosystem service provision.

6.2.3 To compare how the evidence type used for natural capital to ecosystem service linkages can influence maps of ecosystem service provision

Chapters 3 and 4 have shown the importance of holistically accounting for the complexities and context dependencies underlying the links between natural capital and ecosystem services. However, assumptions, limitations and subjectivity are also present in the methods and evidence types used to investigate such linkages. Chapter 5 compares ecosystem service provision maps created using different evidence types. Maps are mandatory instruments for landscape planning, environmental resource management and spatial land use optimisation (Burkhard and Maes, 2017; Crossman et al., 2012; Vorstius and Spray, 2015). Therefore, ecosystem service provision maps are a strong tool for communicating the importance of natural capital to ecosystem service linkages to decision makers (Czúcz et al., 2020; Dunford et al., 2018; Vorstius and Spray, 2015).

Ecosystem service maps are commonly created using land-use matrix approaches based on expert elicitation (Burkhard et al., 2009; Stoll et al., 2015), or by using models (Harrison et al., 2019). In addition (for what the author believes is the first time), this thesis presents ecosystem service maps created solely using literature synthesis evidence from the LiNCAGES platform (filtered to the mapping context of Europe) in a matrix-based transfer approach.

The difference between the ecosystem service maps produced by these three approaches was investigated. This comparison allowed for the identification of the differences in ecosystem service provision between maps created with different evidence types, the characteristics of the evidence type that are responsible for this difference and whether these differences are more apparent in certain locations or for certain applications. Additionally, by comparing the novel literature evidence-based maps to existing ecosystems service mapping approaches the feasibility of using an interactive literature synthesis for mapping ecosystem service provision can be assessed.

The agreement between ecosystem service maps derived from the different evidence types changed according to region and ecosystem service mapped. Good agreement was found between the expert and literature evidence type maps. This was at least partially expected as they both use the same simple land use to ecosystem service matrix lookup-based transfer approaches. Yet, agreement between two evidence types gives greater confidence in both approaches; the more subjective expert derived scores are supported by literature values based off quantitative studies. However much less agreement was found between the comparison of the modelled and other evidence type maps. This is due to the differences in ecosystem service definitions used, and levels of complexity captured by the evidence types. For example, the modelled evidence maps only assigned timber production to the managed forests, whereas the expert and literature evidence maps assigned timber production to all forests regardless of management status. However, this does not necessarily mean that one map is more accurate than another, rather than one may be more relevant than another under certain contexts or user desired applications.

Future ecosystem service provision maps for different socioeconomic and climate scenarios were also created for the three evidence types and compared in Chapter 5. Ecosystem service provision maps for future scenarios are important to inform longer term management decisions (Dunford et al., 2015). They enable considerations of future uncertainties against which the resilience of longer-term management decisions can be assessed. However, we found that future ecosystem service maps created using the three evidence types produced different directions of change in overall ecosystem service provisions for Europe in some cases.

Clearly this variation in agreement between the evidence types for mapping ecosystem service provision is a key issue for decision makers as the choice of evidence type used in the map will influence decisions made. Therefore, it is essential that map makers successfully communicate the definitions of the ecosystem services used, the methods, intended application and strengths and limitations of the evidence underlying their maps.

This will allow decision makers to understand how the evidence underlying the maps may be related to their context so that they can have an informed choice of the best evidence for their application. However, it is important to recognise that no evidence type will be a perfect fit to context, and other considerations, such as expertise, time span, funding available of the project, will also be critical criteria in choosing an appropriate mapping approach (Burkhard and Maes, 2017; Dunford et al., 2018). Where possible, decision makers should compare multiple ecosystems service provision maps created using different evidence types to identify potential areas of disagreement in ecosystem service provision. This can enhance understanding and build confidence in the results for the decision maker and allows for targeting of areas for further research (Dunford et al., 2018).

Chapter 5 compares only three methods that use three different evidence types. There are many more mapping methods available (Harrison et al., 2018), some of which are outlined in Section (existing tools section of lit review). Furthermore, this comparison of maps has been completed at a coarse resolution of 10 x 10 arcmins. A comparison of the different evidence type maps at a smaller scale and higher resolution would provide further insights between the evidence types for decision makers who work at smaller scales. This would also allow comparison with maps based on primary data from ecosystem service indicators.

6.2.4 Show how different evidence types can be combined to further support environmental decision making based off linkages between natural capital and ecosystem services

The combining of evidence types is useful as it allows for additional insights and applications (Dunford et al., 2018). However, the more evidence types that are combined and the greater the complexity of the methods, the harder it is to understand and disentangle the strengths, assumptions, and limitations of the resulting amalgamation of evidence. Hence, the term black box is sometimes used when critiquing complex ecosystem service modelling approaches (Harrison et al., 2018; Shoyama et al., 2017; Vorstius and Spray, 2015); where it is difficult for the intended user to understand the complexities and assumptions that are involved in producing the output they use to make a decision.

This is not to say that black box methods are not useful. Some ecological processes and relationships require extensive expert knowledge to understand them. However, Chapter 3 and

Chapter 4 show high context dependency in the linkages underlying ecosystem service provision, In order for the intended user to understand how methods for evidencing these linkages will perform for specific contexts and applications, greater transparency in black box methods is required.

This thesis has explored and recognised the value of combining evidence of different types. Chapter 3 follows the creation of the LiNCAGES platform to synthesize literature evidence for natural capital attribute to ecosystem service linkages. Literature evidence is made up of peer reviewed scientific studies that use a variety of different methods and evidence types. To account for these differences in evidence type used, the LiNCAGES platform allows the user to weight or filter the studies by their data type (qualitative, quantitative or both), direction of evidence (direct or indirect) and evidence type (empirical, modelling or both). However, when collating such a wide range of evidence based on different approaches, definitions and indicators used by the studies, it is not possible to undertake more specific assessment such as identifying the strength of evidence for a specific linkage (Smith et al., 2017).

In Chapter 4 we combine literature evidence on natural capital attribute to ecosystem service linkages with evidence from primary monitoring data on relationships between multiple natural capital attributes. Primary monitoring data was obtained from the Countryside Survey 2007 as it is a unique dataset of co-located measurements for Great Britain that is well-suited for understanding complex relationships between natural capital attributes. The consistent sample methodology of this survey reduces methodological context dependency of this evidence base. This is a significant advantage over literature evidence as different literature studies use different methodologies.

Combining the primary monitoring evidence with the literature evidence required the identification of biophysical measurements in the Countryside Survey 2007 that were representative of the natural capital attributes in the LiNCAGES platform. This linking of the two datasets can be subjective as some of the natural capital attributes in LiNCAGES such as “soil” were too broad and required further breaking down. This involved re-analysis of the papers evidencing this broad link to break it down into the individual properties, which is time consuming and costly. Natural capital attributes such as “presence of a specific species type” were too broad to be represented by Countryside survey 2007 biophysical measurements; others such as “stem density” were not recorded. In Chapter 4 we only combine two different evidence types to account for both interlinkages between multiple natural capital attributes and how this informs linkages between natural capital to ecosystem service linkages. As more evidence types are combined it is evident that the limitations associated with joining the evidence types by common attributes will increase and could quickly become confused or overlooked (Dunford et al., 2018).

In Chapter 5 we compare maps of future ecosystem service provision for possible climate change and socioeconomic change scenarios created using different evidence types. Modelled evidence can be used to predict future scenarios. However, the expert and literature ecosystem service provision maps are created using a land cover map and a land cover to ecosystem service matrix. Land cover maps are created with remotely sensed data and therefore cannot be used to show possible future land covers; yet modelled evidence can also be used to predict future land use/land cover. In Chapter 5 we combine the modelled future land use/land cover from an integrated model (the IAP2) with the expert or literature land cover to ecosystem service matrix to create expert or literature-based future ecosystem service provision maps respectively.

To effectively join the modelled land use/land cover data to the land cover to ecosystem service matrices a spreadsheet that refined the eight land use allocations (intensive arable, intensive grassland, extensive grassland, very extensive grassland, unmanaged land, managed forest, unmanaged forest and urban) into 22 land classes that could be matched to the CORINE land cover classes used in the expert matrix was created. The refining of the land use allocations was informed by the CORINE 2018 land cover map and other outputs from the IAP2 such as irrigation and climate suitability. This refinement required time and expert knowledge, although the refining of these land classes was informed by scientific reasoning and knowledge, it does add an element of subjectivity.

As expected, we found the comparisons of the future expert and literature ecosystem service provision maps to the modelled evidence maps to show more agreement when based on the modelled land uses. In Chapter 5 we found discrepancies between the simulated baseline of the IAP2 and the CORINE land cover map. So, in combining the modelled evidence with the literature or expert evidence, the resultant maps now share this limitation of the modelled data. Furthermore, the land cover to ecosystem service matrices for both the expert and literature evidence are based on the present-day scores derived from the literature or expert opinion. This link between land cover and ecosystem service provision may change for different future scenarios as future societies may value natural capital and ecosystem service differently than we do at present. However, the ability to create future ecosystem service provision maps with the literature and expert evidence allows for the mapping of many more ecosystem services. For example, the land cover to ecosystem service matrix used in this study includes 31 ecosystem services (Stoll et al., 2015), compared to the 8-10 available in the IAP2 (modelled evidence).

Throughout this thesis, the combining of different evidence types has been shown to provide new important insights. However, as discussed above it does come with challenges and limitations. There will always be the trade-off between simplicity and complexity in communicating ecosystem service provision (Boerema et al., 2017). Yet, it is important to present these strengths, assumptions, and limitations in a transparent and easy to understand format that is understandable to decision makers.

6.2.5 To create a platform to support collation, exploration, and synthesis of evidence on linkages between natural capital and ecosystem services and its communication in environmental decision making.

LiNCAGES was created to provide a platform that presents evidence for natural capital attribute to ecosystem service linkages in a simple, transparent and interactive manner that enables users to account for their own context and desired application (Linney et al., 2020). The LiNCAGES platform helps to show how different evidence types can be combined by collating and synthesizing literature evidence from a wide range of studies which use different evidence types and methodological approaches. This allows the user to better understand the strengths, assumptions, and limitations of the evidence underlying natural capital to ecosystem service linkages. LiNCAGES has been strongly informed by both stakeholder and researcher feedback throughout its development during this PhD. LiNCAGES has been demonstrated at over eight events ranging from exhibit stands at conferences to presentations at a Local Nature Partnership meeting.

Without regular updates live platforms such as LiNCAGES can become outdated. Throughout this PhD the LiNCAGES platform has been updated with new features and refinements to improve the user's experience. With the latest update of LiNCAGES (updated from the first version outlined in Chapter 3) users can add additional papers to the platform. To add a paper the user must fill in the required information about that paper in a user interface within the LiNCAGES platform following the guidance given within the platform. This information is then sent to the admin of LiNCAGES for approval. Once approved the paper is added to the underlying literature base of the platform. This feature supports the collation of literature evidencing natural capital to ecosystem service linkages and supports the longevity of the LiNCAGES platform.

In addition to the applications of the LiNCAGES platform demonstrated in Chapter 3, the LiNCAGES platform can also be used to assess how well common indicators for the provision of particular ecosystem services represent all of the natural capital to ecosystem service linkages underlying that ecosystem service. Maes et al. (2016) has shown that the quality of ecosystems service indicators can vary. It is dangerous to select just one potential indicator without evidence that this indicator is representative of the ecosystem service and the natural capital supporting it as a whole (Boerema et al., 2017; Heink et al., 2016). The LiNCAGES platform can display all the natural capital attributes that influence an ecosystem service. By identifying which of these natural capital attributes underly a common indicator for an ecosystem service and which do not, the LiNCAGES platform can be used to assess how well this ecosystem service indicator represents the provision of the ecosystem service. An example of this application can be found in the documentation within the LiNCAGES platform.

As discussed in Chapter 5, maps are important for decision making (Burkhard and Maes, 2017). The most common approach for creating such maps is using expert elicitation to link land cover classes to ecosystem service provision (Wong et al., 2015). The advantage of these maps is that they are simple to make and understand and are much less time, data, and expertise expensive than other methods. However, they do require a stakeholder participatory workshop to generate the scores in the matrix linking land cover to ecosystem service provision, to ensure that the scores are relevant for the intended application and context (Burkhard et al., 2012). Participatory workshops are expensive. This has led to some studies using existing land use to ecosystem service matrices from other studies with vastly different contexts (Campagne et al., 2020). For example, many studies use the values from Burkhard et al. (2012), which were developed for the context of "normal landscapes" in northern Germany (Campagne et al., 2020). Clearly there is a demand from decision makers and researchers for the quick creation of ecosystem service provision maps using matrix-based approaches that are relevant to their intended application and context, but without the need for a stakeholder workshop to generate the scores.

To contribute towards meeting this demand the latest update to the LiNCAGES platform allows for the creation of ecosystem service provision maps from its underlying literature evidence base. These maps are created using the method outlined in Section 5.3.1. Now that ecosystem service provision maps can be created within the LiNCAGES platform, the user can control the context of the natural capital attribute to ecosystem service linkages that contribute the values in the land cover to ecosystem service matrix that underlies the maps through filtering and weighting. This allows for the creation of ecosystem service provision maps with evidence relevant to the user's context in a few seconds at no cost. This is a huge cost saving over organising a stakeholder workshop to generate expert scores. Furthermore, the flexible nature of the LiNCAGES platform allows the user to control all aspects of the natural capital to ecosystem service linkages that contribute to the values in the land cover to ecosystem service matrix. For example, the user can choose to explore a map for one ecosystem service or for combined ecosystem service provision, through selecting multiple ecosystem services within the LiNCAGES platform.

A major limitation of land cover to ecosystem service matrix-based approaches is that they are entirely land cover based (Burkhard and Maes, 2017; Campagne et al., 2020) and are therefore vulnerable to large errors (Wong et al., 2015). This is also the case for the maps produced in the LiNCAGES platform as the natural capital attribute to ecosystem service linkages evidenced by a study are assigned to the land cover associated with the ecosystem the study was completed in. However, maps within the LiNCAGES platform are created using natural capital to ecosystem service linkages from up to 42 natural capital attributes. This gives the user control over types of natural capital attributes that contribute to their maps of ecosystem service provision. For example, a user could filter for only natural capital attribute to ecosystem service linkages that are from habitat or landscape level natural capital attributes and investigate how the maps change as a result. This flexibility and transparency of the LiNCAGES platform allows a researcher to investigate natural capital attributes of their land cover that are responsible for the ecosystem services they are interested in. This provides additional insight and explanation for the values behind the ecosystem service provision maps that may not be available from expert derived scores.

6.3 Limitations and further work

Throughout this thesis we have used literature synthesis data to evidence natural capital to ecosystem service linkages. A large limitation in literature synthesis data is reporting bias (Adhikari and Hartemink, 2016; Balvanera et al., 2006; Hevia et al., 2017). Although the LiNCAGES platform does attempt to identify this reporting bias by highlighting which natural capital attribute to ecosystem service linkages may be missing evidence, it still suffers from reporting bias due to bias against publishing non-significant or less interesting results (Ricketts et al., 2016; Schwarz et al., 2017).

Furthermore, although LiNCAGES currently allows users to add new studies to the platform, this evidence will not be collected in a systematic representative way. For this reason, the LiNCAGES platform offers the ability to filter the evidence so that only studies from the original OpenNESS review are included. Further work could involve the incorporation of existing and future systematic literature synthesis into the platform, particularly for marine and coastal natural capital to ecosystem service linkages. This will involve working with researchers designing and undertaking future reviews to ensure compatibility.

Throughout this thesis there are many examples of pooling and combining evidence types, from pooling the literature evidence in the LiNCAGES platform in Chapter 3, combining monitoring and literature evidence in Chapter 4, through to combining modelled evidence with literature and expert evidence in Chapter 5. Pooling evidence can sometimes oversimplify and mislead both scientific syntheses and management interventions (Martnez-Harms and Balvanera, 2012; Ricketts et al., 2016). Different evidence types and papers have different definitions for ecosystem services and natural capital attributes (Burkhard and Maes, 2017; Haines-Young and Potschin, 2018; Smith et al., 2017), which makes the combining of evidence types subjective and can leave gaps. For example, in Chapter 4 we could not represent all natural capital attributes in the LiNCAGES platform with biophysical measurements from the Countryside Survey 2007, due to the broad and high-level definition of some of the natural capital attributes in LiNCAGES, whereas the Countryside Survey biophysical metrics are more specific.

Although we used LiNCAGES evidence in Chapter 4, when exploring how accounting for interlinkages across ecosystem service cascades can reveal new indirect natural capital to ecosystem service linkages, this analysis is not incorporated within the LiNCAGES platform. This is due to the inability to represent all the natural capital attributes within the LiNCAGES platform with Countryside survey 2007 biophysical metrics and their limited context to Great Britain. As a result, the inclusion of this analysis would cause more confusion than benefit when incorporated within the LiNCAGES platform. The analysis can instead be found in a separate web application at https://glinney.shinyapps.io/NCA_interlinkages_and_ES/. Further work is required to find larger scale monitoring datasets that can be effectively used to represent all of the natural capital attributes within the LiNCAGES platform.

In this Chapter and Chapter 5 we have discussed the creation, and advantages, of literature-based ecosystem service provision maps. However, the creation of such maps requires the pooling of literature evidence by assigning all the natural capital attribute to ecosystem service linkages evidenced by each study to the ecosystem the study was located in. This leads to the under representation or double counting of studies that span multiple ecosystems depending on whether the user decides to include such studies. Further limitations result from the subjectivity of linking ecosystems to land cover classes. The LiNCAGES platform categorises studies to the 12 ecosystems classified under Action 5 of the EU biodiversity strategy to 2020 (Maes et al., 2016). When connecting these ecosystems to the 44 land cover classes in the CORINE land cover maps the extra detail from the land cover map is lost through aggregation.

However, other existing ecosystem service mapping methods such as InVEST use pooling and aggregation of data (Nelson et al., 2009). Despite the limitations in the literature maps, Chapter 5 shows that literature maps created by the LiNCAGES platform for Europe agree well with expert derived scores that had been adjusted for Europe through consultation with the site expert teams across 28 sites and 11 countries that worked on the European Long-Term Ecological Research network (LTER-Europe) sites (Stoll et al., 2015). Yet it is still important to recognise that, like expert elicitation derived ecosystem service provision maps, ecosystem service provision maps created using the LiNCAGES platform should only be used as a guide, or as a starting point, to improve understanding and awareness of natural capital to ecosystem service linkages that might be important for ecosystem service provision in the absence of local data or detailed models (Burkhard and Maes, 2017; Linney et al., 2020).

Currently, the ecosystem service provision maps that can be generated using the LiNCAGES platform are for Europe and for Great Britain. A future update will add the ability to create global ecosystem service provision maps.

Finally, it must be remembered that this thesis has focused only on one linkage in the ecosystem service cascade: the link between natural capital and ecosystem services. Ecosystem services are not solely based on natural capital. It is widely acknowledged that ecosystem services are produced in socio-ecological systems and human actions are key for driving the provision of ecosystem services (Heink et al., 2016). Furthermore, the provision of many ecosystem services such as food production also require manufactured and human capital such as machines, skills, and labour (Heink et al., 2016). To comprehensively assess ecosystem service provision we must account for each step in the ecosystem service cascade (Boerema et al., 2017; Harrison et al., 2017).

The majority of studies attempt to account for the whole ecosystem service cascade to varying levels of success for one ecosystem service. As we have shown in this thesis, investigating only one ecosystem service will not account for important trade-offs and synergies involved in the supply of multiple ecosystem services. In an effort to achieve the goal of operationalising the ecosystem service cascade we propose the investigation of each link in the cascade holistically while accounting for context dependency. Similar platforms to LiNCAGES could be created for the link between ecosystem services and benefits, benefits to values and drivers to natural capital. These platforms should be designed with their final integration in mind.

6.4 Concluding remarks

The aim of this thesis was to contribute to sustainable environmental management through understanding the linkages between natural capital and ecosystem services. This thesis has shown the importance of holistically investigating the evidence behind natural capital to ecosystem service linkages, while accounting for context dependency. Additionally, it reveals how accounting for interlinkages across ecosystem service cascades can reveal new natural capital to ecosystem service linkages. Furthermore, it shows the importance of communicating the strengths, assumptions and limitations of the evidence underlying tools that communicate natural capital to ecosystem service linkages to decision makers via ecosystem service provision maps. A tool in the form of the LiNCAGES platform has been provided to collate, explore, and synthesise such evidence, including the creation of novel ecosystem service provision maps based on literature evidence. The work presented in this thesis and the LiNCAGES platform provides new insight into the context dependencies in the evidence used to assess ecosystem service provision as well as the promotion of a holistic approach for the assessment of ecosystem services. This will provide a step towards the operationalisation of the ecosystem service cascade to support sustainable and effective environmental management decisions necessary for the preservation of our vital ecosystem services.

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Appendices

A. Supplementary material for Chapter 2

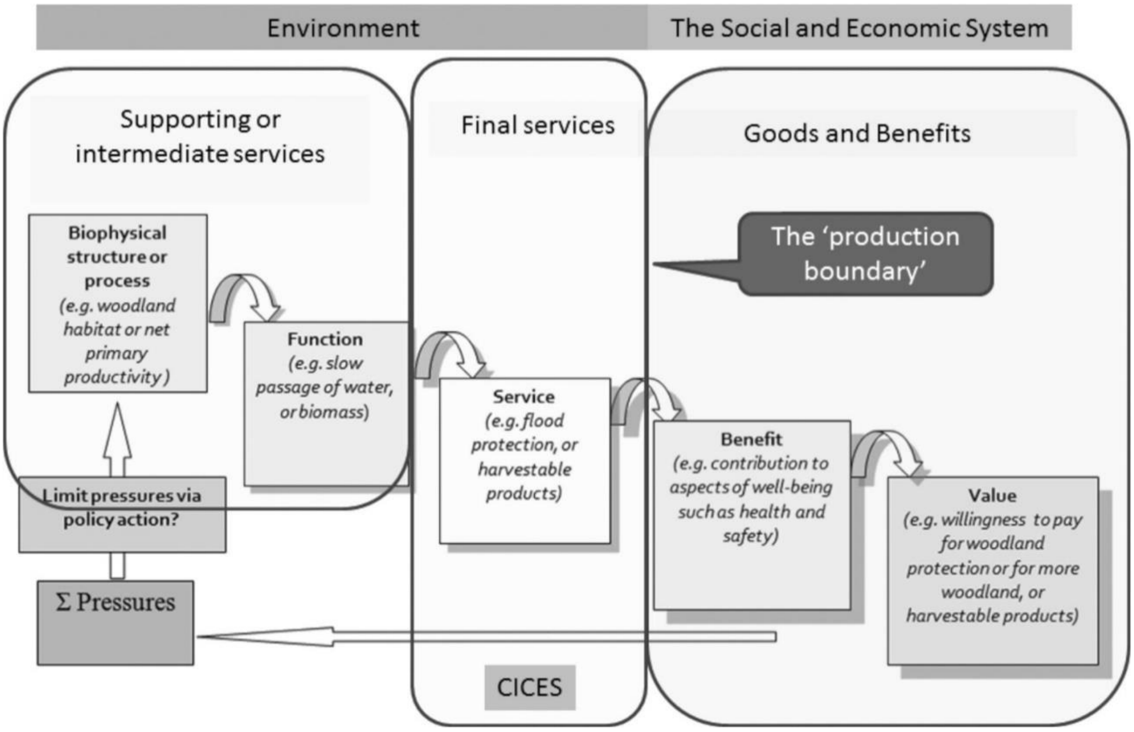


Figure A1: The cascade model (Haines-Young and Potschin, 2018).

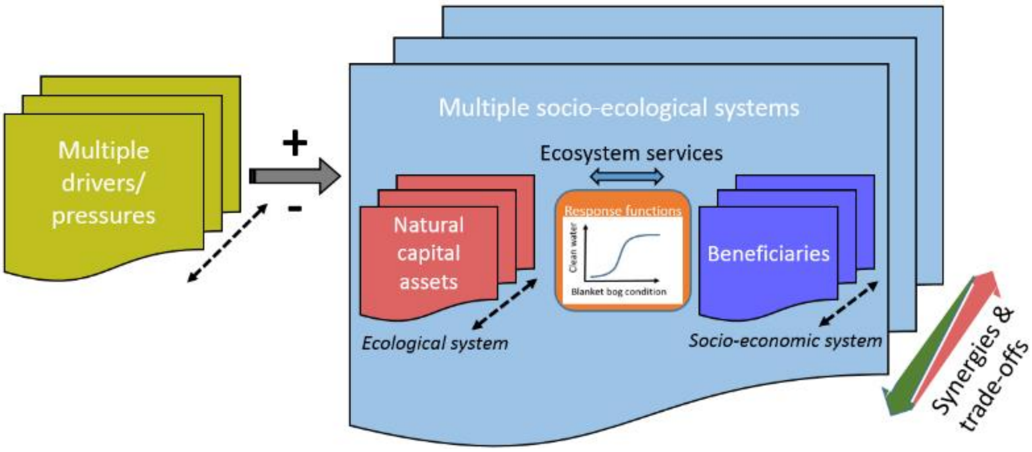


Figure A2: Natural Capital metrics project conceptual framework for multiple socio-ecological systems (Harrison et al., 2017).

B. Supplementary material for Chapter 3

1. Existing systematic reviews on NC-ES linkages or equivalent

Table B1: Comparison of existing systematic reviews on NC-ES linkages or equivalent that could be added to the LiNCAGES platform.

	Harrison *et al.* (2014)	Smith *et al.* (2017)	de Bello *et al.* (2010)	Hevia *et al.* (2017)	Ricketts *et al.* (2016)	Balvanera *et al.* (2006)	Schwarz *et al.* (2017)
	Smith *et al.* (2017) builds on Harrison *et al.* (2014)						
Title	Linkages between biodiversity attributes and ecosystem services: a systematic review	How natural capital delivers ecosystem services: A typology derived from a systematic review	Towards an assessment of multiple ecosystem processes and services via functional traits	Trait--based approaches to analyse links between the drivers of change and ecosystem services: Synthesizing existing evidence and future challenges	Disaggregating the evidence linking biodiversity and ecosystem services	Quantifying the evidence for biodiversity effects on ecosystem functioning and services	Understanding biodiversity-ecosystem service relationships in urban areas: A comprehensive literature review
Aim	Analysed linkages between different biodiversity attributes and 11 ecosystem services.	• Identify the NCA that link natural capital stocks to ecosystem service flows in different contexts. • Develop a simple typology for understanding and classifying the links between NCA and ES delivery.	Synthesize concepts and empirical evidence on linkages between functional traits and ecosystem services across different trophic levels and spatial scales.	Link direct drivers of change and ecosystem services via functional traits of three taxonomic groups (vegetation, invertebrates, and vertebrates) to: • Uncover trends and research biases in this field. • Synthesize existing empirical evidence.	More deeply understand the evidence linking biodiversity and four services: carbon storage, pest control, crop pollination and water purification.	Complete the first rigorous quantitative assessment of biodiversity-ES relationships through meta-analysis of experimental work spanning 50 years to June 2004.	Built on the work of Ziter (2016), by conducting a wider search for publications examining urban biodiversity-ES relationships.
Data	Data from the 530 papers on the direction of relationship between NCA and ES.	780 papers, recording on NCA-ES linkages (42 NCA; 13 ES) until the end of June 2014.	• 247 references documenting trait effects on ecosystem properties/services, evidenced from peer-reviewed literature published up to 2007. • Only studies that showed statistically significant associations between traits and ecosystem processes and/or services, based on quantitative data, were considered. • Only dealt with studies that	• The first database of 125 was used to characterize the current state and trends of trait-based ecosystem services research. • The second database was traits-oriented and only considered those statistically significant relationships		• 103 publication, representing 446 ecosystem property measurements from 1954 to June 2004 using ISI Web of Science and Biological Abstracts database. • Excluded duplicates (same experiment and same measurement reported in a	317 studies published between 1990 and May 2017 that examined urban biodiversity ecosystem service relationships.

	Harrison *et al.* (2014)	Smith *et al.* (2017)	de Bello *et al.* (2010)	Hevia *et al.* (2017)	Ricketts *et al.* (2016)	Balvanera *et al.* (2006)	Schwarz *et al.* (2017)
	Smith *et al.* (2017) builds on Harrison *et al.* (2014)						
			explicitly considered functional traits and/or functional groups of organisms. • Excluded publications that simply reported the effects of species diversity or composition. • Double counting of a given trait–process relationship based on the same experimental evidence was avoided.	among drivers of change, functional traits, and ecosystem services found in the existing literature, consisting of 83 observations, from 71 papers.		different publication or measured in a different year). • Used studies that recorded effect size. Did not include studies that only reported significance.	
Number of ES included	11	13	9	13	4	5	12
Method	**Vote counting** with the parameter "strength of evidence for positive or negative relationship" so that those records offering weak evidence had a smaller influence.	**Vote counting,** same methodology as Harrison *et al.* (2014).	**Vote counting** of statistically significant reported trait-service associations.	**Vote counting** of studies that reported significant evidence.	**Vote counting**, as it allowed the coding of studies consistently across widely varying disciplines.	**Meta-analysis.**	**Vote counting.**

	Harrison *et al.* (2014)	Smith *et al.* (2017)	de Bello *et al.* (2010)	Hevia *et al.* (2017)	Ricketts *et al.* (2016)	Balvanera *et al.* (2006)	Schwarz *et al.* (2017)
	Smith *et al.* (2017) builds on Harrison *et al.* (2014)						
Methods and visualisation	• Number of papers showing evidence for a linkage. • A strength of evidence parameter was also derived for each linkage ranging from 1 (very weak) to 5 (very strong). • Visualized as frequency tables and network diagrams.	• Number of papers showing evidence for a linkage. • Visualized as frequency tables and network diagrams.	• Used multivariate analysis using Detrended Correspondence Analysis (DCA) to cluster traits and services to identify which services are provided by similar traits.	• Conducted six different redundancy analyses (RDAs) on the interlinkages between drivers of change and functional traits, as well as between functional traits and ecosystem services. • The multivariate analyses allowed identification of key functional traits, which have the potential capacity to provide multiple ecosystem services while responding to specific drivers of change.	• Analyzed the distribution of relationships among positive, negative and non-significant categories, and referred to this as the 'balance of evidence'. • Tested differences in the distribution of relationships using likelihood-ratio tests (G-test), and tested differences in scale among positive, negative and non-significant relationships using analysis of variance.	• Used correlation coefficients to assess the strength of the biodiversity-ecosystem property relationships. • Accounted for study sample size, by using it as a weighting factor for ANOVA tests. • Ecosystem properties that could be related to ecosystem services, and thus that could be assigned a positive (or negative) value for human wellbeing, were further analysed based on mean values and standard errors of effect sizes.	• Derived information on the evidence, basis, direction and statistical significance of biodiversity-ES relationships. • The numbers of studies reporting different categories of biodiversity-ES relationships were examined using descriptive statistics.
Methods: limitations	• Based on only the frequencies of citation in the literature, which is not necessarily the same as functional importance. • Additional research is needed into the linkages represented by the thin lines in the networks – the linkages exist, but how strong are their functional roles in joining the different aspects of biodiversity with the provision of ecosystem services in the amounts required by beneficiaries? • Need to take into	• Vote-counting used as meta-analysis was not possible for such a diverse dataset using so many incompatible indicators and approaches. • Number of papers citing a positive or negative link is not proportional to the importance or strength of that link. • The absence of evidence for a link does not necessarily mean that the link does not exist, but only that evidence	• Vote counting will also capture the ease of obtaining associated measurements or the prevalence of the hypothesis from the literature. • The magnitude of each trait–process relationship was not taken into account to avoid biased comparisons across different methodological approaches in different studies and also because this was often not explicitly indicated. • The review most likely does not include all existing studies in the literature on the link between traits and ecosystem functions.	Could not incorporate any weighting of the magnitude of the responses and/or effects, so there may be an overrepresentation of those functional traits that have been most frequently investigated.	• Findings may be sensitive to the search terms and search tool used to identify relevant papers. • Likely include reporting bias, in which non-significant or otherwise less-compelling results are less likely to be published. • The strength of reporting bias may differ among the widely divergent fields contributing relevant literature to the study of ES.	• Some judgment is involved in the assignment of a positive or negative value to ecosystem property – ecosystem service relationships, because a particular ecosystem property may not be seen as the same benefit by all stakeholders.	• Meta-analysis could not be conducted because of the lack of suitable quantitative data. • Because limited to an urban context, might have excluded papers that looked at the indirect effects of biodiversity that are much harder to quantify. • Restricted to peer-reviewed journal papers. • Bias from statistically significant relationships, negative or positive,

	Harrison *et al.* (2014)	Smith *et al.* (2017)	de Bello *et al.* (2010)	Hevia *et al.* (2017)	Ricketts *et al.* (2016)	Balvanera *et al.* (2006)	Schwarz *et al.* (2017)
	Smith *et al.* (2017) builds on Harrison *et al.* (2014)						
	account effects of socio-economic factors and land use decisions on different components of biodiversity and, as a result, ecosystem service delivery.	for it has not been reported in the literature.					being more likely to be published.
Aspects included	• Location of the study. • Spatial scale. • Temporal scale. • Ecosystem Service Provider (ESP). • Important biotic attributes of the ESP. • Abiotic factors which affect service delivery. • Whether the evidence is qualitative, quantitative or both. • Whether the evidence based is off single or multiple observations. • Whether the evidence is direct or indirect. • Whether the linkage explicitly mentioned or only inferred. • Whether the evidence is based on empirical data, modelled information or both.	• The ecosystem service covered. • The location of the study. • Type and condition of ecosystems. • Management. • ESP. • The indicators used to assess the level of service provision. • Any qualitative or quantitative information on interactions between different ecosystem services, and the direction of interaction. • Any qualitative or quantitative information on human input and management, and its direction of impact. • Any evidence for thresholds or tipping points. • However, they do not discuss context dependent aspects. Spatial scale, temporal scale and type of evidence in this paper.	• Nature of the study (a review or primary data publication). • Category of ecosystem service assessed (provisioning, cultural and regulating/supporting services). • Specific ecosystem service assessed. • Ecosystem process underlying the service. • Level of organization at which traits were assessed (i.e. species, functional groups, whole community). • The specific traits. • Whether traits were assessed in combination with other traits or individually. • The relationship of traits with ecosystem processes and services (an increase in trait value/abundance improved or was detrimental to the process, or the relationship was not identifiable). • The organisms providing the service. • The ecosystem type.	Database 1 (125 papers**)** • Publication characteristics (i.e., year of publication, type of research). • Study area. • Methodological approach used (e.g., data source, theoretical or analytical approach). • Taxonomic group studied. • Ecosystem type. • Direct drivers of change analyzed. • Functional traits used. • Category of ecosystem services (i.e., provisioning, regulating, or cultural). • Specific ecosystem services investigated.	• Type of linkage (e.g., spatial, management and functional linkage). • The level of organization at which biodiversity was considered (genetic, species, taxonomic, functional or ecosystem). • The metric used to quantify biodiversity (richness, diversity index or abundance). • ESP. • The ES outcome measured (biophysical supply such as pollinator abundance, or realized benefits such as increased crop yield). • Spatial scale. • Whether the reported relationship was linear or non-linear. • Country in which the study took place. • Whether the study measured ES as biophysical supply or benefits of this supply to people. • Whether the study links ES to expected 'service providers' rather than attributes that are not expected to be related to	• Type of experimental system. • Main cause of diversity change (direct or indirect) • Maximum species number. • Ecosystem type. • Trophic level manipulated and trophic level measured. • Number of trophic links. • Effect form (shape of biodiversity-ecosystem properties relationship). • Ecosystem properties measured. • Organizational level of the ecosystem property measured. • Dominant dynamic of ecosystem property. • Nature of ecosystem property. • Ecosystem service. • Location of study site.	• Evidence type (empirically test or only assumed). • Basis of the linkage (conceptual, correlative analysis or cause effect models). • Statistical significance. • Research design (controlled or observational). • Taxonomic biodiversity metric delivering services. • Origin of the species delivering the service (native, non-native or unknown). • Whether the non-native species is invasive. • Functional biodiversity metrics delivering the service (functional identity, functional diversity or others). • Functional traits mentioned. • Biodiversity-metrics (NCAs). • Ecosystem services.

	Harrison *et al.* (2014)	Smith *et al.* (2017)	de Bello *et al.* (2010)	Hevia *et al.* (2017)	Ricketts *et al.* (2016)	Balvanera *et al.* (2006)	Schwarz *et al.* (2017)
	Smith *et al.* (2017) builds on Harrison *et al.* (2014)						
					service provision. • Level of organisation considered by the study.		
Advancing science?	Built up the scientific knowledge base on ESPs, their biodiversity attributes and the direction and strength of evidence for these relationships and the influence of abiotic factors.	The evidence base can be used to: • Demonstrate the value of NC to ensure the long-term provision of the range of ES. • Can identify opportunities to gain multiple ecosystem service benefits, and to recognise situations where there could be trade-offs between ecosystem services.	• This is the first and most complete attempt to synthesize this very scattered information. • de Bello *et al.*, (2010) believe that it this is a good representation of the knowledge that is available in the literature.	Presents the first systematic review on the entire pathway, from drivers to ecosystem services via traits, across different taxonomic groups.	The synthesis can be used to develop a broad registry of evidence, which would help scientists, managers, and policy-makers match potential interventions with the most relevant scientific information.	First rigorous quantitative assessment of biodiversity-ES relationships through meta-analysis.	Comprehensive review of biodiversity-ES relationships in urban areas.

2. Keywords used for the literature search to create the OpenNESS database.

Taken from Smith et al. (2017)

Standard set of terms related to biotic attributes

*diversity OR *diverse OR species OR habitat* OR trait* OR landscape OR richness OR abundance

Note: "OR mix*" was also used for timber production to cover species mixtures.

Service-specific terms

Ecosystem service	**Ecosystem service terms**	**Additional terms used to refine results**
Freshwater fishing	(*fish*) AND (yield OR catch OR quantity OR 'ecosystem service' OR producti*)	(freshwater OR lake* OR river* OR reservoir* OR floodplain* OR inland)
Timber production	forestry OR plantation* OR timber OR wood	Yield OR producti* OR growth OR supply OR harvest OR "basal area"
Water supply	(water OR freshwater OR groundwater) AND (supply OR provision* OR yield OR budget OR reserve* OR resource*)	(*forest* OR soil OR vegetat* OR ecosystem* OR woodland*) AND (infiltrat* OR recharg* OR runoff)

Food production (crops)	TOPIC: (Food OR crop OR agricultur*) AND TITLE: (Producti* OR yield)	NOT TITLE: grassland OR meadow OR graz* OR pasture OR aquatic OR *alga* OR fish* OR milk OR dairy OR biofuel OR bioenergy OR biodiesel OR miscanthus OR bioethanol OR *foram* OR *benth* OR *plank* OR pest OR pollin* OR predat* OR bird*
Air quality regulation	"air quality" OR "air pollution" OR particulate*	(tree* OR vegetation OR forest* OR wood*) AND (absor* OR remov* OR regulat* OR adsor*)
Atmospheric regulation (carbon storage)	“Carbon storage” OR “carbon sequestration” OR “carbon loss” OR “carbon emissions”	Tree* OR soil* OR biomass
Water flow regulation (flood protection)	Flood* OR “water flow regulation”	(Flow* OR Attenuation OR Storage OR Protection OR Defence OR Prevention OR Runoff OR Evapotanspiration OR Infiltration OR interception) AND (vegetation OR forest OR wetland OR marsh)

Water quality regulation	Water quality OR Water regulation OR Water purification OR Nutrient* retention OR Nutrient* translocation	Tree* OR Soil* OR Forest* OR Vegetation OR Plant* OR Pollutant* OR Wetland* OR Microorganism* OR Accumulation OR Sediment*
Mass flow regulation (erosion protection)	soil OR sediment OR sand AND	Root OR vegetation
	loss OR erosion OR trap* OR runoff OR stabil* OR erodab*	
Pollination	Pollinat*	yield OR Fruit OR "Seed set" OR reproduct*
Pest regulation	"Natural pest control" OR "Pest control" OR "Biological control" OR "Biological pest control"	
Species-based recreation	“species-based recreation” OR eco-tourism OR *watching OR viewing OR birding OR “nature tourism”	satisf* OR visit* OR appreciat* OR motivate* OR prefer*
Aesthetic landscapes	tourism OR recreation OR *esthetic* OR appreciation OR valuation OR preference* OR perception*	

3. Natural capital attributes in OpenNESS

From the OpenNESS deliverable report (Pérez Soba et al., 2017).

Species attributes

- **Presence of a specific species type**
- **Species abundance (**number of individuals of a species expressed per unit area or volume of space, synonymous with species population density).
- **Species richness** (number of different species represented in a set or collection of individuals)
- **Species population diversity** (the number, size, density, distribution and genetic variability of populations of a given species)
- **Species size or weight** (includes body size or weight, diameter at breast height – DBH – for trees, species/vegetation/tree height, basal area defined as the cross section area of the stem or stems of a plant or of all plants in a stand, generally expressed as square units per unit area)
- **Population growth rate** (change in the number of individuals of a species in a population over time)
- **Mortality rate** (number of deaths of individuals per unit time)
- **Natality rate** (number of new individuals produced per unit time)
- **Life span/longevity** (duration of existence of an individual/expected average life span)

Functional group attributes

- **Presence of a specific functional group type**
- **Abundance of a specific functional group**
- **Functional richness** (the number of functional groups or trait attributes in the community)
- **Functional diversity** (range, actual values and relative abundance of functional trait attributes in a given community)
- **Flower-visiting behavioural traits** well suited to the system to provide pollination ecosystem services
- **Predator behavioural traits** well suited to the system to provide biocontrol ecosystem services

Community/habitat attributes

- **Presence of a specific community/habitat type**
- **Community/habitat area** (includes width or diameter, i.e. for buffer zones)
- **Community/habitat structure** (in terms of complexity - amount of structure or variation attributable to absolute abundance of an individual structural component - and

heterogeneity - kinds of structure or variation attributable to the relative abundance of different structural components)

- **Primary productivity** (rate at which plants and other photosynthetic organisms produce organic compounds in an ecosystem)
- **Aboveground biomass** (the total mass of aboveground living matter within a given area)
- **Belowground biomass** (the total mass of belowground living matter within a given area)
- **Sapwood amount** (including allocation of carbon to sapwood and sapwood area)
- **Stem density** (measured as the number of stems/specified area)
- **Wood density** (measured as the weight of a given volume of wood that has been air-dried)
- **Successional stage** (changes in the number of individuals of each species of a community by establishment of new species populations that may gradually replace the original inhabitants; categorised into early and late stages)
- **Habitat/community/stand age** (includes young and old-growth forests, even and uneven-aged forests, or can specify the age)
- **Litter/crop residue quality** (quality of plant litter with respect to decomposition: often defined by the C:N ratio, but ratios of C, N, lignin and polyphenols are other chemical properties and particle size and surface area to mass characteristics are physical properties)
- **Leaf N content**

Abiotic attributes

- **Temperature**
- **Precipitation**
- **Evaporation**
- **Wind**
- **Snow**
- **Soil** (direction not applicable)
- **Geology** (direction not applicable)
- **Water availability**
- **Water quality**
- **Nutrient availability** (soil minerals)
- **Slope** (angle, aspect)

Other attributes

- **Landscape diversity** (diversity of landscapes and landscape features
- **Other abiotic**
- **Other biotic**

4. Context dependent study aspects available in LiNCAGES

Scale

- **Spatial scale:** The spatial scale(s) of the study(s) used in the article.
- **Temporal scale:** The temporal scale(s) of the study(s) used in the article.
- **Continent:** The continent(s) that the study(s) from the article is based in.
- **Ecosystem:** The ecosystems that were mentioned in the article.

Management

- **Management input:** whether the human inputs are directly applied to the ecosystem, or indirect. An example of an indirect input is if the article states that resource application in a neighbouring ecosystem has an impact on the ecosystem service investigated.
- **Management intensity:** Whether the inputs are intensive (significant use of capital and inputs relative to land area) or extensive (low use of capital and inputs relative to land area).
- **Management impact:** The direction of the management on the ecosystem service.

Evidence

- **Observation(s):** Whether the evidence for the linkages from the article was based on single or multiple observations.
- **Data type:** Whether the evidence for the linkages from the article was qualitative, quantitative or both.
- **Direction of evidence:** Whether the evidence for the linkages from the article was direct or indirect (through a surrogate).
- **Evidence type:** Whether the evidence for the linkages from the article was based on empirical data, modelled information or both.

Interactions between ecosystem services

- **Ecosystem services affected:** The ecosystem services that the article has identified to have a linkage with the main ecosystem service that the article is investigating.
- **Direction of effect on other ecosystem services:** The direction of the linkage(s) with the other ecosystem services.

Additional

- **Ecosystem service provider:** The populations, functional groups or communities that contribute to the provision of the ecosystem service.
- **Ecosystem service indicator assessment method:** The assessment method used to measure the indicator for the ecosystem service.
- **Period article published:** Whether the article was published before or after the Millennium Ecosystem Service Assessment (MEA) was released.

5. Working example demonstrating the hierarchical weighting and filtering in LiNCAGES

For this use case scenario, we take the role of a small-scale European forest manager who receives benefits from the ecosystem service of timber production. For the level 2 weightings, our forest manager weighted European based studies greater than non-European studies. As a local landowner, they also weighted local and sub-national studies greater than studies with other spatial scales. Finally, snapshot studies were weighted less than studies with longer temporal scales as from our forest manager discovered that snapshot studies missed the NC-ES linkages for 14 natural capital attributes with timber production. For the level 1 weighting, our forest manager decided that their temporal scale weighting choices were most important (due to the previous findings from Figure 12), spatial scale context second most important, and the continent of study third most important. Our forest manager also decided to filter out scenario analysis studies and does so by assigning scenario analysis studies a weight of zero. These weighting choices are shown by the screenshots in Figure B1. The dark and light blue boxes encompass the level 1 and level 2 study aspect weights respectively.

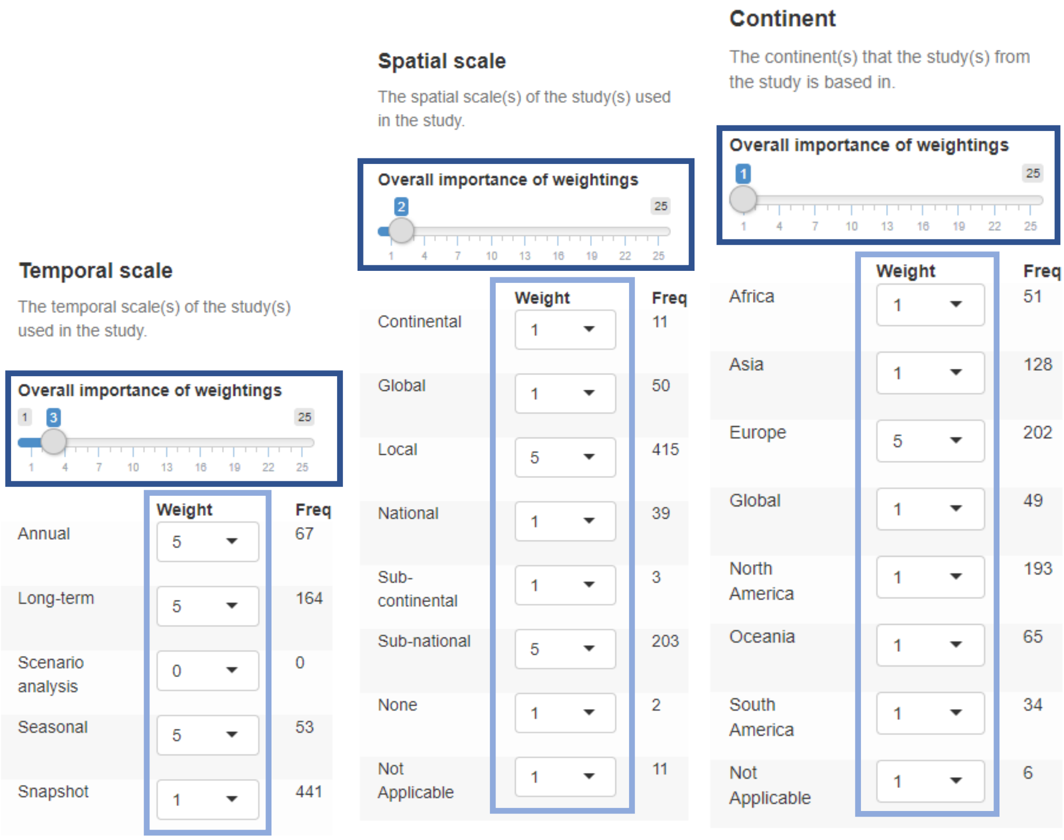


Figure B1: LiNCAGES user interface for weighting the study aspects with example user inputted weights for three of the study aspects to the context discussed above. The frequency of studies with each context is also shown. The dark blue boxes contain the user-selected level 1 aspect weight; the light blue box contains the user-selected level 2 aspect weight. Scenario analysis has been assigned a weight of zero, which will filter out scenario analysis studies from this analysis.

An article aspect dataset recording the level 1 aspects (columns) and level 2 aspects (cells) for each article (rows) is created from the OpenNESS database. See the first table of Figure B2 below. The articles were weighted by the user inputted aspect weights (Figure B1) to calculate a total weight for each article. Figure B2 shows this process.

Article	Temporal scale	Spatial scale	Continent
Article 1	Snapshot	Sub-national	c(Asia, Asia, Africa)
Article 2	Snapshot	National	North America
Article 3	Scenario analysis	Global	Europe
Article 4	Annual	Local	c(Asia, Europe)

1 Assign the user inputted weights in Figure B1 to the level 2 aspects.

Article	Temporal scale	Spatial scale	Continent
Article 1	1	5	c(1,1,1)
Article 2	1	1	1
Article 3	0	1	5
Article 4	5	5	c(1,5)

2 If an article has more than one value (if it has more than 1 level 2 aspect) use the maximum value.

Article	Temporal scale	Spatial scale	Continent
Article 1	1	5	1
Article 2	1	1	1
Article 3	0	1	5
Article 4	5	5	5

3 If '0' has been assigned for any article aspect, drop the article

Article	Temporal scale	Spatial scale	Continent
Article 1	1	5	1
Article 2	1	1	1
Article 4	5	5	5

4 Divide each level 2 aspect weight by the maximum of the level 2 user selected weights.

Article	Temporal scale	Spatial scale	Continent
Article 1	0.2	1	0.2
Article 2	0.2	0.2	0.2
Article 3	1	1	1

5

If all level 2 aspects have been weighted the same, all articles will now have a weight of 1 for that aspect. To prevent this effecting the total article weight, the weights from this aspect are reassigned as 0 (This does

Article	Temporal scale	Spatial scale	Continent
Article 1	0.2	1	0.2
Article 2	0.2	0.2	0.2
Article 3	1	1	1

6

Multiply the level 2 aspect weights by their level 1 aspect weight from Figure B1.

Article	Temporal scale	Spatial scale	Continent
Article 1	0.6	2	0.2
Article 2	0.6	0.4	0.2
Article 3	3	2	1

7

Sum the level 1 aspect weights for each article. In this example, there are only 3 level 1 aspects. In reality, there are 16.

Article	Total Weight
Article 1	2.8
Article 2	1.2
Article 3	6

Figure B2: Hierarchical weighting and filtering method process. Zigzag lines show that in reality the table continues in the direction(s) of the zigzag line.

The total article weight is applied to all of the linkages between attributes of natural capital and the ecosystem service that is mentioned by that article.

The linkages displayed in the visualisations can be filtered by their direction and the ecosystem service and natural capital attribute they are linked to.

See Figure B3 for a stacked bar chart weighted to this context ordered by the number of studies evidencing NC-ES linkages of all directions with all 13 of the ecosystem services. See Figure B4 for a comparison of the positive and negative linkage directions between all of the natural capital attributes and ecosystem services available within LiNCAGES. Alternatively, the linkages can be visualised as a network diagram. See Figure B5 for a network diagram of only the positive LiNCAGES. Within LiNCAGES the edges (linkages) in the network diagram can be selected. See Table B2 for a table of references for all of the studies evidencing positive, negative and unclear linkages between the natural capital attribute presence of a specific species type and the ecosystem service of timber production (the selected edge in Figure B5) under the context shown in Figure B1.

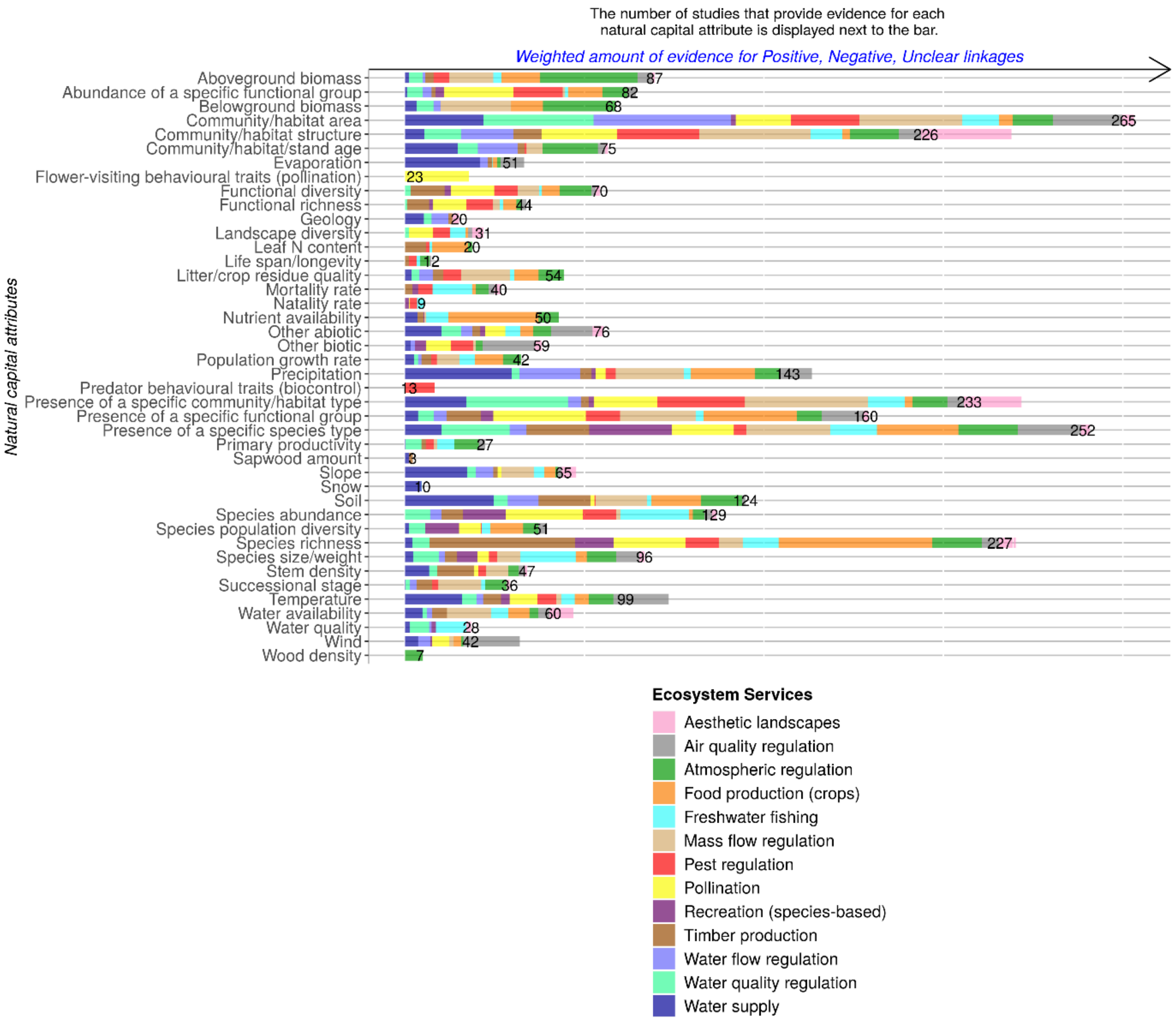


Figure B3: Stacked bar plot showing the weighted amount of evidence for all directions of linkages together (positive, negative and unclear) between all of the natural capital attributes and all of the ecosystem services within LiNCAGES, weighted to the contexts shown in Figure B1. Each bar is labelled with the amount of studies that evidence the linkage so that the effect of weighting can be easily observed.

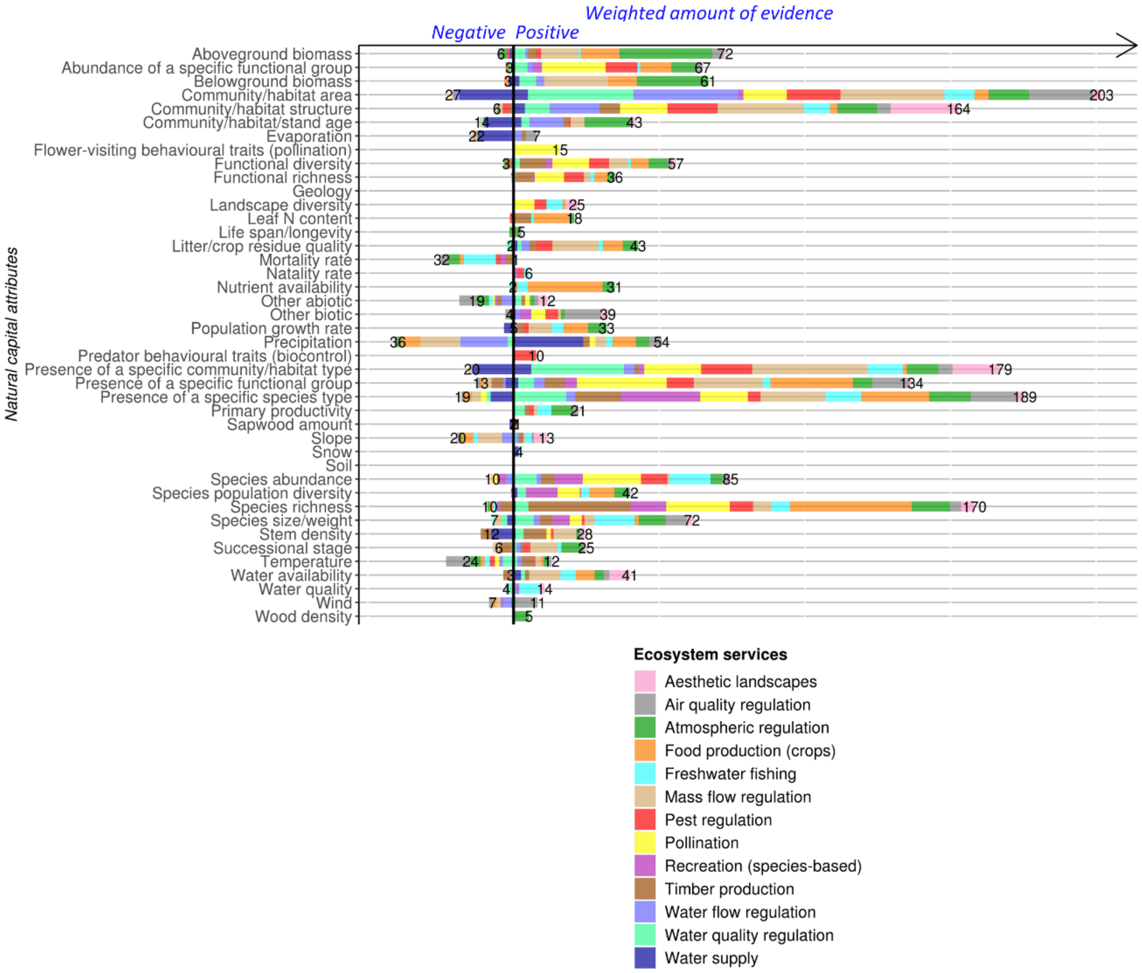


Figure B4: Stacked bar plot showing the weighted amount of evidence for positive and negative linkages between all of the natural capital attributes and all of the ecosystem services within LiNCAGES, weighted to the contexts shown in Figure B1. Each bar is labelled with the amount of studies that evidence the linkage so that the effect of weighting can be easily observed.

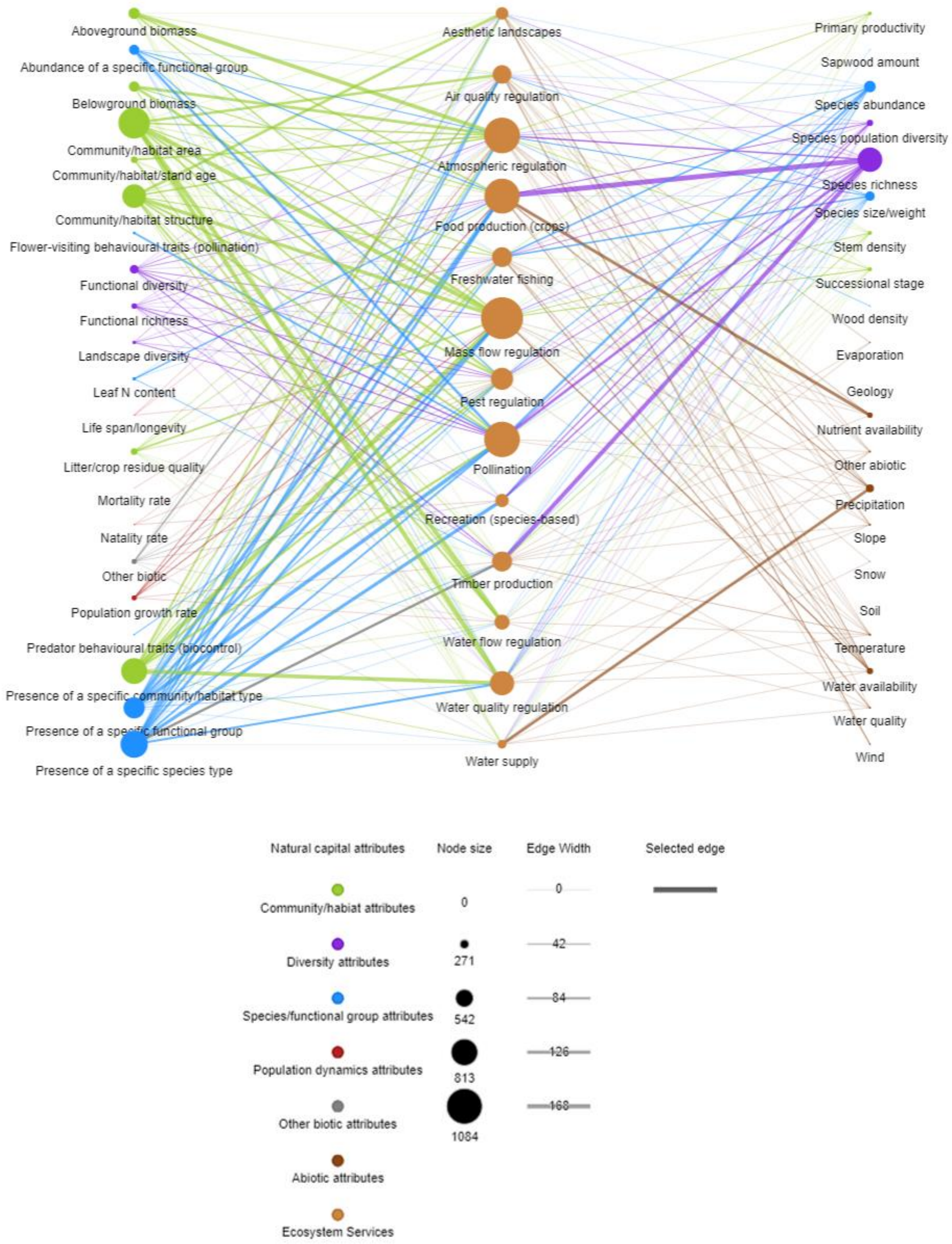


Figure B5: Weighted network diagram to the contexts shown in Figure B1 showing positive linkages between all of the natural capital attributes (nodes on the sides) and all of the ecosystem services (nodes in the middle) available in LiNCAGES. The amount of evidence for a linkage is shown by the thickness of the edge (link between two nodes) and the size of the node is dependent on the number and size of the edges connected to it. In this network diagram the linkage between presence of a specific species has been highlighted for further investigation shown in Table B2.

Table B2: Table of references for all of the studies evidencing positive linkages between the natural capital attribute presence of a specific species type and the ecosystem service of timber production (the selected edge in Figure B4) under the context shown in Figure B1. This table has been ordered by the total weight of the study. Studies with a greater total weight will be more relevant to the users weighted context.

Article title	Weight
Relationships between stand growth and structural diversity in spruce-dominated forests in New Brunswick, Canada	5.20
Productivity, nutrient cycling, and succession in single- and mixrd-species plantations of Casuarina equisetifolia, Eucalyptus robusta, and Leucaena leucocephala in Puerto Rico	5.20
Partitioning the effects of biodiversity and environmental heterogeneity for productivity and mortality in a tropical tree plantation	5.20
Effect of larch (Larix gmelini Rupr.) root exudates on Manchurian walnut (Juglans mandshurica Maxim.) growth and soil juglone in a mixed-species plantation	5.20
Growth of native tree species planted in open pasture, young secondary forest and mature forest in humid tropical Costa Rica	5.20
Productivity of temperate broad-leaved forest stands differing in tree species diversity	3.60
The effect of biodiversity on tree productivity: from temperate to boreal forests	3.60
Vertical structure and basal area development in second-growth Nothofagus stands in Chile	2.80
Comparing productivity of pure and mixed Douglas-fir and western hemlock plantations in the Pacific Northwest	2.80
Are mixed-species stands more productive than single-species stands: an empirical test of three forest types in British Columbia and Alberta	2.80
Comparison of growth and nutrition of young monocultures and mixed stands of Eucalyptus globulus and Acacia mearnsii	2.80
Aboveground productivity of western hemlock and western red cedar mixed-species stands in southern coastal British Columbia	2.80
Comparing productivity of pure and mixed Douglas-fir and western hemlock plantations in the Pacific Northwest	2.80
Pure and mixed forest plantations with native species of the dry tropics of Costa Rica: a comparison of growth and productivity	2.80
Land protection and timber harvesting along productivity and diversity gradients in the Northern Rocky Mountains	2.80
Biodiversity enhances individual performance but does not affect survivorship in tropical trees	2.80
Neighborhood effects and size-asymmetric competition in a tree plantation varying in diversity	2.80
Effects of species richness, identity and environmental variables on growth in planted mangroves in Kenya.	1.20

Table B3: Table of references for all of the studies evidencing positive linkages between the natural capital attribute presence of a specific species type and the ecosystem service of timber production (the selected edge in Figure B4) showing how the users weighting context (Figure B1) influences each of the studies total weight.

Study reference	Temporal scale	Spatial scale	Continent	Observation(s)	Data type	Direction of evidence	Evidence type	Ecosystem	Ecosystem service provider	Management input	Mangement intensity	Mangement impact	Ecosystem service indicator assessment method	Ecosystem services affected	Direction of effect on ecosystem services	Period article published	Total weight
Lei X., War	3	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	5.2
Parrotta J.	3	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	5.2
Healy, C., I	3	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	5.2
Yang, L., P.	3	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	5.2
Piotto, D. (	3	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	5.2
Jacob M. ,L	0.6	2	1	0	0	0	0	0	0	0	0	0	0	0	0	0	3.6
Paquette A	3	0.4	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	3.6
Lusk, C. an	0.6	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	2.8
Amoroso M	0.6	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	2.8
Chen H., K	0.6	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	2.8
Khanna P.	0.6	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	2.8
Chen H., K	0.6	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	2.8
Amoroso M	0.6	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	2.8
Piotto D., V	0.6	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	2.8
Belote R.T	0.6	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	2.8
PotvinC. ar	0.6	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	2.8
POTVIN C.,	0.6	2	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	2.8
Kirui B.,Ka	0.6	0.4	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	1.2

6. Alternative visualisation of Figure 13 and Figure 14 of the manuscript

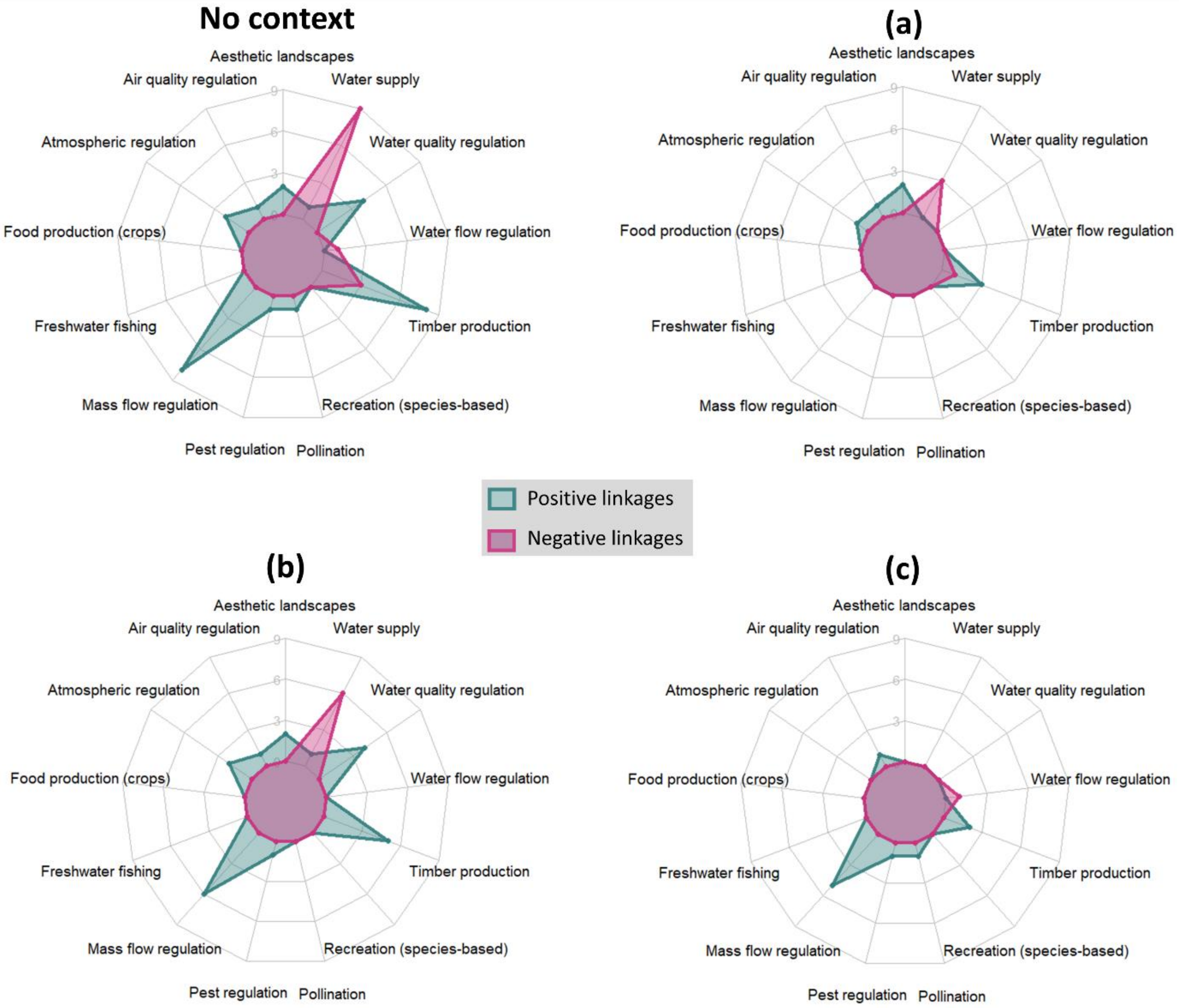


Figure B6: Radar plots showing positive and negative ES-NC linkages with the natural capital attribute of stem density filtered for three different contexts: (a) evidence from studies with spatial scales larger than local, (b) evidence from studies with a snapshot temporal scale and (c) evidence from studies undertaken in Europe. For comparison, an unfiltered radar plot (no context) is also shown.

C. Supplementary material for Chapter 4

Table C1: Criteria required for including natural capital attributes (NCA) from LiNCAGES. Included NCA are shown in bold.

** For this NCA to be included in the analysis the literature evidence for linkages with this NCA has been further interrogated to break it down into NCAs that can be represented by CS2007 data as shown in Table C3.*

NCA	Number of NCA-ES linkages of all directions with all ecosystem services	Is the NCA too broad to be represented by CS2007 data?	Available CS2007 data?	CS2007 data for >50% of the CS squares
Aboveground biomass	90	N	N	
Abundance of a specific functional group	83	Y		
Belowground biomass	71	N	N	
Community/habitat area*	288	Y*	Y	Y
Community/habitat structure	242	Y		
Community/habitat/stand age	79	Y		
Evaporation	57	N	N	
Flower-visiting behavioural traits (pollination)	23	Y		
Functional diversity	71	N	Y	Y
Functional richness	44	N	Y	Y
Geology	22	Y		
Landscape diversity	32	N	Y	Y
Leaf N content	21	N	N	
Life span/longevity	13	Y		
Litter/crop residue quality	57	Y		
Mortality rate	44	Y		
Natality rate	10	Y		
Nutrient availability	50	N	Y	N
Other abiotic	87	Y		
Other biotic	65	Y		
Population growth rate	45	Y		
Precipitation	153	N	N	
Predator behavioural traits (biocontrol)	15	Y		
Presence of a specific community/habitat type	243	Y	N	N
Presence of a specific functional group*	164	Y*	Y	Y
Presence of a specific species	264	Y		

type				
Primary productivity	29	N	Y	Y
Sapwood amount	4	N	N	N
Slope	69	N	Y	Y
Snow	10	N	N	
Soil*	131	Y*	Y	Y
Species abundance	132	Y		
Species population diversity	52	Y		
Species richness	229	N	Y	Y
Species size/weight	104	Y		
Stem density	49	N	N	
Successional stage	36	N	Y	Y
Temperature	112	N	N	
Water availability	61	Y		
Water quality	29	N	Y	Y
Wind	51	N	N	
Wood density	8	N	N	N

Table C2: Showing the breakdown of the NCAs marked with an asterisk in in Table C1

**The breakdown of this NCA was also informed by comments on linkages between the NCAs Flower-visiting behavioural traits (pollination), Abundance of a specific functional group, Predator behavioural traits (biocontrol) and ESs.*

Original NCA	**Number of NCA-ES linkages of all directions with all ES**	**New NCA**	**Number of new NCA-ES links of all directions with all ES**
Presence of a specific functional group*	164	Broadleaf trees	10
		Coniferous trees	12
		Nitrogen fixers	24
		Canopy cover	10
		Pollinators	32
		Birds	4
		herbaceous plants and grass	15
Community/habitat area	288	Grassland habitat area	82
		Heathland and shrub habitat area	34
		Rivers and lakes habitat area	41
		Wetlands habitat area	42
		Woodland and forest habitat area	154
		Cropland habitat area	91
		Sparsely vegetated land habitat area	23
Soil	131	Soil pH (high)	3
		Soil pH (low)	3
		Bulk density	5
		Soil organic matter	8
		Soil carbon concentration	6
		Soil moisture content	13

Table C3: Example of how studies evidencing linkages with the broad NCA of soil were broken down into new NCA through interrogation of the literature reviewer's comment in the OpenNESS literature base. The directions of NCA-ES linkages with the new NCA extracted from the literature reviewer's comment are shown in the table as "p" (positive), "n" (negative), and "u" (unclear).

Service	**Direction**	**Attribute**	**Study ID**	**New NCA**						**Literature reviewer's comment**
				Moisture content	**pH (high)**	**pH (low)**	**Organic matter content**	**Carbon concentration**	**Bulk density**	
Atmospheric regulation	Unclear	Soil	1059							Range of soil types
Atmospheric regulation	Unclear	Soil	1061				p			Above-ground biomass (and hence C storage) is estimated to be half that of a similar Mexican forest on more fertile volcanic soil
Atmospheric regulation	Unclear	Soil	1062		u	u				Slower C sequestration at the Kudu reserve was attributed partly to shallower soil and greater volume of stones. However, high NPP is partly ascribed to favourable soil nutrient and pH levels.
Atmospheric regulation	Unclear	Soil	1064			p		p		Soil pH had a significant impact on SOC: SOC was higher with lower pH for control sites (probably due to inhibition of microbial breakdown of soil C) but this effect was lower for Miscanthus sites.
Water flow regulation	Unclear	Soil	1114				p			Wetland soil had a high porosity (40-80%, max 89%), possibly related to its high organic content, which provided very high water storage capacity.

Table C4: *Table 4 with detailed CS2007 data descriptions*

NCA	**Number of positive and negative linkages**	**CS2007 data description**	**Unit**	**Spatial scale (km^2)**	**Evidence for suitability of representation (references)**
Landscape diversity	26	Shannon index of diversity for habitats surveyed.	-	1	(Nagendra, 2002)
Primary productivity	23	Cover-weighted specific leaf area.	-	0.2	(Maskell et al., 2020)
Slope	34	Slope of the surveyed plot (ordinal 1 = flat to 5 = steep).	Ordinal	0.2	-
Species richness	182	Plant species richness.	Species/area	0.2	(Garnier et al., 2004)
Successional stage	31	The ordination technique detrended correspondence analysis was applied to the species presence absence data from CS2007. The second DCA axis is a measure of disturbance. This axis is interpreted as being most strongly correlated with light availability and in turn correlated with successional stage and disturbance. High scores are associated with later successional, taller vegetation.	-	0.2	(Smart et al., 2003)
Water quality	18	Average Score per Taxon for macroinvertebrates in headwater streams.	-	1	(Dunbar et al., 2010)
Cropland habitat area		Broad habitat area of arable (% cover).	% cover	1	(Smart et al., 2003)*
Grassland habitat area	76	Broad habitat area of acid grassland, improved grassland, neutral grassland and calcareous grassland (% cover).	% cover	1	(Smart et al., 2003)*
Heathland and shrub habitat area	72	Broad habitat area of shrub heath and bracken (% cover).	% cover	1	(Smart et al., 2003)*
Rivers and lakes habitat area	32	Broad habitat area of open water and river (% cover).	% cover	1	(Smart et al., 2003)*
Sparsely vegetated land habitat area	31	Broad habitat area of montane and inland rock (% cover).	% cover	1	(Smart et al., 2003)*
Wetlands habitat area	22	Broad habitat area of bog and fen (% cover).	% cover	1	(Smart et al., 2003)*
Woodland and forest habitat area	38	Broad habitat area of conifer and broadleaf (% cover).	% cover	1	(Smart et al., 2003)*
Birds	137	Abundance of plant species surveyed that are important for the diet of birds.	Abundance	0.2	(Smart et al., 2000)
Broadleaf trees	4	Broadleaf broad habitat area (% cover).	% cover	1	(Smart et al., 2003)*
Canopy cover	10	Abundance of plant species surveyed that are found in shade (Ellenberg L range 5 – 1; semi-shade plant, rarely in full light, but generally with more than 10% relative illumination when trees are in leaf - plant in deep shade).	Ellenberg L score	0.2	(Hill et al., 2000)h
Coniferous trees	10	Broad habitat area of conifer (% cover).	% cover	1	(Smart et al., 2003)*
Herbaceous plants and grass	12	Abundance of plant species surveyed in each plot that are forbs or grasses.	Abundance	0.2	-
Nitrogen fixers	15	Abundance of plant species surveyed in each plot that are nitrogen fixers.	Abundance	0.2	-
Pollinators	24	Abundance of plant species surveyed in each plot that are nectar plants for bees.	Abundance	0.2	(Smart et al., 2010)
Soil organic matter content	32	Loss on ignition on approximately 10g of even dried soil at 375°C for 16 hours.	g/kg	0.2	(Reynolds et al., 2013)
Soil carbon concentration	8	Soil carbon concentration of loss on ignition. Data from elemental analysis demonstrated that 55% of loss on ignition was accounted for by carbon. Soil carbon concentration was subsequently estimated using the equation: C concentration ($gCkg^{-1}$) = LOI (%) x 0.55 x 10.	g/kg	0.2	(Reynolds et al., 2013)
Soil bulk density	6	Bulk density was estimated for each soil sample by recording the exact dimensions of each sampled soil core before the soil was extruded from the plastic tube. On extrusion, the soil was weighed, homogenized, and reweighed before drying at room temperature for up to 2 weeks. At the end of this period, the soil was again weighed and then sieved to 2 mm, and the weight and volume of the unsieved debris recorded. A subsample of 10 g of sieved soil was then accurately weighed and dried overnight at 105°C, before LOI determination. Bulk density was then estimated using the following equation: $\text{Bulk density}\left(\text{gcm}^{-3}\right) = \frac{[\text{oven dry weight of soil (g)} - \text{weight of unsicved debris (g)}]}{\left[\text{volumeofcorecollected}\left(\text{cm}^3\right) - \text{volume of unsieved debris}\left(\text{cm}^3\right)\right]}$	g/cm^3	0.2	(Reynolds et al., 2013)
Soil moisture content	5	The moisture loss at each stage of the process for estimating bulk density was used to estimate the initial moisture content of the soil and from that the initial dry weight of the soil.		0.2	(Reynolds et al., 2013)
Soil pH (high)	13	Extent that soil pH is greater than neutral (7) (soil pH - 7) if soil pH >7. The soil pH was measured from a 0-15cm core of fresh soil from each plot.	-	0.2	(Reynolds et al., 2013)
Soil pH (low)	3	Extent that soil pH is less than neutral (7) (7 - soil pH) if soil pH <7. The soil pH was measured from a 0-15cm core of fresh soil from each plot.	-	0.2	(Reynolds et al., 2013)

** The Broad Habitat classification for Britain provides a framework of 22 generalised land cover types designed to allow the whole land surface to be mapped. During the field survey the total land area within each 1 km square was mapped to Broad Habitat (Smart et al., 2003).*

__Box C1:__ Method for how the random number dependence of the k-means algorithm in generating cluster centres was accounted for.

We used the k-means algorithm to cluster the NCAs at 1km^2 scale into seven clusters for 1000 unique seeds and recorded how often the different NCA cluster together using a matrix. This matrix revealed that the clustering shown in Table 2 is observed for over 50% of the seeds. A seed that gives this clustering at 1km^2 scale ignoring context was used for all subsequent clustering under the different contexts.

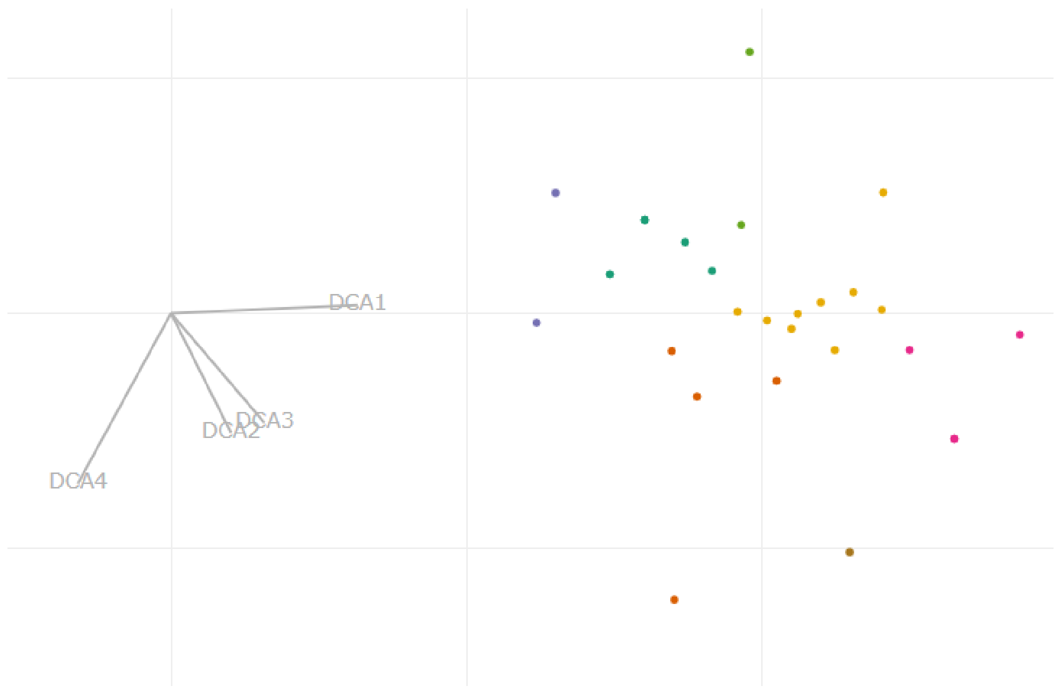


__Figure C1:__ Screenshot of an interactive Grand Tour (Cook et al., 1995) for exploring the clustering of the NCAs at 1km^2 scale. See https://glinney.shinyapps.io/NCA_interlinkages_and_ES/ to interact with this Grand tour.

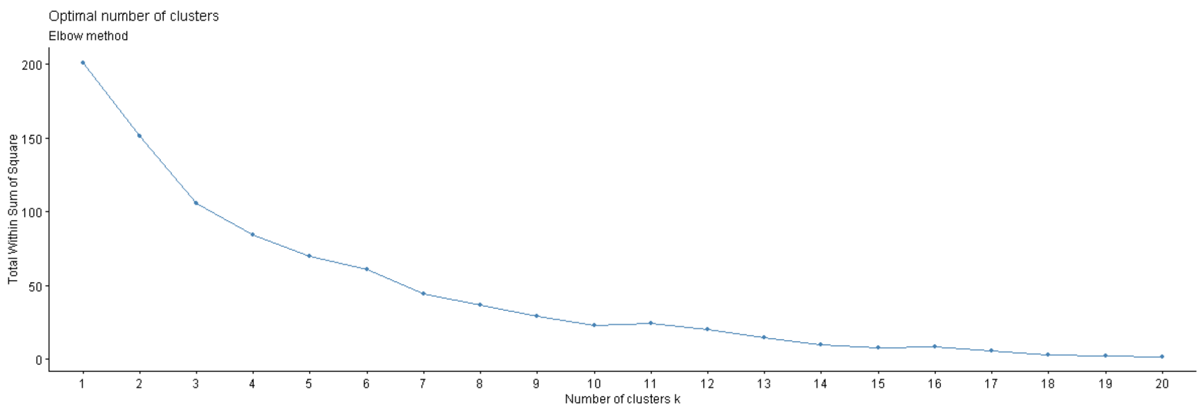


__Figure C2:__ Elbow plot showing how the total within cluster sum of squares changes with number of clusters.

Box C2: Shiny app used for assessing number of clusters and creating the visualisations

A shiny app for generating visualisations to assess the clustering of the 26 NCAs included in this study. The shiny app also generates many of the visualisations shown in this paper. These visualisations include: grand tour (interactive) (Figure C1), elbow plot (Figure C2), table of clusters of NCAs and all radar plots shown in this paper.
The shiny app allows a user to select the NCAs to be included in the clustering and the spatial scale and location context of the data that is used to represent the NCAs.
The shiny app can be accessed here: https://glinney.shinyapps.io/NCA_interlinkages_and_ES/

Table C5: *Grouping CS2007 squares by spatial scale and location context*

<table>
<tr><th>Context</th><th>Method</th></tr>
<tr><td>b</td><td rowspan="2">1-km^2 squares grouped to upland (b) and lowland (c) by their environmental zones, which are an amalgamation of land classes. See (ref) for how these environmental zones were derived. Zones 1, 2, 4 and 8 were classified as lowland, while zones 3, 5, 6 and 9 were classified as upland.</td></tr>
<tr><td>c</td></tr>
<tr><td>D</td><td>Spatial overlay of any 1-km^2 square that is included or partially included in the NNR/SSSI for England, Scotland and Wales.
This was created by merging the shapefiles for NNR and SSSI locations for England, Scotland and Wales. The “select by location” function in ArcMap was used to select the CS2007 sites from a shape file of the CS2007 site locations (target layer) that “intersects (3d) the source layer feature”, where the merged SSSI and NNR shape files were the source layer.</td></tr>
</table>

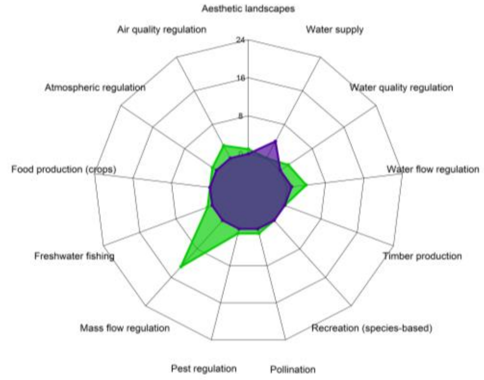
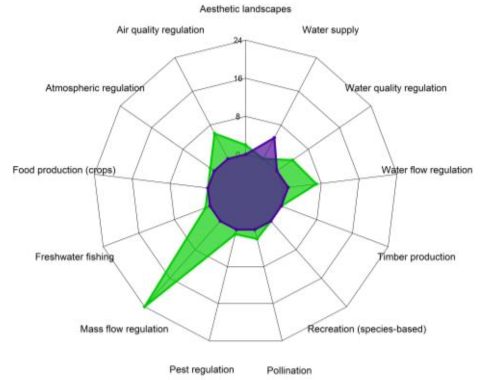
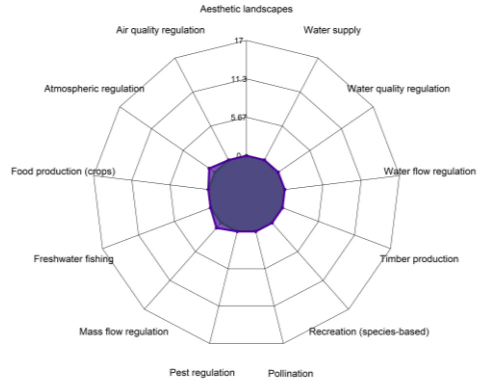
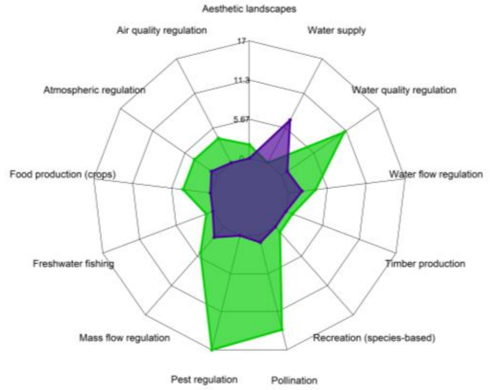
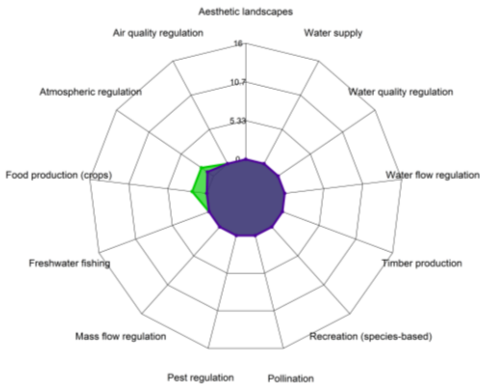
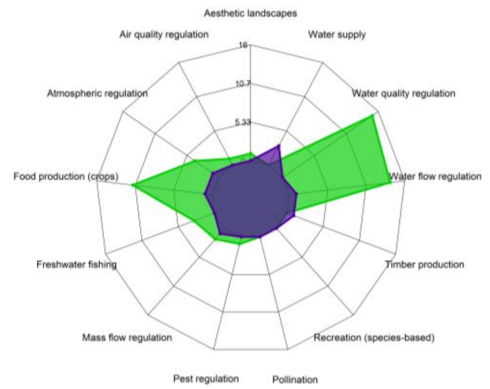
NCA	Linkages between the NCA of interest and all ESs	Linkages between all NCAs clustered with the NCA of interest and all ESs
Heathland and shrub habitat area		
Soil pH (high)		
Landscape diversity		
Soil carbon conc.		

Figure C3: Radar plots comparing the evidence for NCA-ES linkages with just the natural capital of interest to the NCA-ES linkages for all of the NCAs that share a cluster with the natural capital of interest. The direction of the NCA-ES linkages is coloured in the plot: positive (green) and negative (purple). The radar plots are scaled consistently for each NCA.

Table C6: NCA clusters containing landscape diversity under each context

NCAs clustered with landscape diversity under context:			
(a) Ignoring context dependency	**(b) Upland**	**(c) Lowland**	**(d) SSSI/NNR**
Landscape diversity	Landscape diversity	Landscape diversity	Landscape diversity
Primary productivity	Primary productivity	Primary productivity	Primary productivity
Slope	Slope	Slope	Slope
Successional stage	Successional stage	Successional stage	Successional stage
Species richness	Species richness	Soil bulk density	Species richness
Grassland habitat area	Soil bulk density	Cropland habitat area	Soil bulk density
Herbaceous plants and grass	Cropland habitat area	Soil pH (high)	Grassland habitat area
Birds	Pollinators		Herbaceous plants and grass
Nitrogen fixers	Broadleaf trees		Pollinators
	Canopy cover		Birds
			Nitrogen fixers

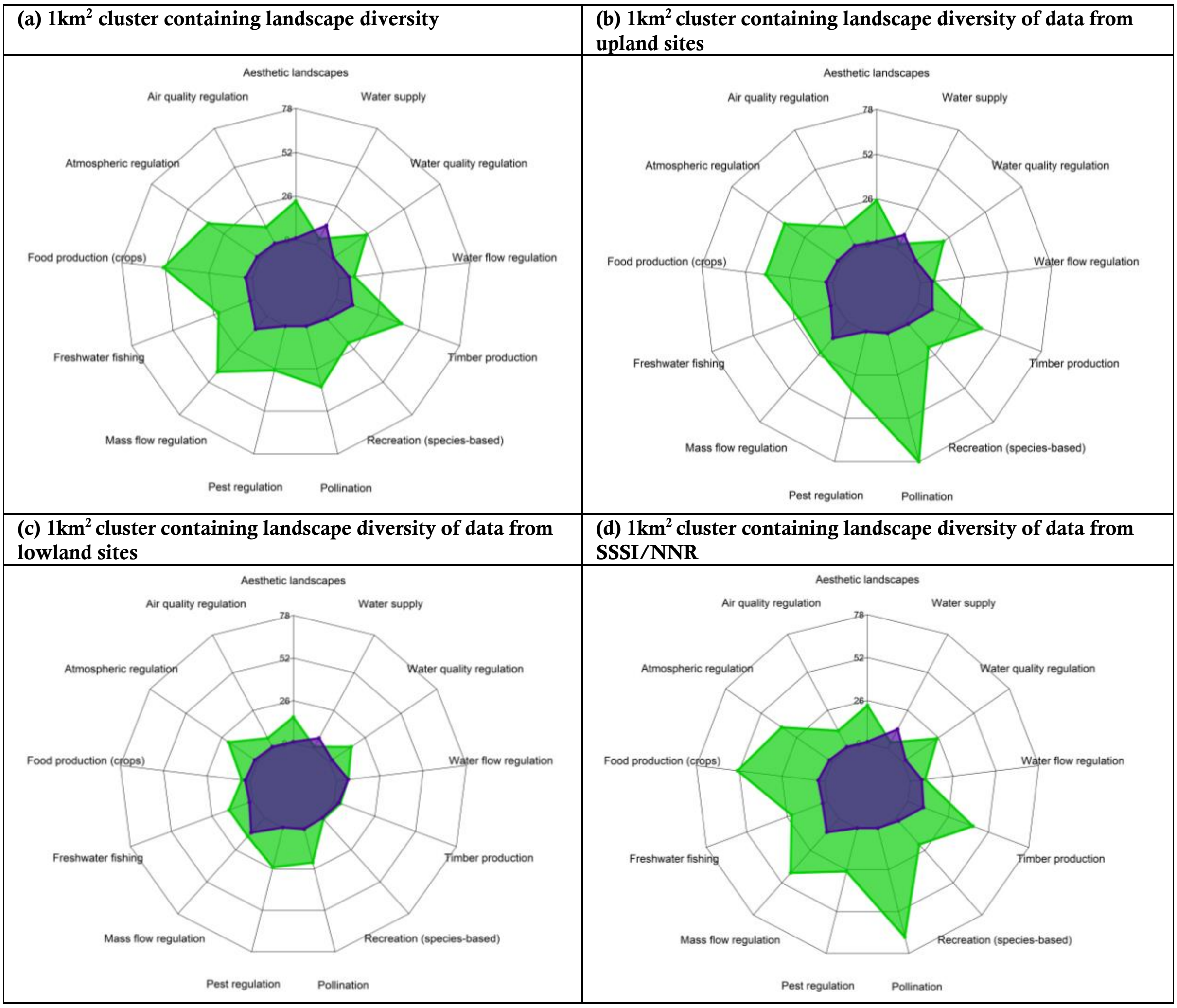


Figure C4: *Radar plots showing how the evidence for NCA-ES linkages changes for each cluster containing the NCA of landscape diversity under the three location scenarios. The direction of the NCA-ES linkages if coloured in the plot: positive (green) and negative (purple). The radar plots are scaled consistently.*

Box C3: *Description of the location context dependencies for NCA-NCA interlinkages when following the NCA of landscape diversity.*

We assess how changing location context influences the clustering of the NCAs with landscape diversity (Table C6). Landscape diversity is clustered with broadleaf trees, cropland habitat area, and species richness in the uplands. However, these interlinkages with species richness could be at the expense of low nutrient habitat specialists. Landscape diversity is clustered with a more heterogeneous landscape at the marginal uplands, which is a possible explanation for the wide variety of NCAs clustered with landscape diversity under upland context. Landscape diversity loses NCA-NCA interlinkages with canopy cover, broadleaf trees, pollinates and species richness at lowland context, but also gains NCA-NCA interlinkages with soil pH (high). This could be due to the fragmentation of habitats in the lowlands. Context (d) shows very similar NCA-NCA interlinkages to context (a) with the addition of NCA-NCA interlinkages with pollinators and soil bulk density. Pollinators are typically more prevalent in SSSIs and NNRs.

Figure C4 shows how the effect of location context on the NCA-NCA interlinkages in turn influences the NCA-ES linkages identified for the NCA of landscape diversity. At upland context (context (b)) we observe more positive NCA-ES linkages with pollination due to NCA-NCA interlinkages with pollinators and more positive NCA-ES linkages with atmospheric regulation due to NCA-NCA interlinkages with canopy cover and broadleaf trees. At lowland context (context (c)) we lose positive linkages with recreation (species-based) due to the exclusion of NCA-NCA interlinkages with species richness. SSSI/NNR context (context (d)) features new positive NCA-ES linkages with pollination due to the NCA-NCA interlinkages with pollinators under this context.

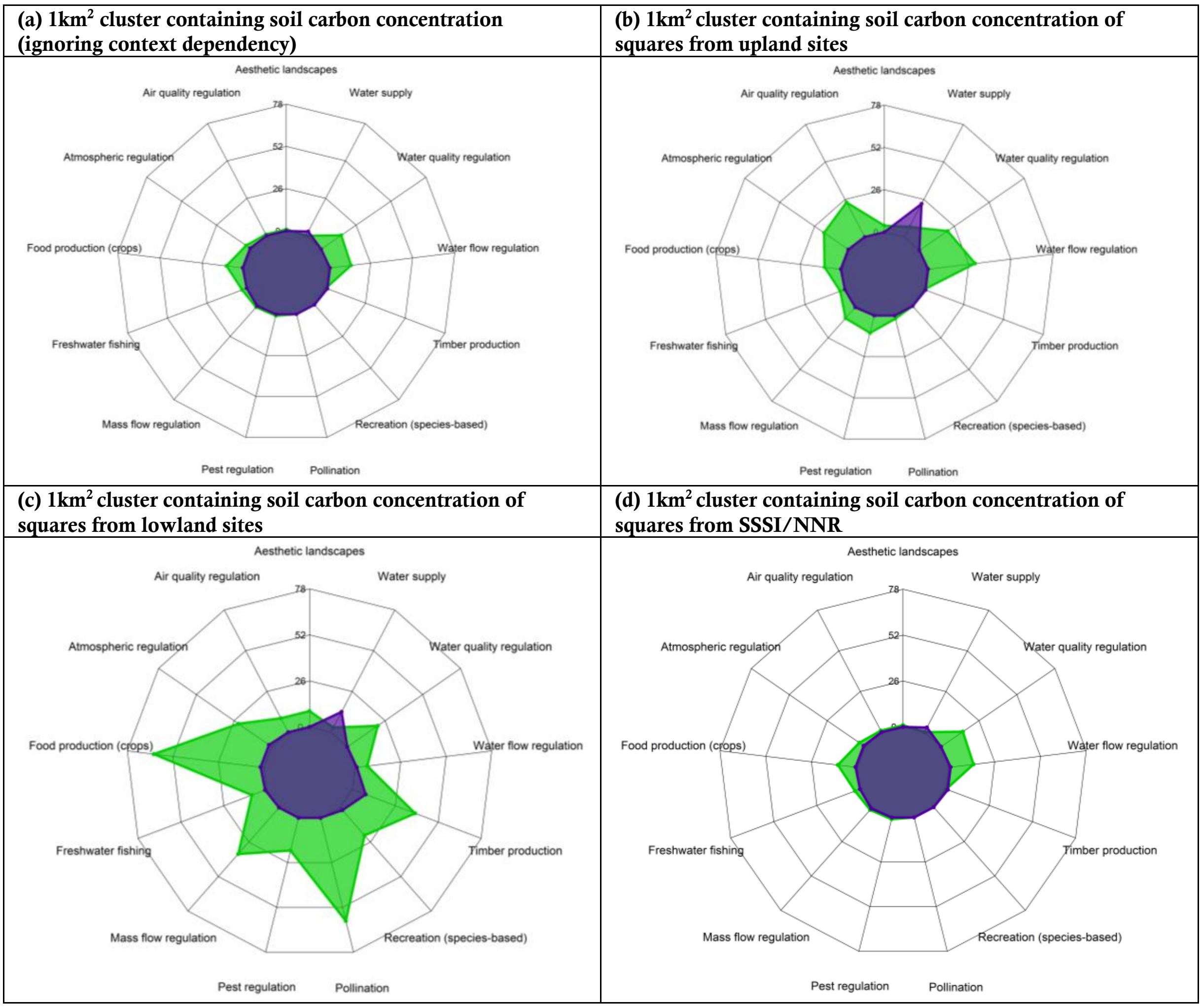


__Figure C5.__ Radar plots showing how the evidence for NCA-ES linkages changes for each cluster containing the NCA of soil carbon concentration under the three location contexts. The direction of the NCA-ES linkages is coloured in the plot: positive (green) and negative (purple). The radar plots are scaled consistently for each NCA.

Table C7: Clustering of the NCAs under each of the spatial scale and location contexts investigated.

Ignoring context		Upland		Lowland		SSSI/NNR	
1	Soil carbon concentration	1	Heathland and shrub habitat area	1	Sparsely vegetated land habitat area	1	Soil carbon concentration
	Soil moisture content		Wetlands habitat area	2	Water quality		Soil moisture content
	Soil organic matter content	2	Birds	3	Pollinators		Soil organic matter content
	Soil pH (low)		Grassland habitat area		Birds		Soil pH (low)
	Wetlands habitat area		Herbaceous plants and grass		Grassland habitat area		Wetlands habitat area
2	Birds		Nitrogen fixers		Herbaceous plants and grass	2	Cropland habitat area
	Grassland habitat area		Soil pH (high)		Nitrogen fixers		Soil pH (high)
	Herbaceous plants and grass	3	Coniferous trees		Soil carbon concentration		Water quality
	Nitrogen fixers		Soil carbon concentration		Soil moisture content	3	Coniferous trees
	Landscape diversity		Soil moisture content		Soil organic matter content	4	Broadleaf trees
	Primary productivity		Soil organic matter content		Species richness		Woodland and forest habitat area
	Slope		Soil pH (low)	4	Broadleaf trees	5	Canopy cover
	Successional stage		Woodland and forest habitat area		Coniferous trees		Rivers and lakes habitat area
	Species richness	4	Water quality		Canopy cover	6	Heathland and shrub habitat area
3	Coniferous trees	5	Rivers and lakes habitat area		Soil pH (low)		Sparsely vegetated land habitat area
	Woodland and forest habitat area	6	Sparsely vegetated land habitat area		Woodland and forest habitat area	7	Pollinators
4	Pollinators	7	Pollinators	5	Rivers and lakes habitat area		Birds
	Broadleaf trees		Broadleaf trees	6	Heathland and shrub habitat area		Soil bulk density
	Canopy cover		Soil bulk density		Wetlands habitat area		Grassland habitat area
	Rivers and lakes habitat area		Cropland habitat area	7	Soil bulk density		Herbaceous plants and grass
5	Water quality		Canopy cover		Cropland habitat area		Nitrogen fixers
6	Heathland and shrub habitat area		Landscape diversity		Landscape diversity		Landscape diversity
	Sparsely vegetated land habitat area		Primary productivity		Primary productivity		Primary productivity
7	Soil bulk density		Slope		Slope		Slope
	Cropland habitat area		Successional stage		Soil pH (high)		Successional stage
	Soil pH (high)		Species richness		Successional stage		Species richness

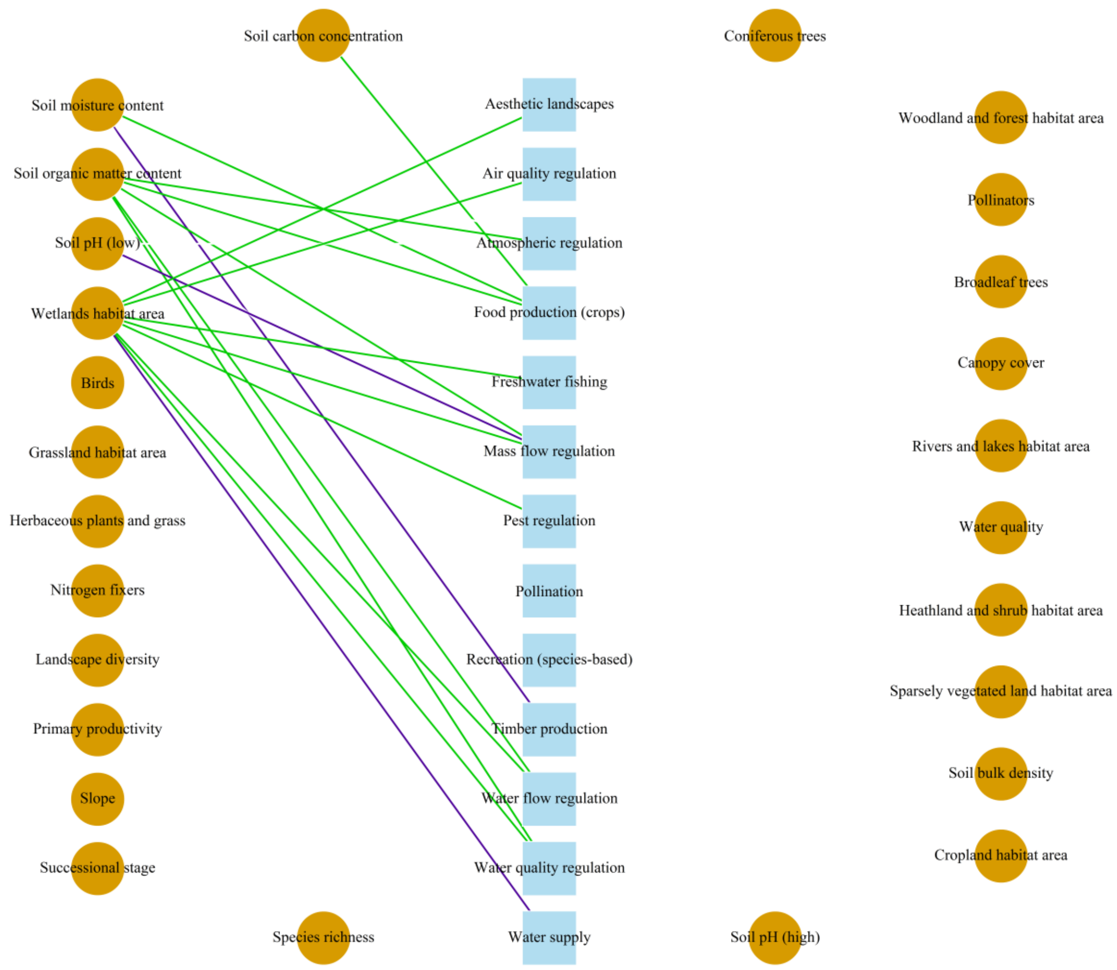


Figure C6a: Hypothetical schematic from Figure 1 populated with evidence for NCA-ES linkages with the NCA in cluster 1 (wet soil). NCA nodes are orange circles and ES nodes are blue squares. Edges show only the presence of a NCA-ES linkage and not the number of studies evidencing it. The direction of the NCA-ES linkages is coloured: positive (green) and negative (purple). Only NCA-ES linkages that have over 50% of their linkages of positive or negative direction are shown.

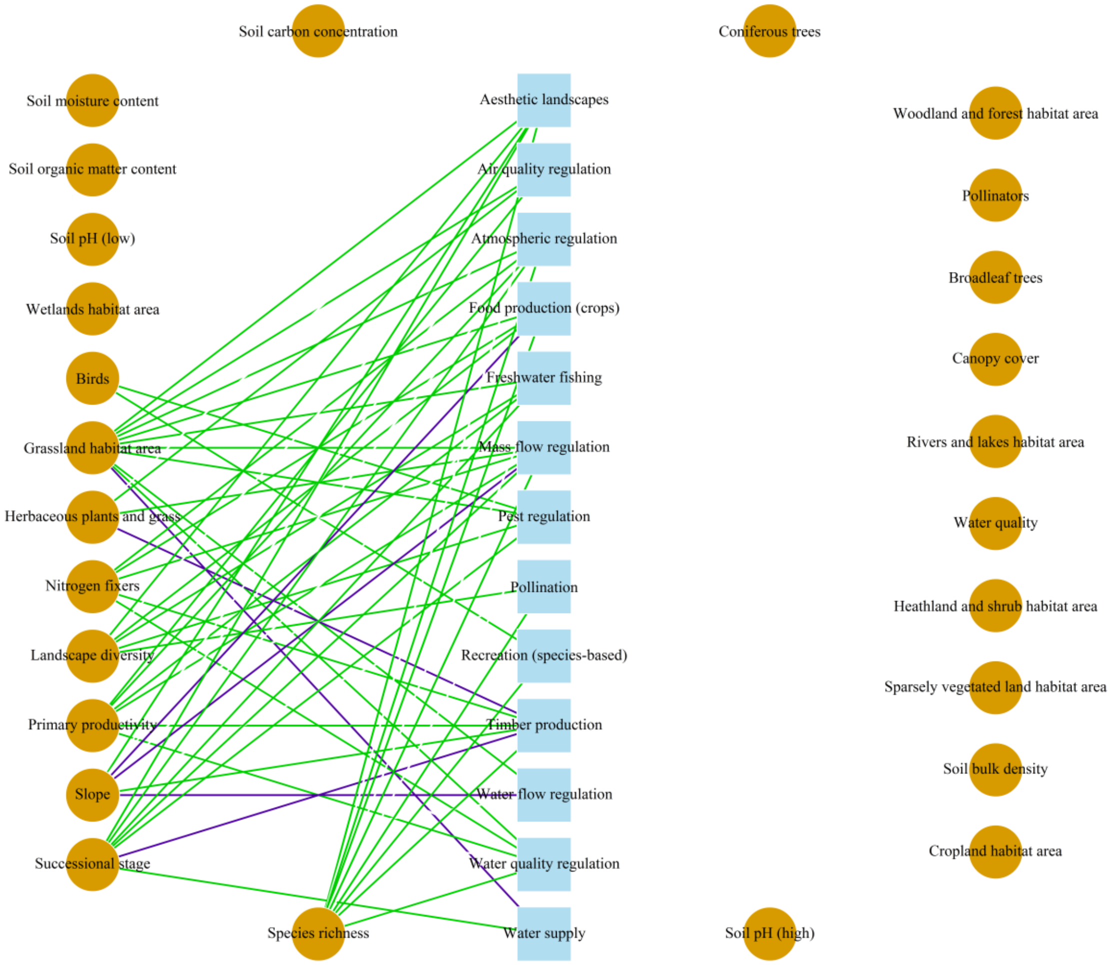


Figure C6b: *Hypothetical schematic from Figure 1 populated with evidence for NCA-ES linkages with the NCA in cluster 2 (diversity) and all ES included within this study. NCA nodes are orange circles and ES nodes are blue squares. Edges show only the presence of a NCA-ES linkage and not the number of studies evidencing it. The direction of the NCA-ES linkages is coloured: positive (green) and negative (purple). Only NCA-ES linkages that have over 50% of their linkages of positive or negative direction are shown.*

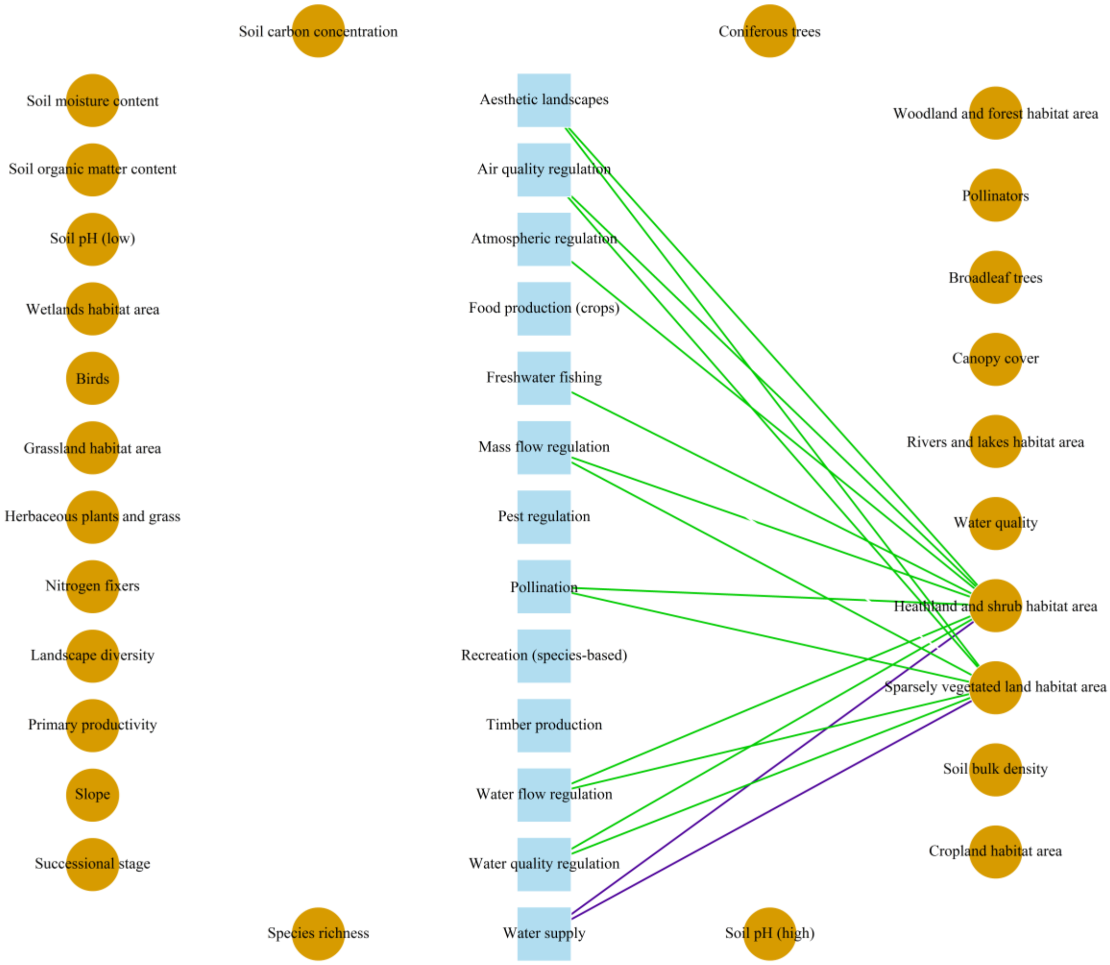


Figure C6c: *Hypothetical schematic from Figure 1 populated with evidence for NCA-ES linkages with the NCA in cluster 3 (upland) and all ES included within this study. NCA nodes are orange circles and ES nodes are blue squares. Edges show only the presence of a NCA-ES linkage and not the number of studies evidencing it. The direction of the NCA-ES linkages is coloured: positive (green) and negative (purple). Only NCA-ES linkages that have over 50% of their linkages of positive or negative direction are shown.*

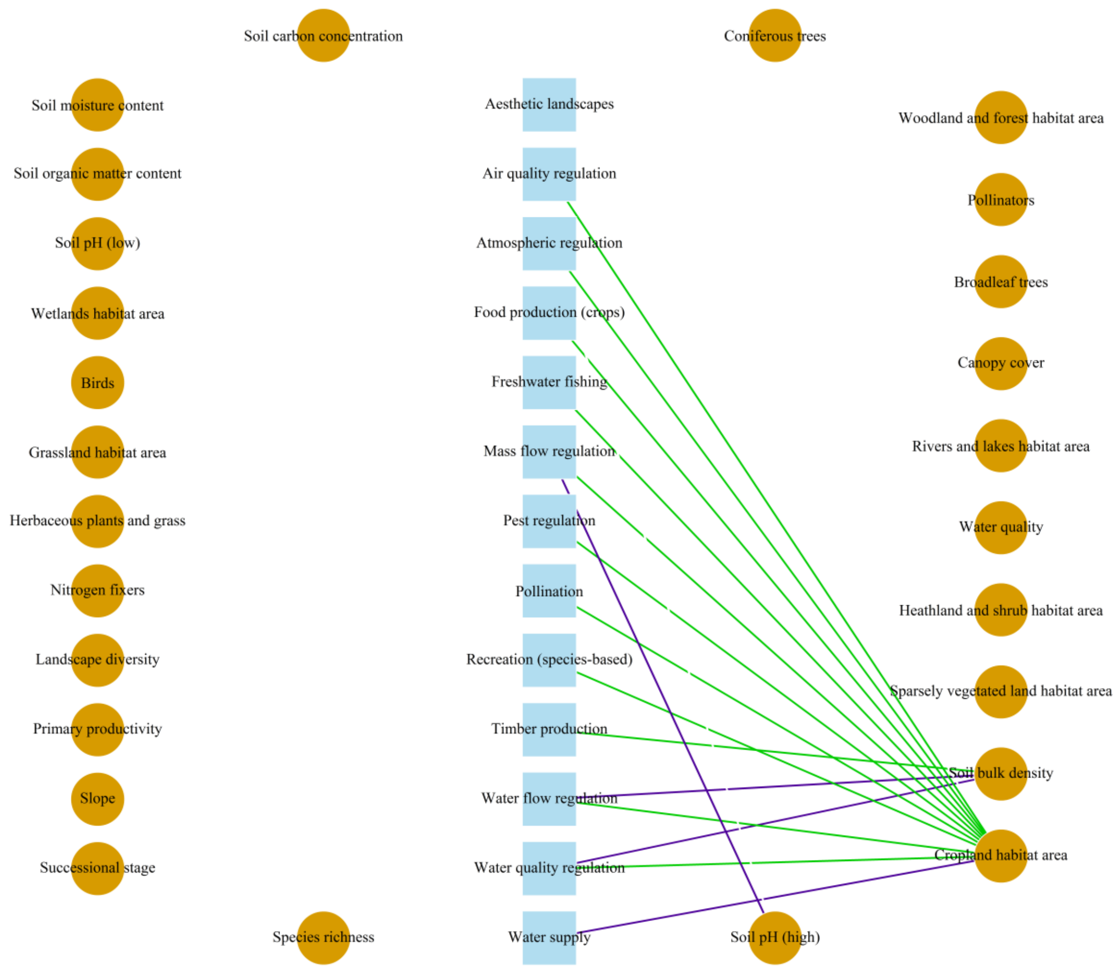


Figure C6d: *Hypothetical schematic from Figure 1 populated with evidence for NCA-ES linkages with the NCA in cluster 4 (agricultural production) and all ES included within this study. NCA nodes are orange circles and ES nodes are blue squares. Edges show only the presence of a NCA-ES linkage and not the number of studies evidencing it. The direction of the NCA-ES linkages is coloured: positive (green) and negative (purple). Only NCA-ES linkages that have over 50% of their linkages of positive or negative direction are shown.*

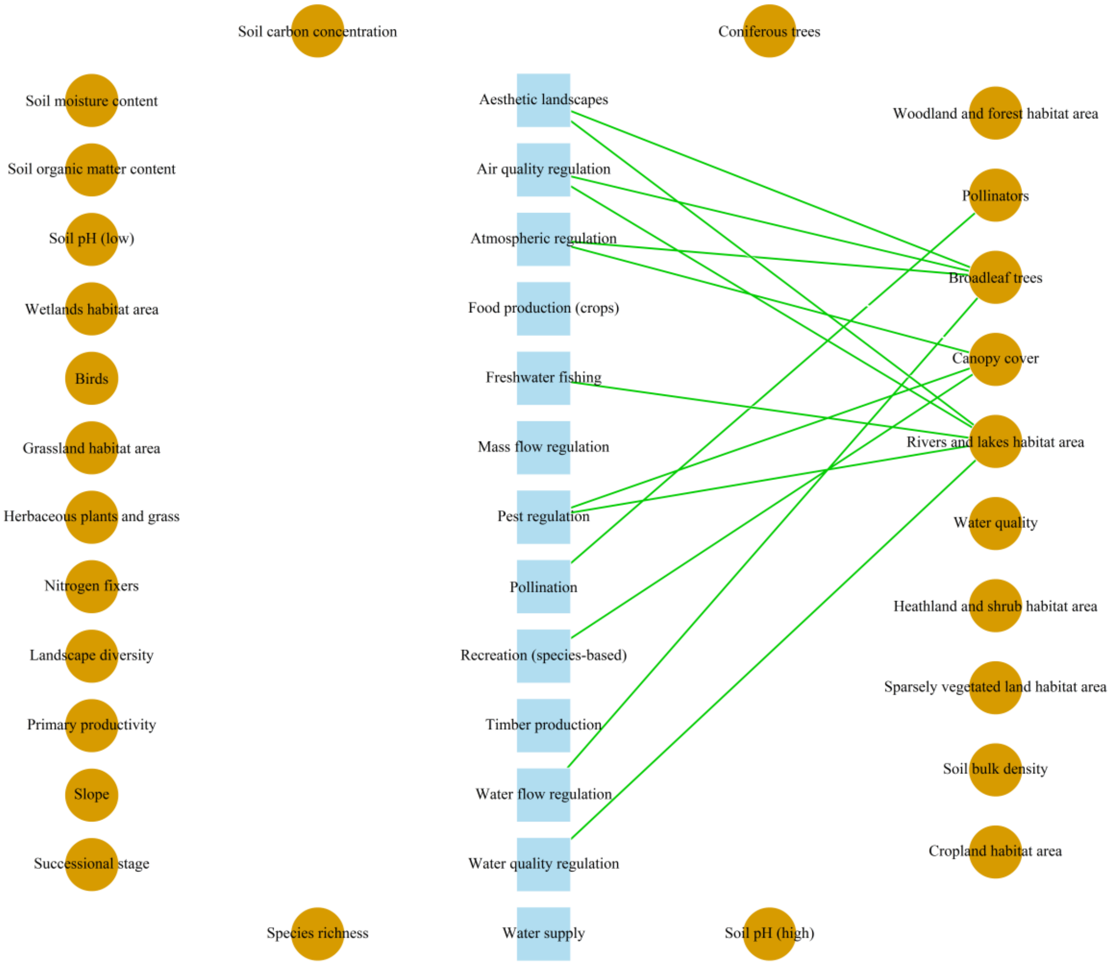


Figure C6e: *Hypothetical schematic from Figure 1 populated with evidence for NCA-ES linkages with the NCA in cluster 5 (woodlands and forest) and all ES included within this study. NCA nodes are orange circles and ES nodes are blue squares. Edges show only the presence of a NCA-ES linkage and not the number of studies evidencing it. The direction of the NCA-ES linkages is coloured: positive (green) and negative (purple). Only NCA-ES linkages that have over 50% of their linkages of positive or negative direction are shown.*

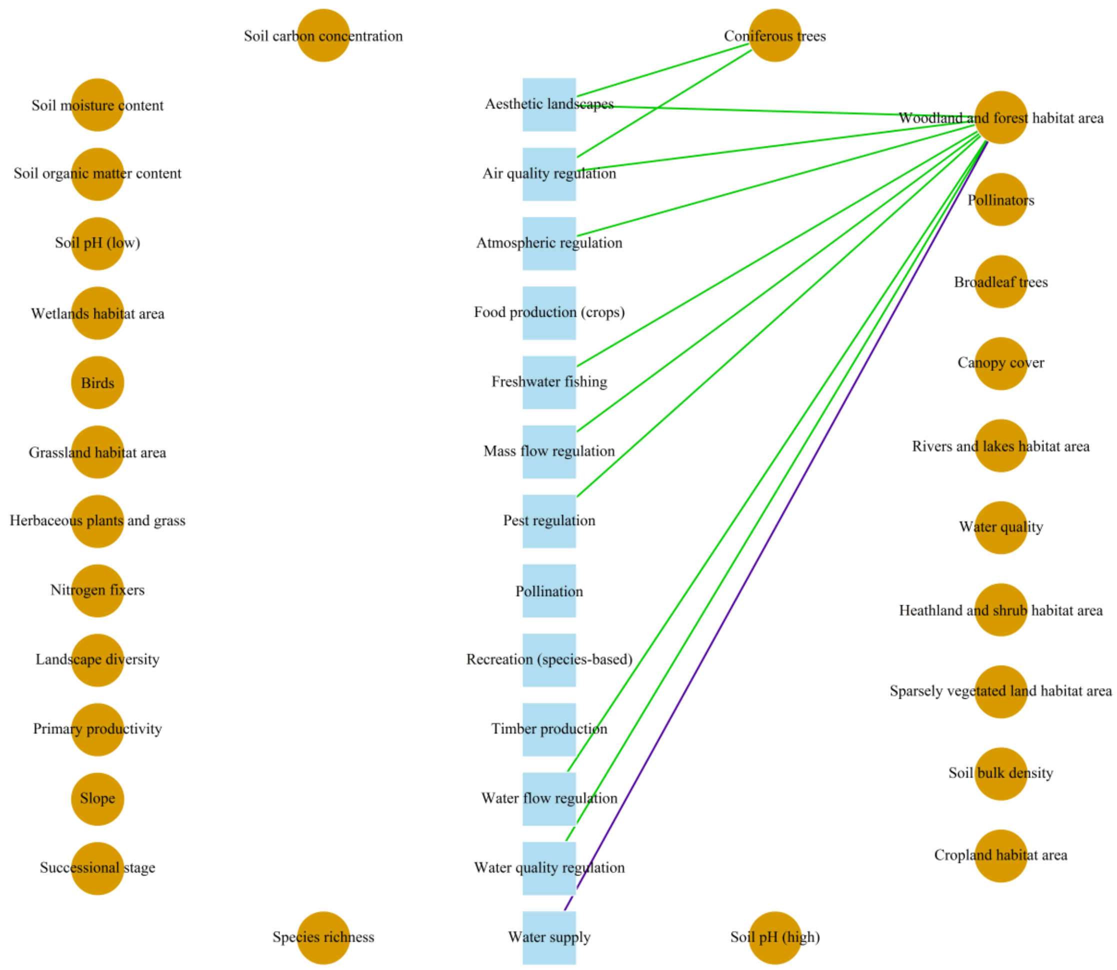


Figure C6f: *Hypothetical schematic from Figure 1 populated with evidence for NCA-ES linkages with the NCA in cluster 6 (coniferous woodlands) and all ES included within this study. NCA nodes are orange circles and ES nodes are blue squares. Edges show only the presence of a NCA-ES linkage and not the number of studies evidencing it. The direction of the NCA-ES linkages is coloured: positive (green) and negative (purple). Only NCA-ES linkages that have over 50% of their linkages of positive or negative direction are shown.*

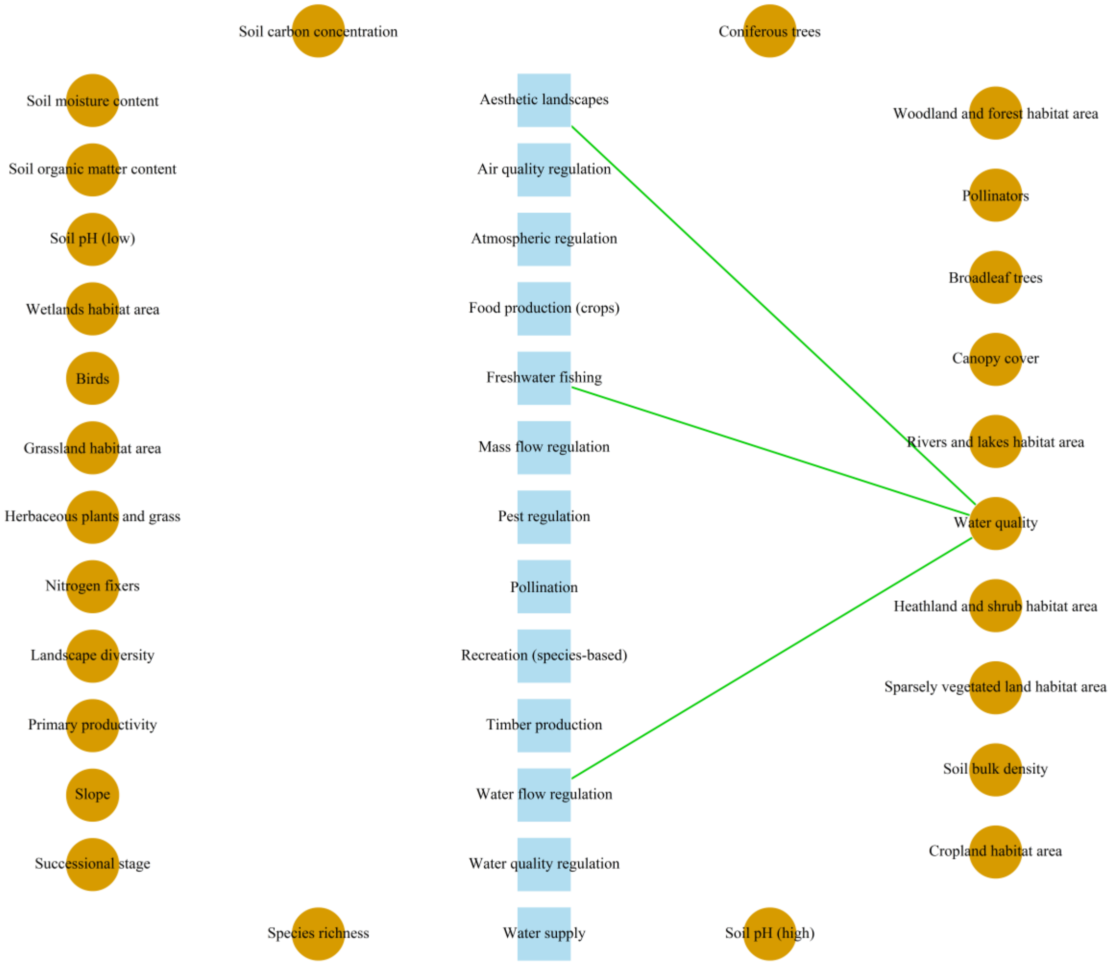


Figure C6g: Hypothetical schematic from Figure 1 populated with evidence for NCA-ES linkages with the NCA in cluster 7 (water quality) and all ES included within this study. NCA nodes are orange circles and ES nodes are blue squares. Edges show only the presence of a NCA-ES linkage and not the number of studies evidencing it. The direction of the NCA-ES linkages is coloured: positive (green) and negative (purple). Only NCA-ES linkages that have over 50% of their linkages of positive or negative direction are shown.

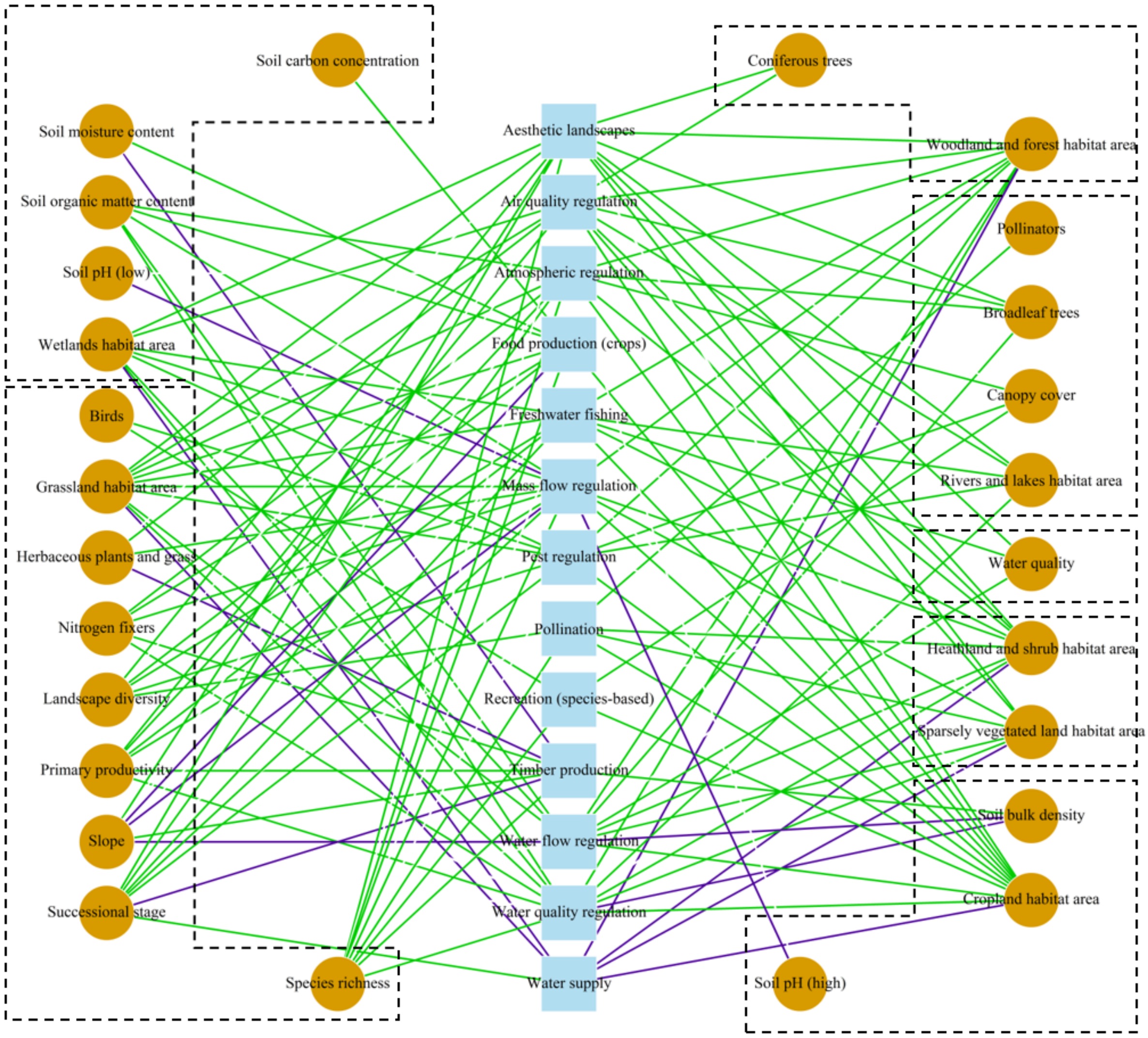


Figure C6h: *Hypothetical schematic from Figure 1 populated with evidence for NCA-NCA interlinkages and NCA-ES linkages with all NCA and ES included within this study. NCA nodes are orange circles and ES nodes are blue squares. Edges show only the presence of a NCA-ES linkage and not the number of studies evidencing it. The direction of the NCA-ES linkages is coloured: positive (green) and negative (purple). Only NCA-ES linkages that have over 50% of their linkages of positive or negative direction are shown. NCA that are interlinked (clustered together) are encompassed by dashed lines.*

D. Supplementary material for Chapter 5

1. Data sources

Table D1: Detailed version of Table 7 including the different definitions used by each data source.

Evidence type	**Ecosystem service**	**Data source**				
		Approach	**Indicator name and definition**	**Spatial resolution**	**Units**	**Reference**
Empirical	Timber production	Eurostat data (Eurostat, 2020)	Volume of timber 2020	Country	Thousands of tonnes	(Eurostat, 2020)
	Carbon sequestration	Global Forest Resources Assessment data (FAO, 2020a); see FAO (2020b) for the report.	Carbon in aboveground biomass 2020	Country	Tonnes per hectare	Data (FAO, 2020a) Report (FAO, 2020b).
	Aesthetic landscapes	NA*	NA	NA	NA	NA
Modelled	Timber production	The IMPRESSIONS Integrated Assessment Platform 2 (IAP2) is a suite of ten sectoral meta models which outputs eight land uses and various metrics that can represent ecosystem services.	Timber production Mt	10` × 10`	Mt	(Harrison et al., 2013)
	Carbon sequestration		Potential carbon stock, *defined as Mt of carbon stored in forests* (Harrison et al., 2019)		t/ha	
	Aesthetic landscapes		Land use diversity *calculated at the grid-cell level as the Shannon index of six major land use classes (forestry, arable, intensive grassland, extensive grassland, unmanaged land and urban)* (Dunford et al., 2015).		Score 0 - 1	
Expert	Timber production	Matrix linking land cover to 31 ecosystem services using expert derived scores Burkhard et al. (2014).	Timber, *defined as presence of trees or plants with potential use for timber (Burkhard et al, 2012)*	Spatially independent**	Score 0 – 5 of how well a land use supplies a service (0 = low and 5 high)	(Burkhard *et al*, 2014)
	Carbon sequestration		Global climate regulation *defined as, supply and demand of CO2 sequestration in t CO2/ha per year* (Burkhard et al., 2012)			
	Aesthetic landscapes		Recreation & aesthetic values *refers specifically to landscape and visual qualities of the resp. case study area (scenery, scenic beauty). The benefit is the sense of beauty people get from looking at the landscape* (Burkhard et al., 2012)			
Literature	Timber production	LiNCAGES platform. An interactive platform linking natural capital attributes to 13 ecosystem services using literature synthesis using a vote counting approach.	Timber production, *defined as Fibre (timber and wood fuel) (MA), Materials/biomass (timber) (CICES)* (Pérez Soba et al., 2017)	Spatially independent**	Number of studies evidencing a positive linkage with the ecosystem service.	(Linney et al., 2020)
	Carbon sequestration		Atmospheric regulation, *defined as the maintenance of physical, chemical, biological conditions/atmospheric composition and climate regulation* (Harrison et al., 2014)			
	Aesthetic landscapes		Aesthetic landscapes, *defined as aesthetic values (MA) Intellectual and representational interactions (CICES)* (Pérez Soba et al., 2017)			

*no data found for aesthetic landscapes that can represent empirical evidence to the same extent as the other two services.

**Spatially independent, requires a land cover or land use map as input data to create an ESP map. Resulting ecosystem service provision map will be same spatial resolution as the input data.

2. Expert and literature land cover/ecosystem to ecosystem service matrices

Table D2: matrix linking CORINE land cover classes to ecosystem service provision using expert and literature evidence. Expert evidence is an expert opinion score between 0 and 5, where 0 is no relevant potential and 5 is high relevant potential. See Stoll et al. (2015) *for the full matrix. Literature evidence is the number of positive natural capital to ecosystem service linkages identified by studies completed in each ecosystem. See Table D4 for the literature matrix including all the services available in LiNCAGES.*

CORINE Land cover class	**Expert (score 0-5)**			**LiNCAGES ecosystem**	**Literature (number of studies)**		
	Timber	**Global climate regulation**	**Landscape aesthetics & Inspiration**		**Timber production**	**Atmospheric regulation**	**Aesthetic landscapes**
Continuous urban fabric	0	0	3	Urban	0	0	13
Discontinuous urban fabric	0	0	2	Urban	0	0	13
Industrial or commercial units	0	0	1	Urban	0	0	13
Road and rail networks	0	0	0	Urban	0	0	13
Port areas	0	0	2	Urban	0	0	13
Airports	0	0	0	Urban	0	0	13
Mineral extraction sites	0	0	0	Urban	0	0	13
Dump sites	0	0	0	Urban	0	0	13
Construction sites	0	0	0	Urban	0	0	13
Green urban areas	0	1	3	Urban	0	0	13
Sport and leisure facilities	0	1	1	Urban	0	0	13
Non-irrigated arable land	0	1	1	Cropland	0	0	0
Permanently irrigated land	0	1	1	Cropland	0	0	0
Ricefields	0	0	1	Cropland	0	0	0
Vineyards	0	1	2	Cropland	0	0	0
Fruit trees and berries	4	2	2	Cropland	0	0	0
Olive groves	4	1	2	Cropland	0	0	0
Pastures	0	1	2	Grassland	0	31	4
Annual and permanent crops	0	1	1	Cropland	0	0	0
Complex cultivation patterns	0	1	2	Cropland	0	0	0
Agriculture & Natural vegetation	3	2	2	Cropland	0	0	0
Agro-forestry areas	3	1	2	No eco	0	0	0
Broad-leaved forest	5	4	5	Woodland and forest	89	79	16
Coniferous forest	5	4	5	Woodland and forest	89	79	16
Mixed forest	5	4	5	Woodland and forest	89	79	16
Natural grassland	0	3	4	Grassland	0	31	4
Moors and heathland	0	3	4	Upland	0	0	0
Sclerophyllous vegetation	0	1	3	Upland	0	0	0
Transitional woodland shrub	0	0	3	Woodland and forest	89	79	16
Beaches, dunes and sand plains	0	0	4	Coastal	0	0	4

Bare rock	0	0	3	No eco	0	0	0
Sparsely vegetated areas	0	0	1	Upland	0	8	0
Burnt areas	0	0	0	No eco	0	0	0
Glaciers and perpetual snow	0	3	5	No eco	0	0	0
Inland marshes	0	2	2	Wetlands	0	0	0
Peatbogs	0	5	2	Upland	0	8	0
Salt marshes	0	0	2	Marine inlets and transitional waters	0	0	2
Salines	0	0	2	Marine inlets and transitional waters	0	0	2
Intertidal flats	0	0	2	Marine inlets and transitional waters	0	0	2
Water courses	0	0	4	Rivers and lakes	0	0	0
Water bodies	0	1	4	Rivers and lakes	0	0	0
Coastal lagoons	0	0	4	Marine inlets and transitional waters	0	0	2
Estuaries	0	0	5	Marine inlets and transitional waters	0	0	2
Sea and ocean	0	5	5	No eco	0	0	0

Table D3: Matrix linking the IAP2 land use allocations that have been assigned to CORINE land cover classes (by the method outlined in Box D2) to ecosystem service provision using expert and literature evidence. Expert evidence is an expert opinion score between 0 and 5, where 0 is no relevant potential and 5 is high relevant potential. See Burkhard et al. (2014) for the full matrix. Literature evidence is the number of positive natural capital to ecosystem service linkages identified by studies completed in each ecosystem. See Table D4 for the literature matrix including all of the services available in LiNCAGES.

CORINE Land class	**Expert (score 0-5)**			**LiNCAGES ecosystem**	**Literature (number of studies)**		
	Timber	**Global climate regulation**	**Landscape aesthetics & Inspiration**		**Timber production**	**Atmospheric regulation**	**Aesthetic landscapes**
Continuous urban fabric	0	0	3	Urban	0	0	13
Discontinuous urban fabric	0	0	2	Urban	0	0	13
Permanently irrigated land	0	1	1	Cropland	0	0	0
Non-irrigated arable land	0	1	1	Cropland	0	0	0
Pastures	0	1	2	Grassland	0	31	4
Broad-leaved forest and mixed forest*	5	4	5	Woodland and forest	89	79	16
Transitional woodland shrub	0	0	3	Woodland and forest	89	79	16
Salt marshes	0	0	2	Marine inlets and transitional waters	0	0	2
Intertidal flats	0	0	2	Marine inlets and transitional waters	0	0	2
Inland marshes	0	2	2	Wetlands	0	0	0
Moors and heathland	0	3	4	Upland	0	8	0
Peatbogs	0	5	2	Wetlands	0	0	0
Natural grassland	0	3	4	Grassland	0	31	4
Agriculture & Natural vegetation	3	2	2	Grassland	0	31	4
Agriculture & Natural vegetation	3	2	2	Grassland	0	31	4
Beaches, dunes and sand plains	0	0	4	Coastal	0	0	4
Bare rock	0	0	3	No eco	0	0	0
Water courses	0	0	4	Rivers and lakes	0	0	0
Water bodies	0	1	4	Rivers and lakes	0	0	0
Coastal lagoons	0	0	4	Coastal	0	0	4
Estuaries	0	0	5	Marine inlets and transitional waters	0	0	2
Glaciers and perpetual snow	0	5	5	No eco	0	0	0

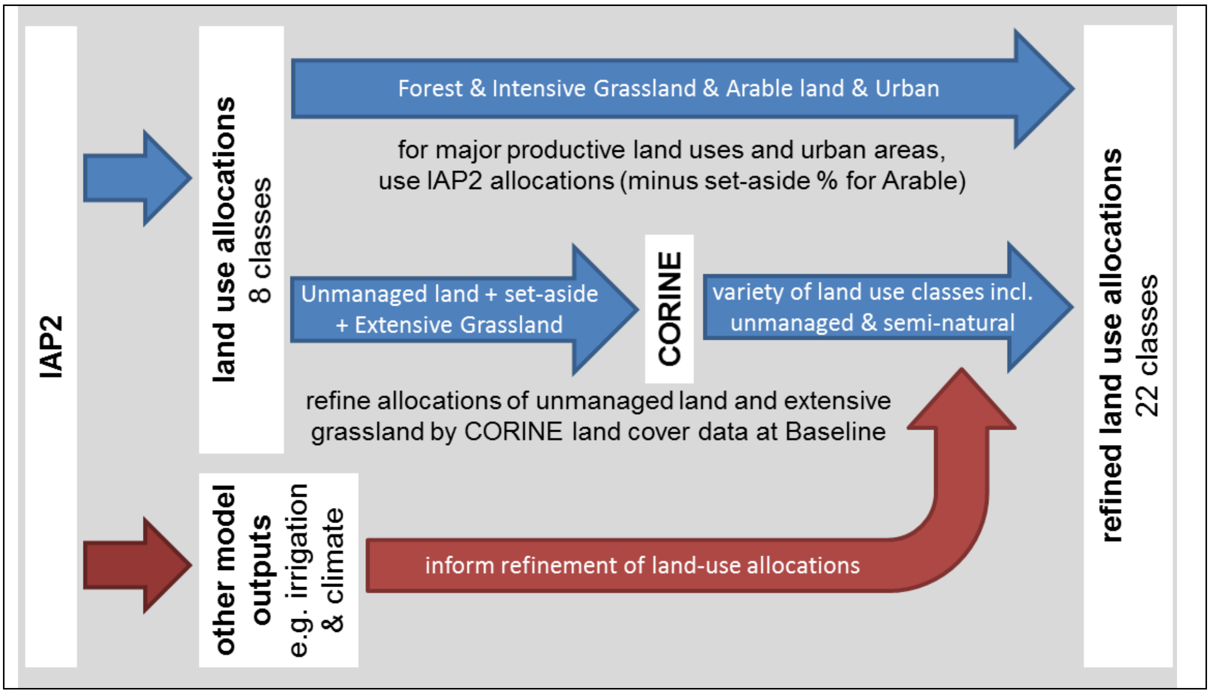
*In this case the IAP2 refined land use was represented by two classes in CORINE. For this class the maximum score of the two classes was used to represent each ecosystem service.

3. Supplementary methods

Box D1: Parameters used for each scenario

BL = "baseline";
SSP1 = "SSP1"; "RCP2.6"; 2080s; Climate model: EC-EARTH_RCA4
SSP3 = "SSP3"; "RCP8.5"; 2080s; Climate model: HadGEM2-ES_RCA4;
SSP4 = "SSP4"; "RCP4.5"; 2080s; Climate model: HadGEM2-ES_RCA4;

Box D2: More details on how the IAP2 land uses were refined



4. Histograms showing the raw data behind the ecosystem service provision maps

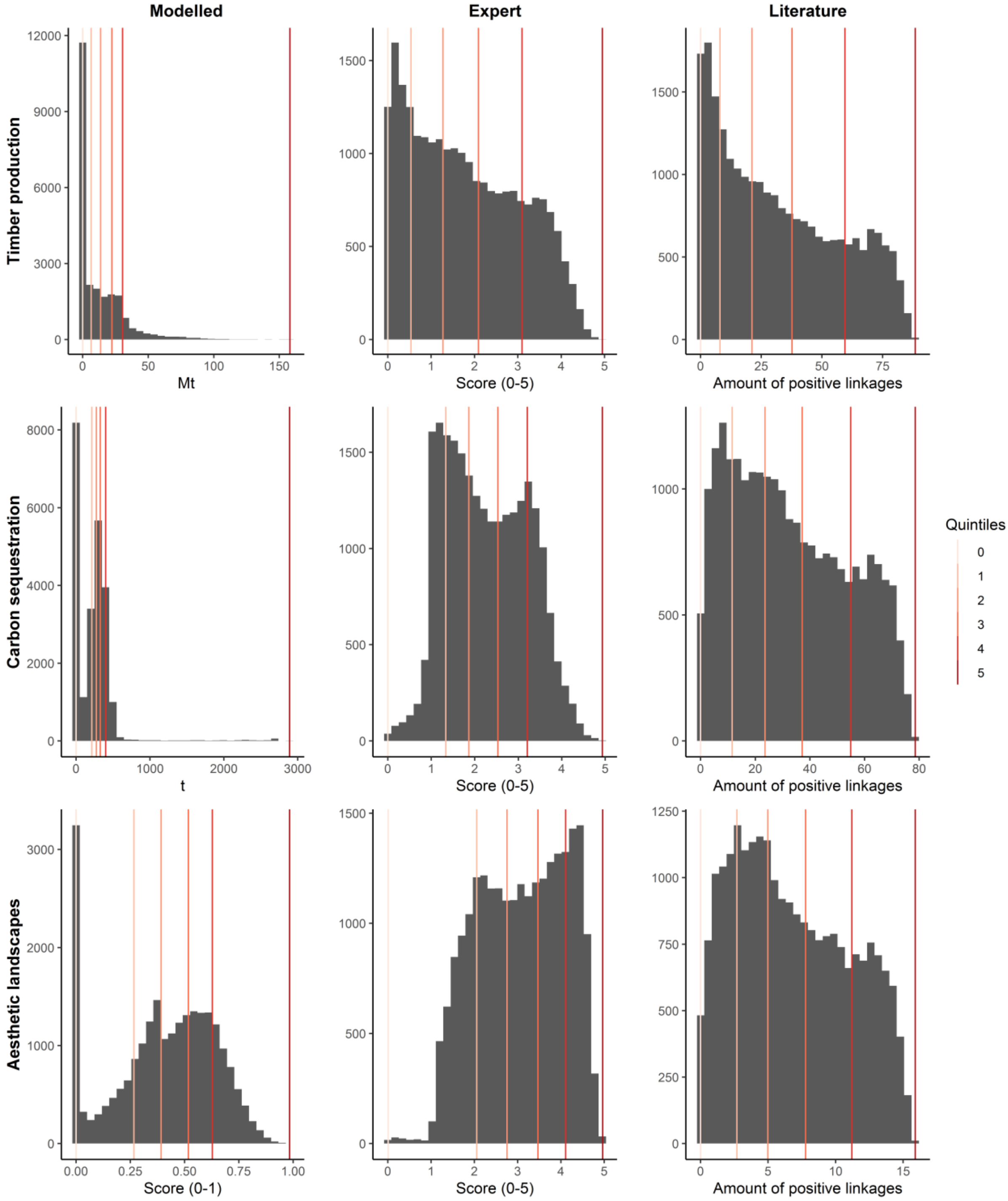


Figure D1: Histograms of the information source used to represent the different evidence type maps in Figure 23. The x intercepts show the value of the quintiles that the ecosystem service provision in the maps is colour categorised to. Due to the high skew in the modelled data quintiles are based off the non-zero data

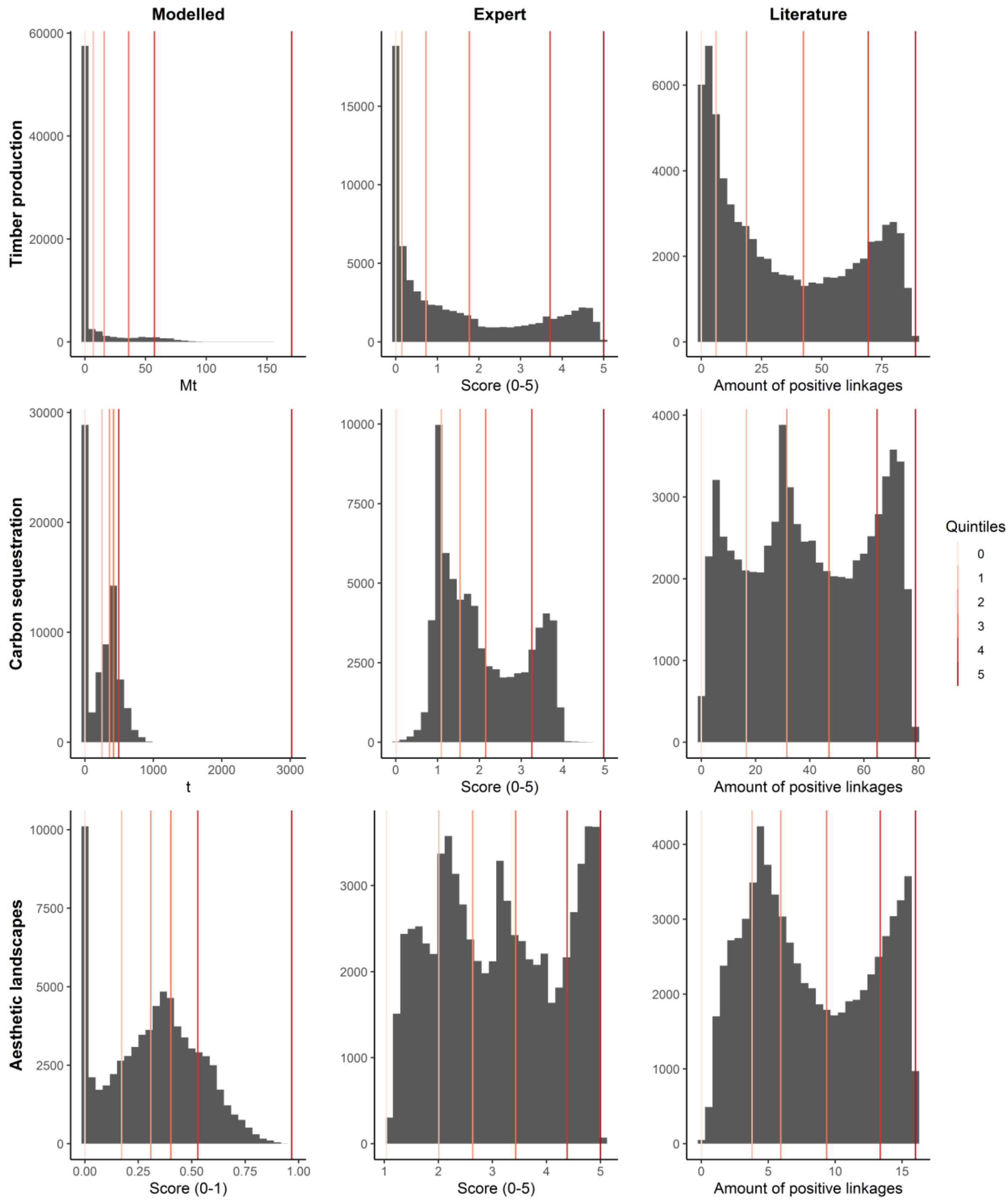


Figure D2: Histograms of the information source used to represent the different evidence type maps for all scenarios in Figure 25 of the manuscript and Figures D12 and D13. The x intercepts show the value of the quintiles that the ecosystem service provision in the maps is colour categorised to. Due to the high skew in the modelled data quintiles are based off the non-zero data but the zero data is shown in the histograms.

5. Land cover/use maps

CORINE land cover maps

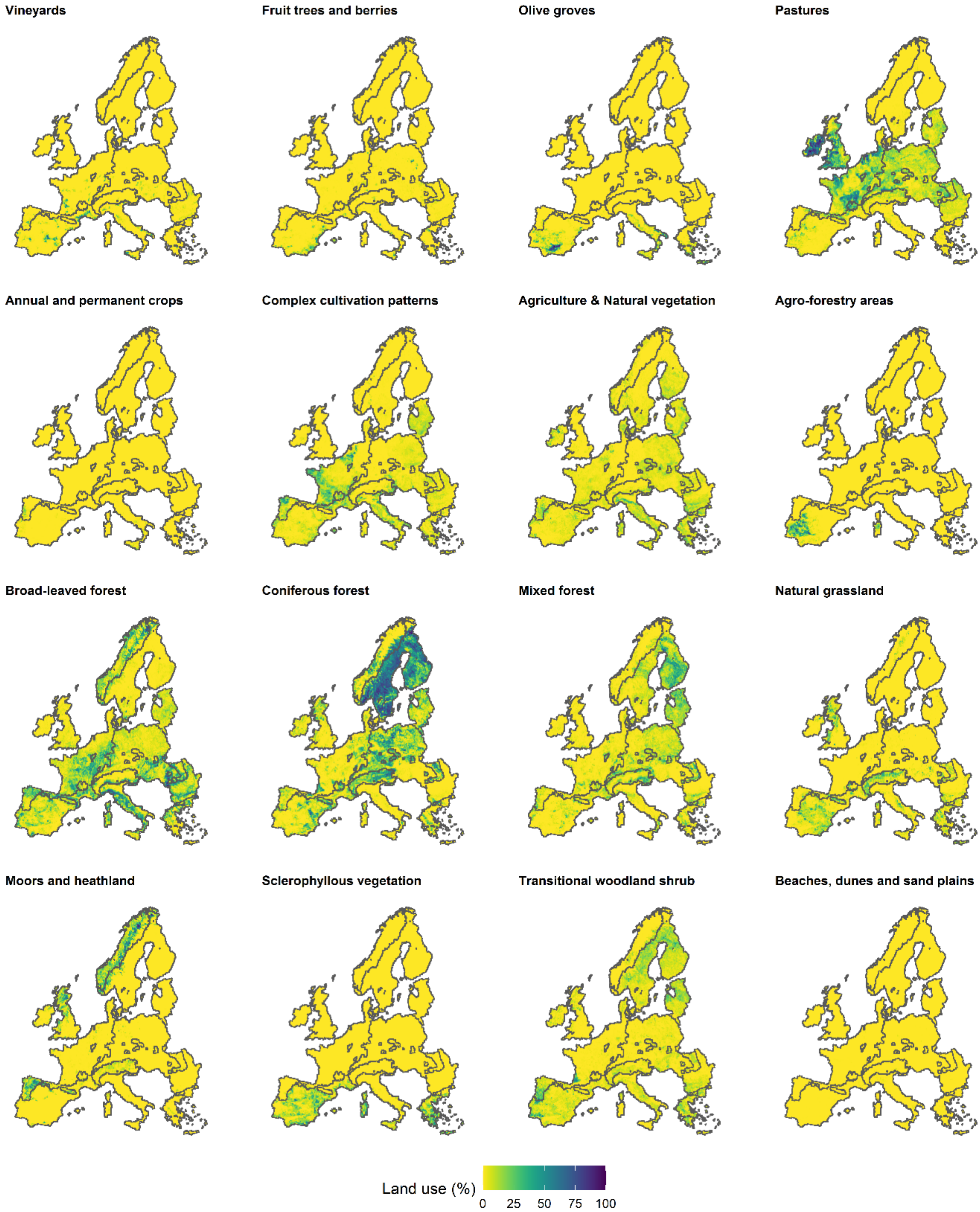

Vineyards
Fruit trees and berries
Olive groves
Pastures
Annual and permanent crops
Complex cultivation patterns
Agriculture & Natural vegetation
Agro-forestry areas
Broad-leaved forest
Coniferous forest
Mixed forest
Natural grassland
Moors and heathland
Sclerophyllous vegetation
Transitional woodland shrub
Beaches, dunes and sand plains
Land use (%)
0
25
50
75
100

Figure D3: Maps showing the amount the different CORINE2018 land cover types within each 10` x 10` cell resolution calculated using the Zonal Histogram function in QGIS (QGIS.org, 2022).

IAP2 land use maps

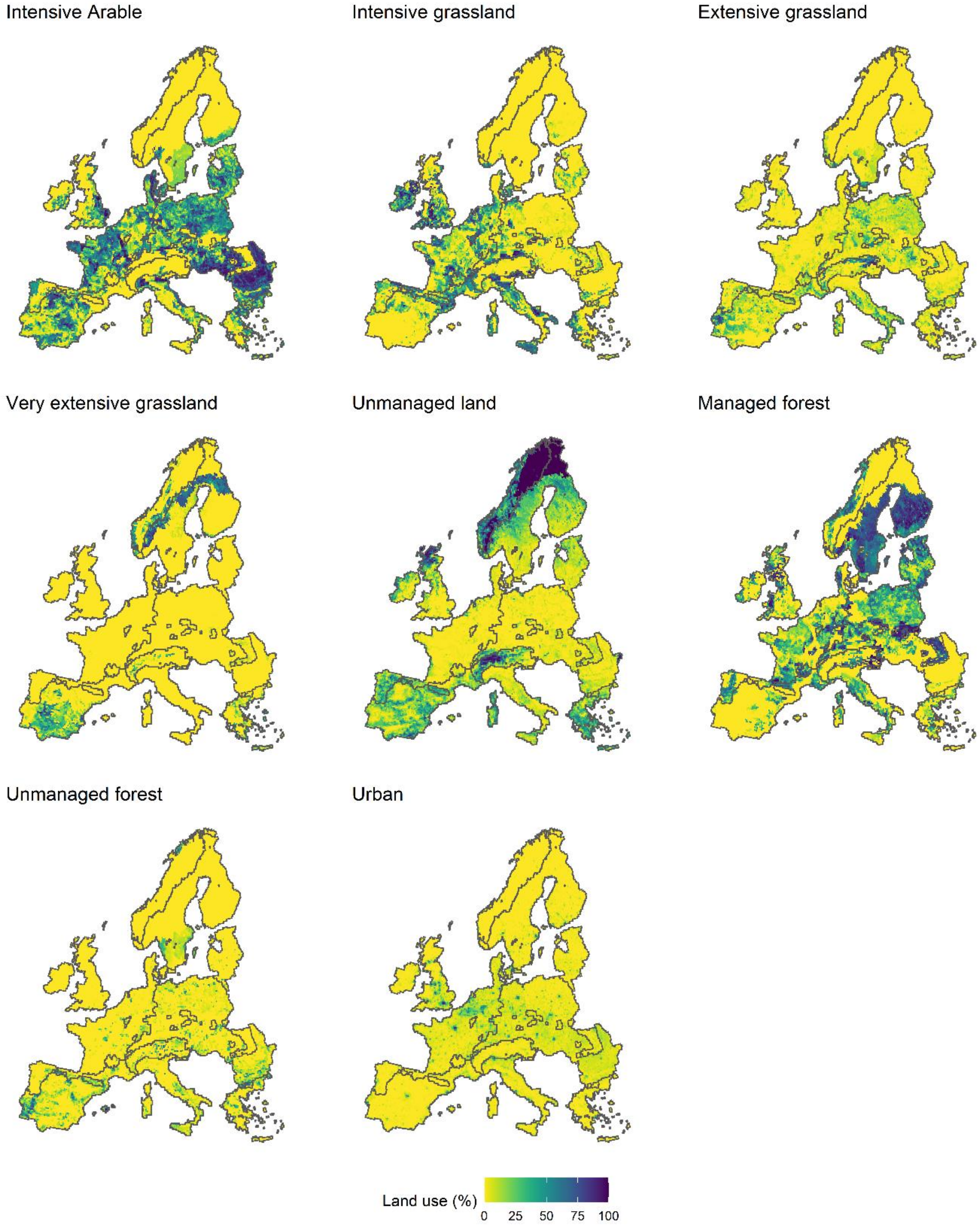


Figure D4: Amount of each different land use allocated by the IAP2 to each 10` by 10` cell for the baseline scenario.

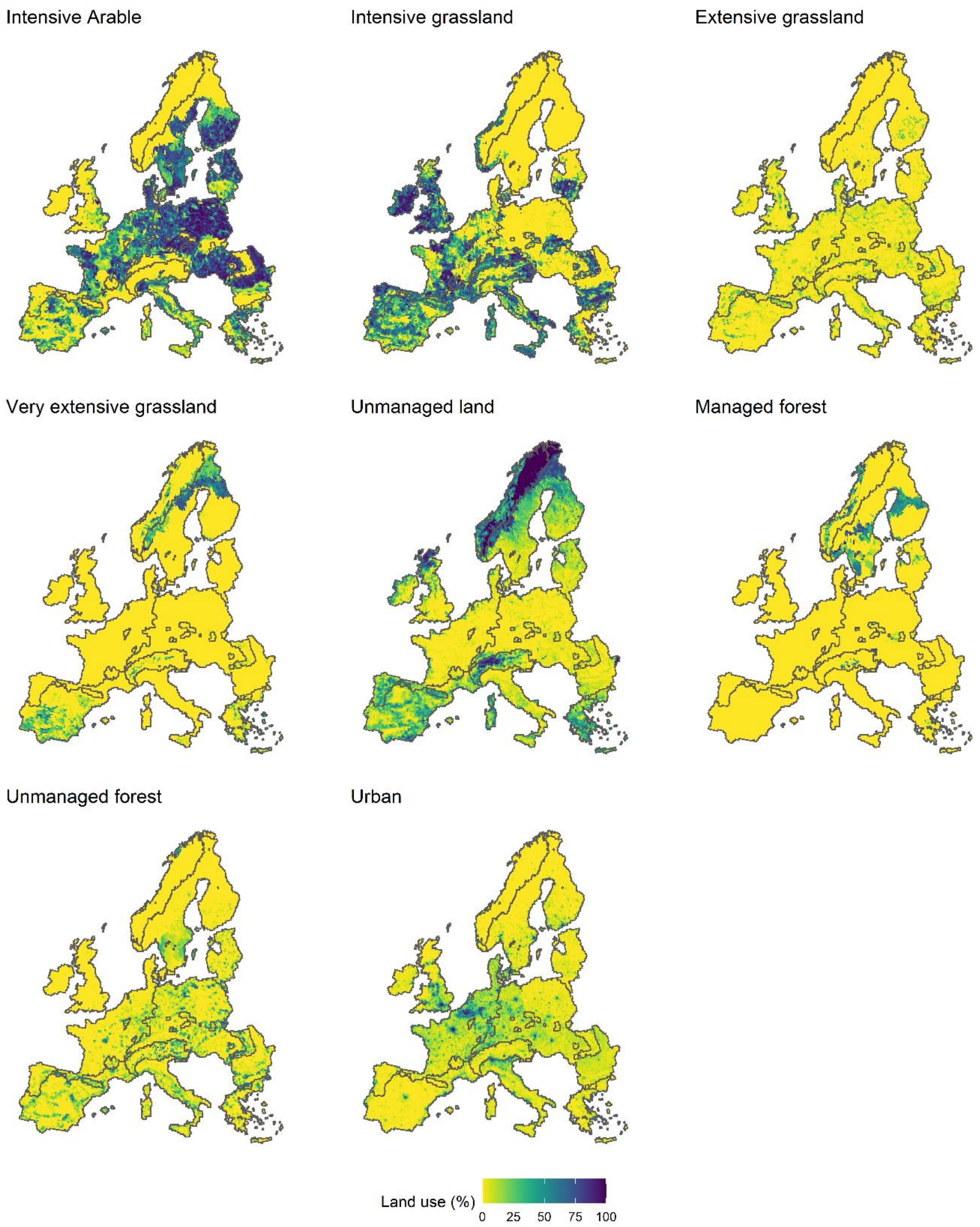


Figure D5: Amount of each different land use allocated by the IAP2 to each 10` by 10` cell for the SSP1xRCP2.6 scenario.

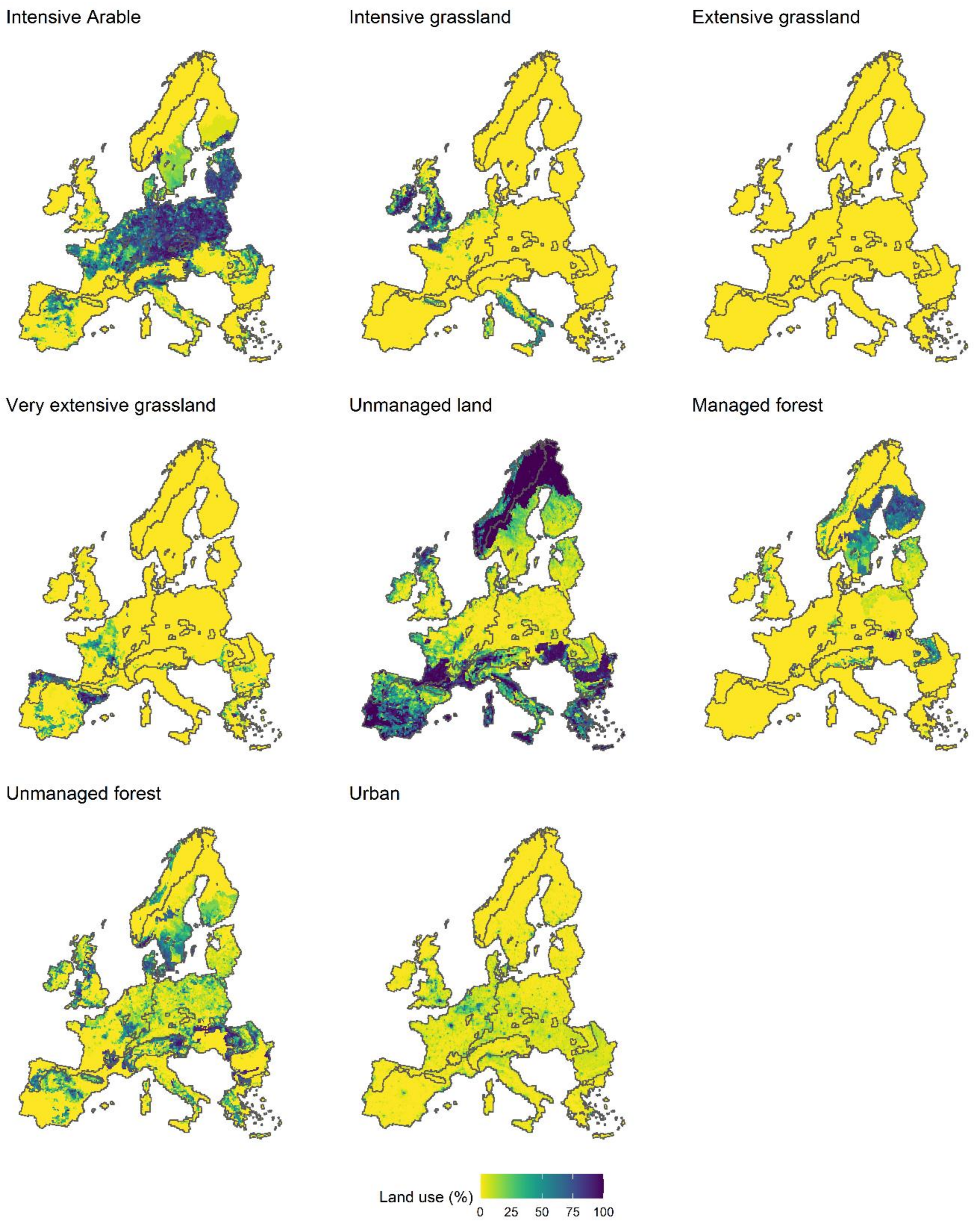


Figure D6: Amount of each different land use allocated by the IAP2 to each 10` by 10` cell for the SSP3xRCP8.5 scenario.

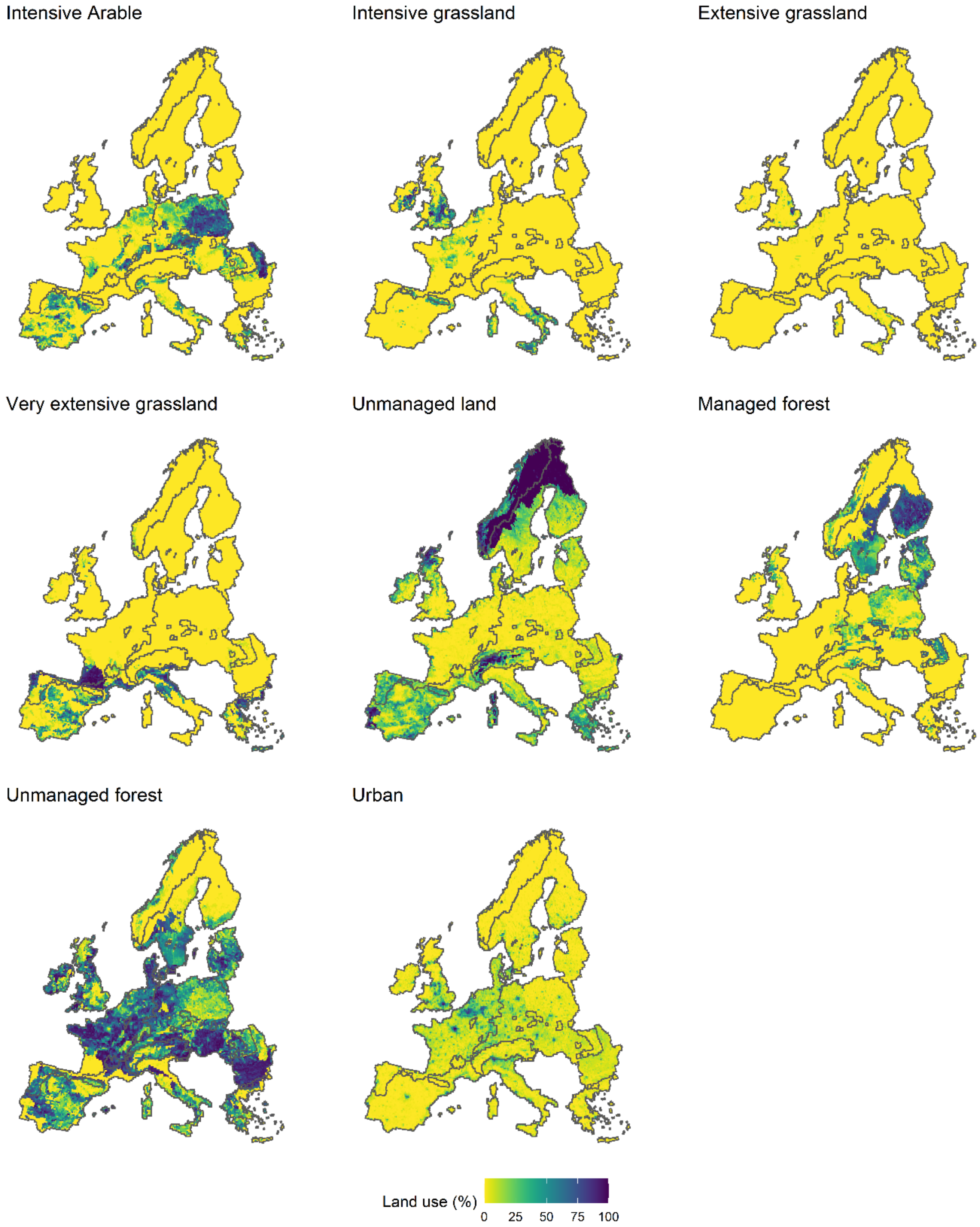
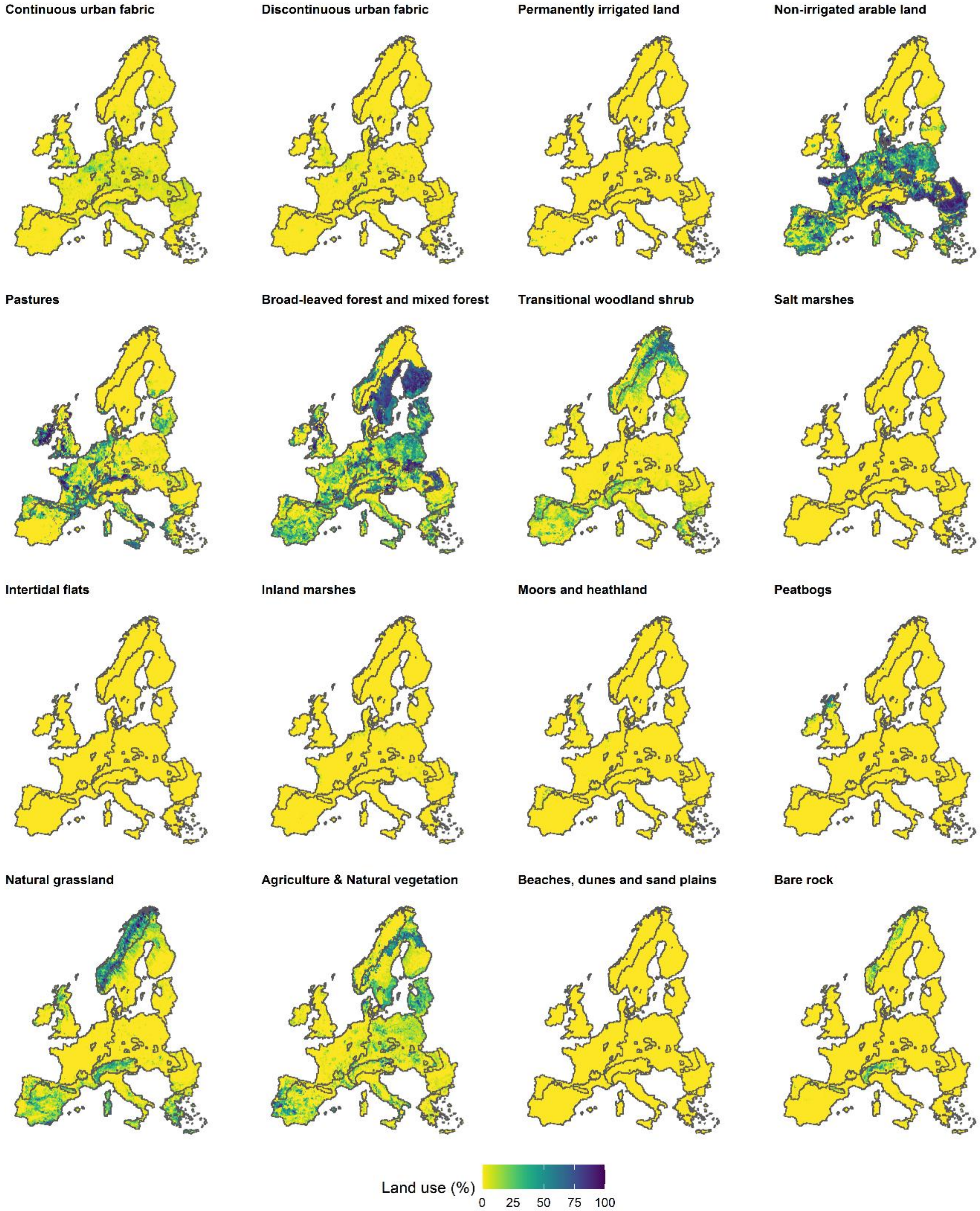


Figure D7: Amount of each different land use allocated by the IAP2 to each 10` by 10` cell for the SSP4xRCP4.5 scenario.

Refined IAP2 land use maps



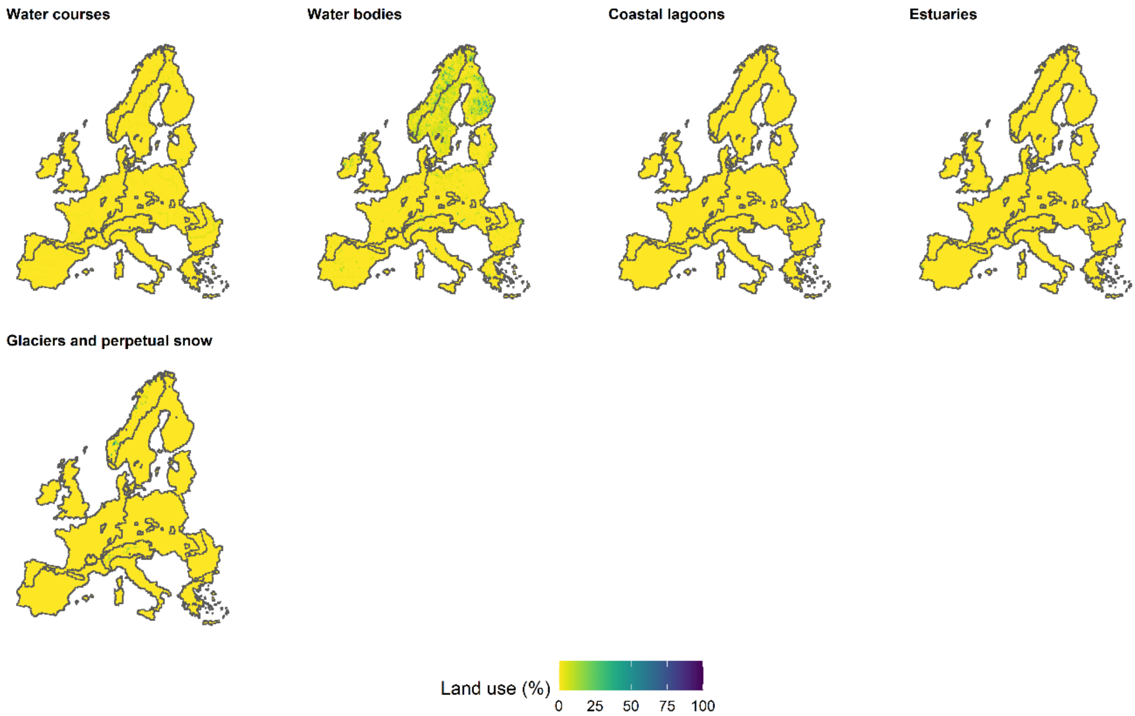


Figure D8: Amount of each different refined land use allocated to each 10` by 10` cell for the baseline scenario.

Continuous urban fabric
Discontinuous urban fabric
Permanently irrigated land
Non-irrigated arable land
Pastures
Broad-leaved forest and mixed forest
Transitional woodland shrub
Salt marshes
Intertidal flats
Inland marshes
Moors and heathland
Peatbogs
Natural grassland
Agriculture & Natural vegetation
Beaches, dunes and sand plains
Bare rock
Land use (%)
0
25
50
75
100

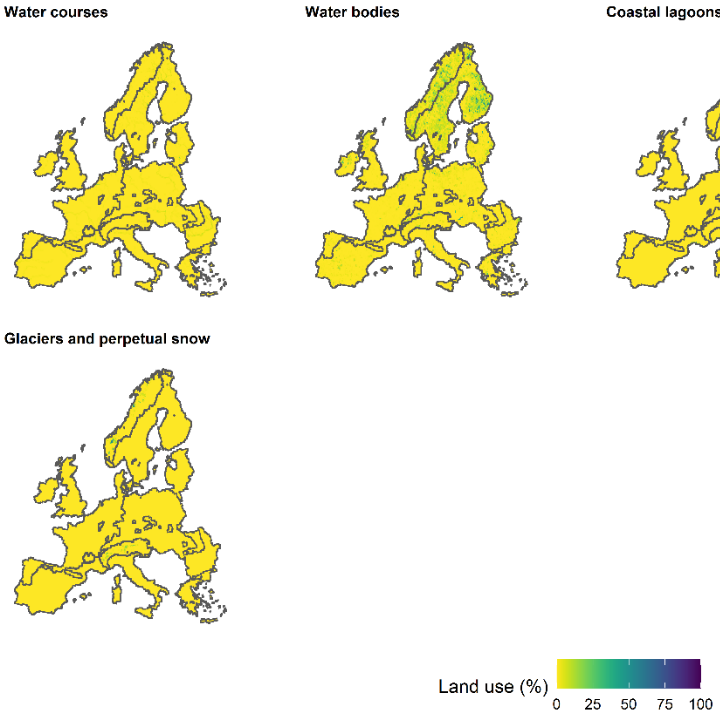
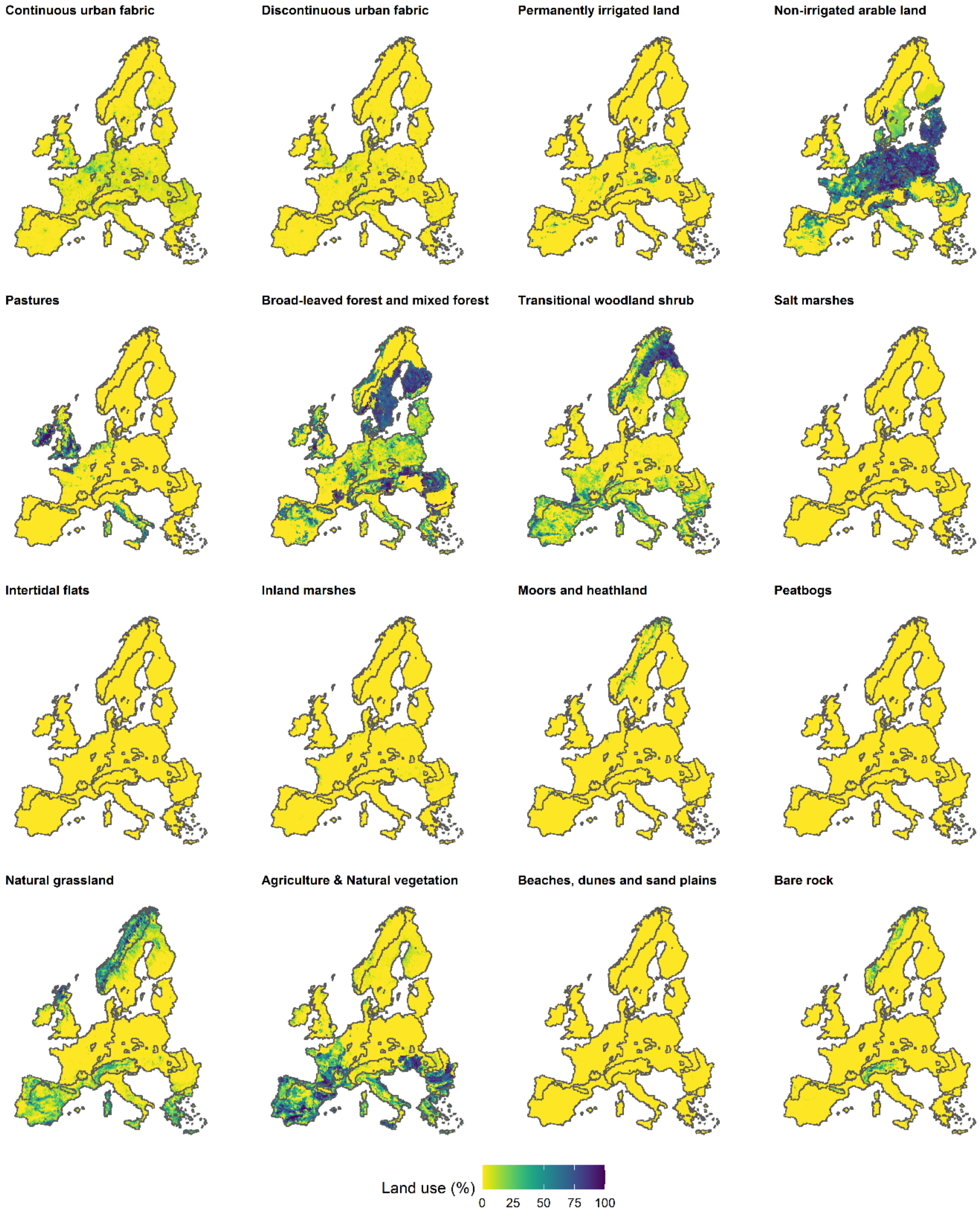


Figure D9: Amount of each different refined land use allocated to each 10` by 10` cell for the SSP1xRCP2.6 scenario.


Continuous urban fabric
Discontinuous urban fabric
Permanently irrigated land
Non-irrigated arable land
Pastures
Broad-leaved forest and mixed forest
Transitional woodland shrub
Salt marshes
Intertidal flats
Inland marshes
Moors and heathland
Peatbogs
Natural grassland
Agriculture & Natural vegetation
Beaches, dunes and sand plains
Bare rock
Land use (%)
0 25 50 75 100

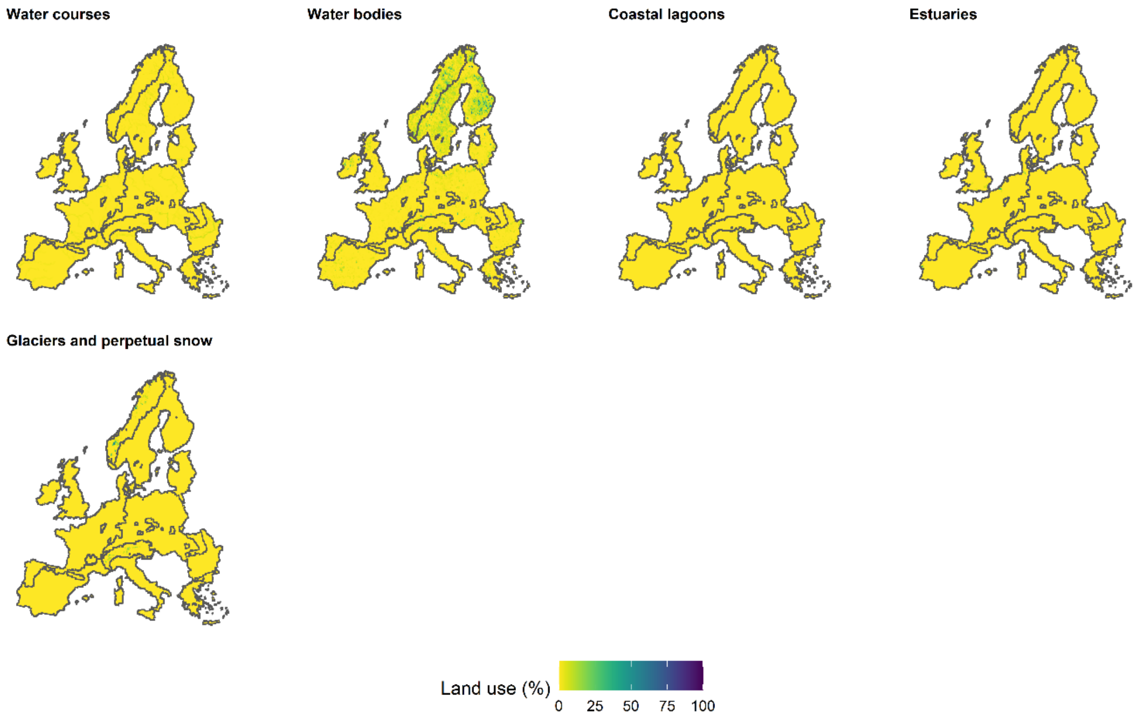
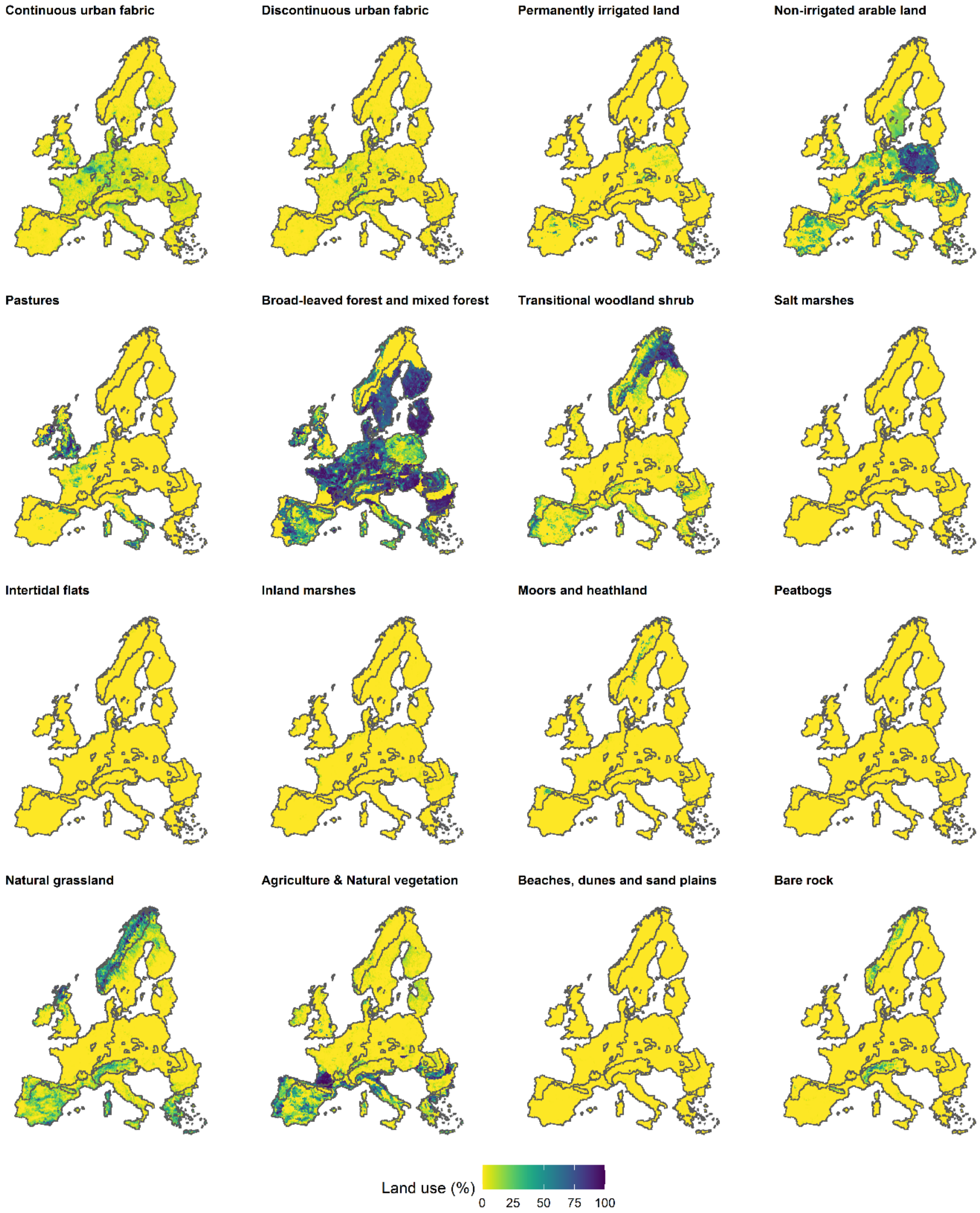


Figure D10: Amount of each different refined land use allocated to each 10` by 10` cell for the SSP3xRCP8.5 scenario.


Continuous urban fabric
Discontinuous urban fabric
Permanently irrigated land
Non-irrigated arable land
Pastures
Broad-leaved forest and mixed forest
Transitional woodland shrub
Salt marshes
Intertidal flats
Inland marshes
Moors and heathland
Peatbogs
Natural grassland
Agriculture & Natural vegetation
Beaches, dunes and sand plains
Bare rock
Land use (%)
0 25 50 75 100

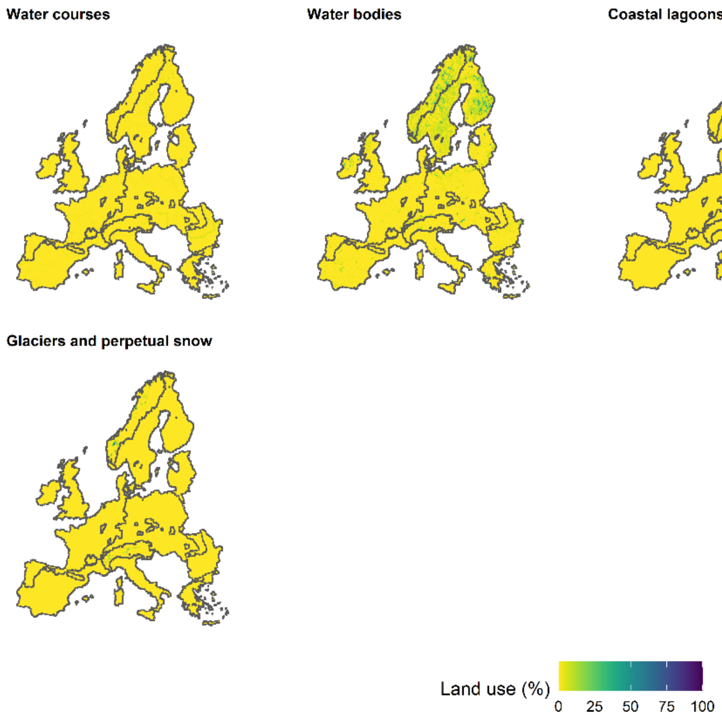


Figure D11: Amount of each different refined land use allocated to each 10` by 10` cell for the SSP4xRCP4.5 scenario.

6. Full Literature ecosystem to ecosystem service matrix

Table D4: LiNCAGES ecosystem to ecosystem service matrix. The values are the number of studies completed in each ecosystem that evidence positive natural capital attribute to ecosystem service linkages with each ecosystem service.

Ecosystem	Aesthetic landscapes	Air quality regulation	Atmospheric regulation	Food production (crops)	Freshwater fishing	Mass flow regulation	Pest regulation	Pollination	Recreation (species-based)	Timber production	Water flow regulation	Water quality regulation	Water supply
Coastal	4	0	0	0	0	0	0	0	19	0	0	5	0
Cropland	0	0	0	65	0	20	39	68	2	0	0	0	0
Grassland	4	0	31	0	0	10	8	16	0	0	1	4	1
Marine inlets and transitional waters	2	0	0	0	0	0	0	0	0	0	0	0	0
Rivers and lakes	0	0	0	0	60	0	0	0	6	0	2	18	0
Upland	0	0	8	0	0	36	0	0	0	0	3	0	0
Urban	13	24	0	0	0	0	0	0	3	0	3	0	0
Wetlands	0	0	0	0	0	0	0	0	0	0	4	42	0
Woodland and forest	16	7	79	0	0	0	3	0	5	89	27	8	12

7. Future ecosystem service provision maps

Table D5: Spearman correlation coefficient for the comparison of future timber production evidence types maps by region for each scenario for timber production. All comparisons were made at the cell level. A Spearman correlation coefficient of 1 indicates perfect positive correlation and a value of -1 indicates perfect negative correlation. The significance of the correlations are denoted by "" where: *** is p <0.001, ** is p<0.01 and * is p<0.05.*

Evidence compared	Scenario	Europe	Northern	Alpine	Atlantic	Continental	Southern
Modelled and literature	SSP1/RCP2.6	**0.39*** **	0.54***	0.28***	0.07***	0.19***	0.08***
	SSP3/RCP8.5	**0.43*** **	0.32***	0.37***	0.26***	0.24***	0.11***
	SSP4/RCP4.5	**0.39*** **	0.36***	0.35***	0.09***	0.07***	0.05**
Modelled and expert	SSP1/RCP2.6	**0.46*** **	0.7***	0.55***	0.1***	0.2***	0.09***
	SSP3/RCP8.5	**0.52*** **	0.83***	0.58***	0.28***	0.18***	0.12***
	SSP4/RCP4.5	**0.45*** **	0.67***	0.55***	0.08***	0.07***	0.1***
Expert and literature	SSP1/RCP2.6	**0.52*** **	0.51***	0.4***	0.45***	0.76***	0.53***
	SSP3/RCP8.5	**0.51*** **	0.39***	0.58***	0.78***	0.75***	0.13***
	SSP4/RCP4.5	**0.78*** **	0.65***	0.69***	0.98***	1***	0.78***

0.75 - 1
0.5 - 0.75
0.25 - 0.5
0 - 0.25
< 0

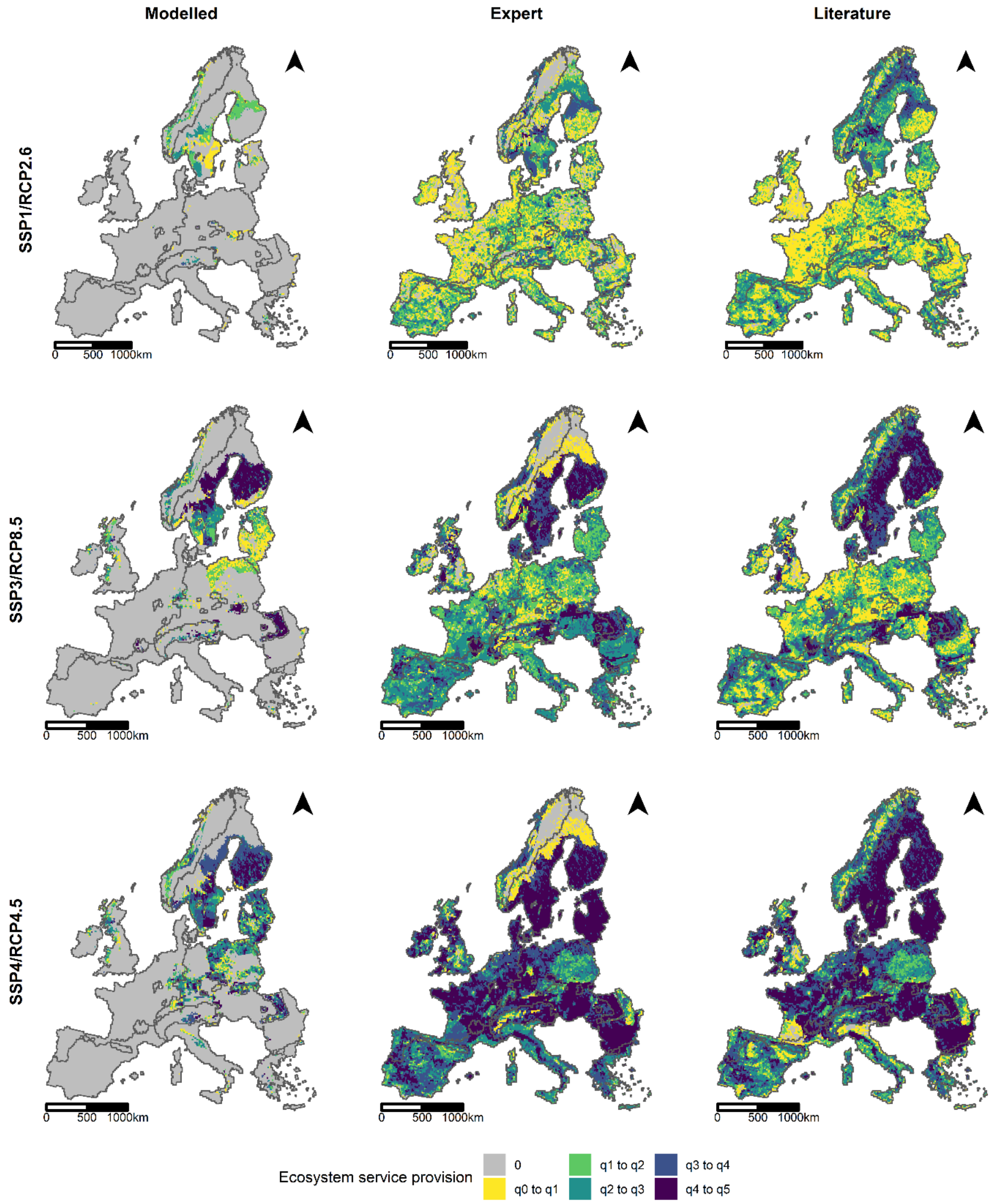


Figure D12: Timber production future ecosystem service provision maps for three possible future scenarios created using the three evidence types. Ecosystem service provision is coloured by quintiles (q) of the non-zero values of the data due to the diverse and skewed nature of the information sources representing the evidence types. Areas with zero ecosystem service provision are shown in grey. See Figure D2 for locations of the quintile boundaries shown on histograms of each information source behind each map.

Table D6: Spearman correlation coefficient for the comparison of carbon sequestration future evidence types by region for each scenario for carbon sequestration. A Spearman correlation coefficient of 1 indicates perfect positive correlation and a value of -1 indicates perfect negative correlation. The significance of the correlations are denoted by "" where: *** is p <0.001, ** is p<0.01 and * is p<0.05.*

Evidence compared	Scenario	Europe	Northern	Alpine	Atlantic	Continental	Southern
Modelled and literature	SSP1/RCP2.6	**0.25*** **	0.21***	0.19***	0.23***	0.54***	0.32***
	SSP3/RCP8.5	**0.15*** **	-0.29***	0.39***	0.32***	0.22***	0.35***
	SSP4/RCP4.5	**0.37*** **	0.24***	0.41***	0.29***	0.37***	0.56***
Modelled and expert	SSP1/RCP2.6	**0.16*** **	0.39***	0.15***	0.22***	0.29***	0.07***
	SSP3/RCP8.5	**0.39*** **	0.31***	0.65***	0.41***	0.33***	0.4***
	SSP4/RCP4.5	**0.56*** **	0.62***	0.66***	0.33***	0.35***	0.52***
Expert and literature	SSP1/RCP2.6	**0.66*** **	0.74***	0.25***	0.51***	0.63***	0.58***
	SSP3/RCP8.5	**0.65*** **	0.39***	0.41***	0.77***	0.9***	0.33***
	SSP4/RCP4.5	**0.75*** **	0.55***	0.5***	0.92***	0.99***	0.87***

- 0.75 - 1
- 0.5 - 0.75
- 0.25 - 0.5
- 0 - 0.25
- < 0

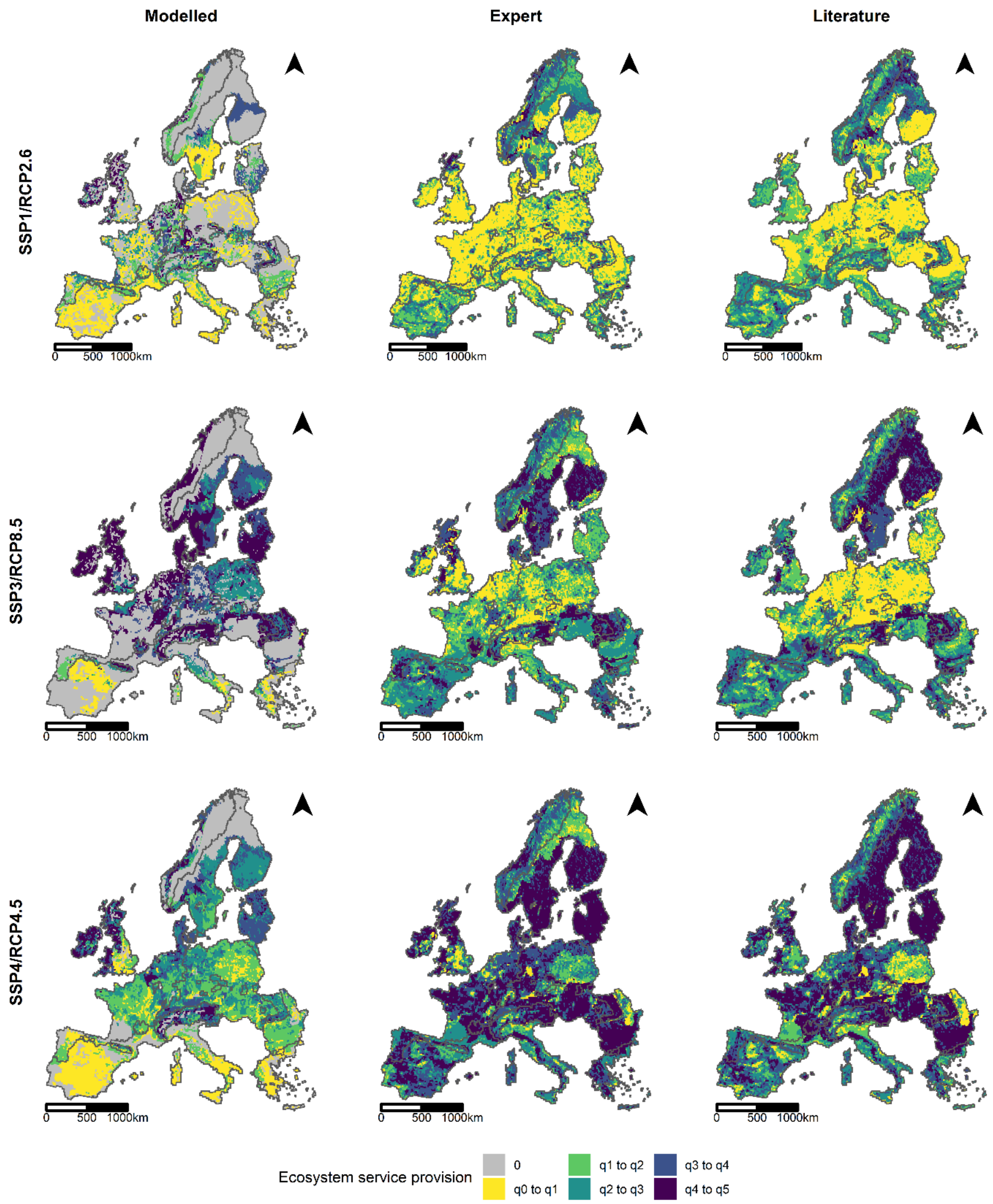


Figure D13: Carbon sequestration future ecosystem service provision maps for three possible future scenarios created using the three evidence types. Ecosystem service provision is coloured by quintiles (q) of the non-zero values of the data due to the diverse and skewed nature of the information sources representing the evidence types. Areas with zero ecosystem service provision are shown in grey. See Figure D2 for locations of the quintile boundaries shown on histograms of each information source behind each map.

8. Empirical comparison country

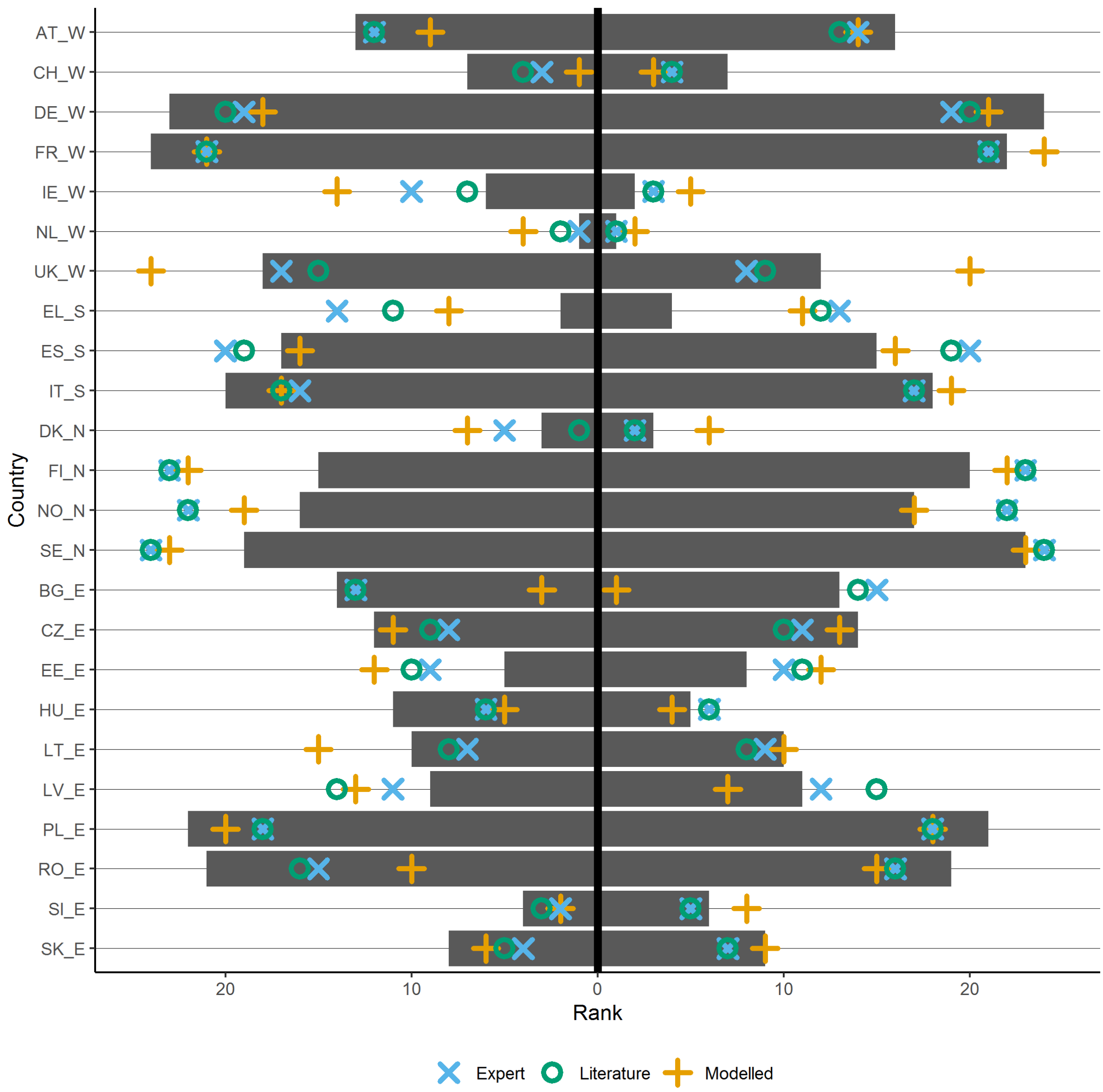


Figure D14: Mirrored bar plot comparing the ranked total ecosystem service provision by country between the three evidence sources (points) and empirical evidence (bars). The comparison for timber production is shown on the right and the comparison for carbon sequestration is shown on the left. The countries are ordered by the region of Europe they are located within (West (_W), South (_S), North (_N) and East (_E). Belgium and Denmark are excluded from this analysis due to lack of empirical data. Aesthetic landscapes is not included in this comparison due to lack of empirical data. For the glossary of country codes to country names see https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Glossary:Country_codes

Table D7: Spearman correlation coefficient for the evidence comparisons of the ranked total ecosystem service provision for Europe. All comparisons were completed at the country level. A Spearman correlation coefficient of 1 indicates perfect positive correlation and a value of -1 indicates perfect negative correlation. Aesthetic landscapes is not included in this comparison due to lack of empirical data.

	Evidence comparison					
Ecosystem service	Modelled and empirical	Expert and empirical	Literature and empirical	Modelled and expert	Modelled and literature	Expert and literature
Timber production	**0.84***	**0.89***	**0.9***	0.79***	0.8***	0.99***
Carbon sequestration	**0.69***	**0.78***	**0.83***	0.84***	0.82***	0.97***

0.75 - 1
0.5 - 0.75
0.25 - 0.5
0 - 0.25
< 0

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