A SPATIALLY EXPLICIT FRAMEWORK TO ASSESS SOIL EROSION PREVENTION VULNERABILITY

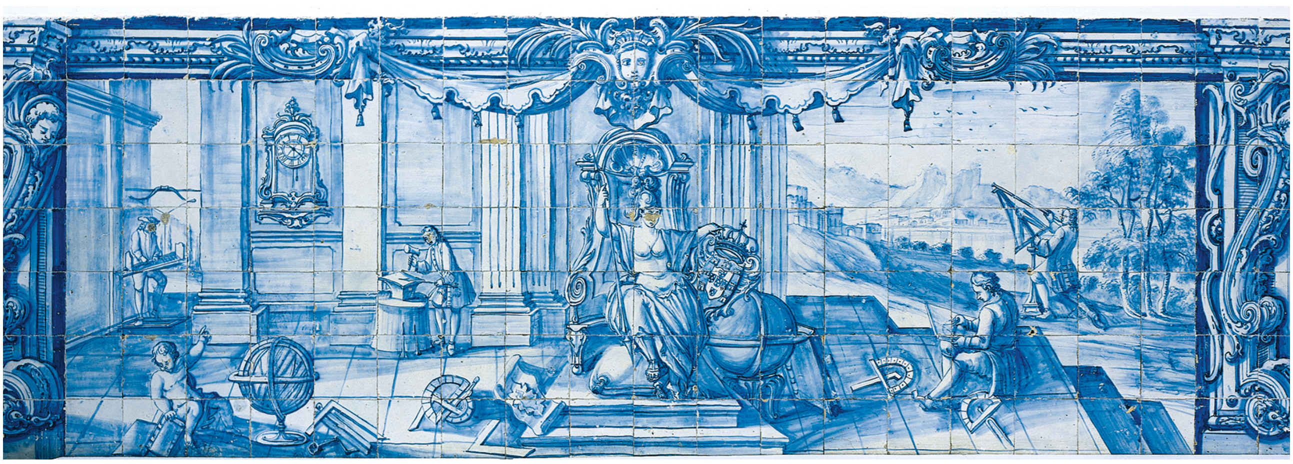
CONTRIBUTIONS FOR ECOSYSTEM SERVICE MAPPING AND MANAGEMENT IN MEDITERRANEAN LAND USE SYSTEMS

Carlos António Guerra

Tese apresentada à Universidade de Évora
para obtenção do Grau de Doutor em Gestão Interdisciplinar da Paisagem

ORIENTADORES: *Maria Teresa Pinto-Correia*
Marc J. Metzger

ÉVORA, OUTUBRO DE 2015



INSTITUTO DE INVESTIGAÇÃO E FORMAÇÃO AVANÇADA

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Liberdade

Ai que prazer
Não cumprir um dever,
Ter um livro para ler
E não fazer!
Ler é maçada,
Estudar é nada.

Sol doira
Sem literatura
O rio corre, bem ou mal,
Sem edição original.
E a brisa, essa,
De tão naturalmente matinal,
Como o tempo não tem pressa...

Livros são papéis pintados com tinta.
Estudar é uma coisa em que está indistinta
A distinção entre nada e coisa nenhuma.

Quanto é melhor, quanto há bruma,
Esperar por D. Sebastião,
Quer venha ou não!

Grande é a poesia, a bondade e as danças...
Mas o melhor do mundo são as crianças,

Flores, música, o luar, e o sol que peca
Só quando, em vez de criar, seca.

Mais que isto
É Jesus Cristo,
Que não sabia nada de finanças
Nem consta que tivesse biblioteca...

(Fernando Pessoa)

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Chapter 3

Guerra, C.A., Metzger, M.J., Maes, J., Pinto-Correia, T., 2015. Policy impacts on regulating ecosystem services: looking at the implications of 60 years of landscape change on soil erosion prevention in a Mediterranean silvo-pastoral system. Landscape Ecology. 10.1007/s10980-015-0241-1

Chapter 4

Guerra, C.A., Pinto-Correia, T., 2015. Linking farm management and ecosystem service provision: challenges and opportunities for soil erosion prevention in Mediterranean agroforestry systems. Land Use Policy. [Accepted]

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Guerra, C.A., Maes, J., Geijzendorffer, I., Metzger, M.J., 2016. An assessment of soil erosion prevention in Mediterranean Europe: current trends of ecosystem service provision. Ecological Indicators. 60, 213–222. 10.1016/j.ecolind.2015.06.043

LIST OF OTHER PEER-REVIEWED PUBLICATIONS 2012-2015

Vicente, J., Alagador, D., **Guerra**, C., Cabral, J., Honrado, J., Alonso, J., Araújo, M. & Fernandes, R., [submitted]. A modelling framework to guide monitoring of alien plant invaders under environmental change scenarios. Journal of Applied Ecology.

Maes, J., Fabrega, N., Zulian, G., Barbosa, A., Vizcaino, P., Ivits, E., Polce, C., Vandecasteele, I., Marí Rivero, I., **Guerra**, C., Perpiña Castillo, C., Vallecillo, S., Baranzelli, C., Barranco, R., Batista e Silva, F., Jacobs-Crisoni, C., Trombetti, M., Lavalle, C., 2015. Mapping and Assessment of Ecosystems and their Services: Trends in ecosystems and ecosystem services in the European Union between 2000 and 2010. JRC Technical Reports, Joint Research Center - European Commission Ispra. 138 pp.

Barbosa A., Lavalle C., Vandecasteele I., Viscaino P., Vallecillo S., Perpiña C., Riviera I.M., **Guerra** C., Baranzelli C., Jacobs-Crisioni C., Batista e Silva F., Zulian G., Maes J., 2014. Progress towards 'Green Growth' Europe: An assessment of the efficient use of natural resources in the EU-28 up to 2050. JRC Technical Reports, Joint Research Center - European Commission, Ispra. 52 pp.

Pinto-Correia, T., Guiomar, N., **Guerra**, C., Carvalho-Ribeiro, S., 2015. Towards a new typology of rural Europe: integrating the role of countryside consumption and protection in the assessment of new modes of rural occupance, Land Use Policy.

Lomba, A., **Guerra**, C., Alonso, J., Jongman, R. & McCracken, D., 2014. Mapping and monitoring High Nature Value farmlands: challenges in European landscapes. Journal of Environmental Management, 143C, DOI: 10.1016/j.jenvman.2014.04.029

Almeida, M., **Guerra**, C. & Pinto-Correia, T., 2013. Developing methodological approaches to build indicator sets able to measure the effects of policies in rural areas - Case study in Alentejo, South Portugal. Danish Journal of Geography. DOI: 10.1080/00167223.2013.848611

Guerra, C., Metzger, M.J., Honrado, J., Alonso, J., 2013. A spatially explicit methodology for a priori estimation of field survey effort in environmental observation networks. International Journal of Geographical Information Science. DOI:10.1080/13658816.2013.805760

Ortega, M., **Guerra**, C., Honrado, J.P., Metzger, M.J., Bunce, R.G.H., Jongman, R.H.G., 2013. Surveillance of habitats and plant diversity indicators across a regional gradient. Ecological Indicators, 33, 36-44 pp. DOI: 10.1016/j.ecolind.2012.12.004

Costa, F., Marques, A., Arnaud-Fassetta, G., Alonso, J., Martins, I. & **Guerra**, C., 2013. Methodological deepening for flood-risk analysis in cross-border regions: a case study for the Lima catchment (NW Spain/Portugal) according to the implementation of the European Water Framework Directive.125-136 pp.. *in* Arnaud-Fassetta, G., Masson, E., & Reynard, E., European Continental Hydrosystems under Changing Water Policy, Friedrich Pfeil Verlag, München, 344 pp.

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"O valor das coisas não está no tempo que elas duram, mas na intensidade com que acontecem. Por isso existem momentos inesquecíveis, coisas inexplicáveis e pessoas incomparáveis."

Fernando Pessoa

"The value of things it is not on the time that they last, but in the intensity that they happen. That is why there are unforgettable moments, unexplained things and incomparable people."

Fernando Pessoa

This is the story of my PhD, a roller coaster of unforgettable moments, unexplained things and incomparable people.

The unforgettable moments

I decided to begin this PhD project when my Son was born. Something in me opened to the possibility of doing something greater then myself. I still remember the day that I went with my 3 months old Son to Évora to discuss my PhD. At the time Teresa must have thought that she made a mistake accepting me as her student and I… well I drove away thinking that I would never be successful in a PhD. Between that summer day and today I have met several people, discussed with many others, and learned many things but maybe the most important one came from the question: "*Why do we have to study people to understand landscape dynamics?*". The answer changed entirely my perspective and became a central piece in the development of my ideas. In the end I decided to make my own unforgettable moment, bought a caravan and decided to go and work in Évora.

The unexplained things

Crazy things happen in life and even today I wonder why did a person that only saw one of my papers invited me to work with him. I can exactly remember my surprise and my excitement… This was one of the moments in recent times that helped me to regain some of my naive way of seeing life and for that I have to thank Joachim.

The incomparable people

Deciding to begin a PhD project is not something that one does lightly. In my case it was no different and I give my decision to my wife Andreia that always supported me in this decision and in all its consequences. Together with Afonso, my Son, they have been my source of love, inspiration, strength and resilience during all the difficult and happy times. I just hope to live up to their expectations and to be able to give back everything that they gave me transformed into something even better. They deserve it.

Two persons that are really incomparable for their constant effort to "the cause" are Marc and Teresa. Not only they kept up with my endless requests but also they gave me the inspiration, the tools, the attention, and the everlasting support to conclude my PhD. Thank you for everything. In the role of incomparable people that were with me during the last 3 years it is impossible for me to not mention Joachim. After inviting me to go to Italy he literally received me in his home and made the distance to

my family shorter and more bearable. These three persons more than mentors became my friends and I owe them the motivation and advices that allowed me to conclude my PhD the way I did.

Although I am an antisocial person, many other people have directly and indirectly contributed to my PhD, and it would be impossible to refer to them all. Despite this a special thank you has to go to some of the people that with their support and discussions contributed to my construction as a person and as a researcher and were a great part of my choices and scientific and personal decisions, even if they don't know that. In this group of unforgettable people are some of my dearest friends including Maria Amélia (my mother), Alonso, João, Joana, Ângela, Nuno, Luciano, Lina, Ermelinda, Domingos, and so many others that profoundly populate my ideas with innovation and creativity.

Although it would be great to tell everyone that this is a product of my effort, in reality this dissertation and all that let to it are a product of the effort of all of the above mentioned people. My little work was just to be open enough to receive all their love, patience, and inspiration and with their square and compass put it in writing.

Thank you all for your help and support!

ABSTRACT

This Thesis presents a spatially and temporally explicit and novel framework to identify, quantify and map soil erosion prevention aimed at establishing the links between the provision of ecosystem services and policy implementation and land management. Recent studies show that introducing the ecosystem service concept into policy and decision-making requires spatially explicit information on the state and trends of ecosystems and their services. Current spatially explicit approaches often fail to make a distinction between the actual ecosystem service provision and the underlying ecosystem capacity to provide a specific service.

Here we propose and test a spatially and temporally explicit ecosystem service assessment framework to assess the provision of the soil erosion prevention service based on the combination of biogeographic, ecological and land management factors. This ecosystem service is of particular importance in Mediterranean areas dominated by silvo-pastoral systems, which are prone to soil degradation, often caused by a wide variety of social and policy drivers. The proposed framework estimates the service provision based on the spatial and temporal arrangement of potential impacts and its relation to the actual and potential ecosystem service provision, allowing to identify thresholds of management change and vulnerability hotspots.

Results show that it is not possible to directly relate the capacity for ecosystem service provision with the actual ecosystem service provision. In the particular case of regulating services, establishing this link in mapping and assessment exercises will result in incorrect evaluations due to the spatial and temporal mismatch between these two indicators. The results also show the importance and impacts of land management and policy implementation in the provision of soil erosion prevention in Mediterranean land use systems. Overall, the combination of the different aspects of this Thesis provides a comprehensive overview of the potential applications of the proposed ecosystem service indicators to quantify and map soil erosion prevention and to support monitor and decision-making exercises at both land management and policy levels.

Um quadro de análise espacialmente explícito para a avaliação da vulnerabilidade da prevenção de erosão hídrica: contribuições para o mapeamento e gestão de serviços de ecossistema em usos do solo mediterrânicos

RESUMO

Esta Tese apresenta um novo quadro de análise espacial e temporalmente explícito para identificar, quantificar e mapear o serviço de ecossistema de prevenção de erosão do solo com o objectivo de estabelecer relações entre a provisão de serviços de ecossistema e a implementação de políticas e a gestão da terra. Estudos recentes mostram que a introdução do conceito de serviços de ecossistema esfera política e na tomada de decisão requer informação espacialmente explícita sobre o estado e as tendências dos ecossistemas e seus serviços. Atualmente as abordagens espacialmente explícitas muitas vezes não conseguem capturar a distinção entre a provisão de serviços de ecossistema efetiva e da capacidade do ecossistema para fornecer um serviço específico.

Aqui propomos e testamos um quadro de análise, espacial e temporalmente explícito, para avaliar a provisão do serviço de ecossistema de prevenção de erosão do solo com base na combinação de fatores biogeográficos, ecológicos e de gestão da terra. Este serviço de ecossistema é de particular importância nas zonas mediterrânicas dominadas por sistemas silvo-pastoris, que são propensas a degradação do solo, muitas vezes causada por uma grande variedade de fatores sociais e políticas. O quadro de análise proposto estima a provisão de serviços de ecossistema com base no arranjo espacial e temporal do impacte potencial e sua relação com a efetiva e potencial provisão de serviços de ecossistema, permitindo identificar limiares de mudança de gestão e *hotspots* de vulnerabilidade.

Os resultados mostram que não é possível relacionar diretamente a capacidade de provisão de serviços de ecossistema com a efetiva provisão de serviços de ecossistema. No caso particular dos serviços de regulação, estabelecer esta relação em exercícios de mapeamento e avaliação irá necessariamente resultar em avaliações incorretas devido ao desencontro espacial e temporal entre esses dois indicadores. Os resultados também mostram a importância e os impactes resultantes da gestão da terra e da implementação de políticas na prevenção da erosão do solo em sistemas de uso da terra Mediterrânicos. De uma forma geral, a combinação dos diferentes aspectos desta Tese fornece uma visão abrangente das potenciais aplicações dos indicadores de serviços de ecossistema propostos para quantificar e mapear a prevenção da erosão do solo e apoiar exercícios de monitorização e de tomada de decisão, tanto ao nível da gestão da terra como da definição de políticas.

PREAMBLE

During the last decades, European ecosystems and particularly Mediterranean ones have been under constant stress due to a high dynamic in human activities and constant policy shifts and productivist guidelines. In this context it is of the utmost importance to devise monitoring systems that can collect relevant data that can contribute to future policy design at the same time that can be useful to land managers to fulfill the policy requirements. This would allow to determine vulnerable areas where action is needed and to give guidelines to tackle the identified vulnerabilities.

The challenge is that this process of supporting decision-making has to include indicators that reflect the ecosystem dynamics both at the land management and the policy scales. To overcome this challenge it is necessary to develop an analytical framework focused on process related indicators that are suitable to monitor highly dynamic realities. These have also to be representative both in space and time, as most ecological processes vary in these two dimensions. So the question now is what and how to monitor.

The ecosystem service concept provides a step forward in the communication of natural values to decision makers. Although current methodological approaches focus on proxy indicators of ecosystem service provision, there is a significant amount of scientific background to develop new frameworks to tackle the previously identified issues. In the case of regulating services, ecosystem service provision results from the mitigation of a potential impact by the ecosystem service provider. If it is possible to measure and assess the spatial and temporal distribution of the ecosystem service provision, the next step would be to test and evaluate the sensitivity of this indicator for policy and land management support. Within the current scientific background, illustrating the potential of ecosystem service indicators to monitor, assess and evaluate specific policies and land management practices requires a novel conceptual and methodological framework and the selection of a relevant ecosystem service to test it.

This is the beginning of this PhD project and the main idea behind this dissertation.

1

INTRODUCTION

1.1 Fundamental challenges and research opportunities

The ecological balance of our planet is rapidly changing (Ellis 2011; Hughes et al. 2013) and these changes are expected to accelerate during the next decades (MA 2005a; IPCC 2007; EEA 2015). Over the last few centuries, the world's cultural and natural landscapes have changed profoundly (Thuiller et al. 2008; Darnhofer et al. 2010), affecting their functions, processes and values (Castro et al. 2010). As these problems become more apparent, world leaders have also become more aware of the social, economic and ecological implications resulting from these impacts (Daily and Matson 2008).

In this context, several conventions and protocols meant to protect natural values were signed both at the European[1] and global[2] levels. These and other policy guidelines aim to improve global and regional sustainable management of ecosystem resources and natural values, and imply a coordinated response from signatory countries (Le Blanc et al. 2012). They also brought visibility to nature's values and have raised awareness to the need to prioritize biodiversity, ecosystems and the services they provide in several areas of decision-making and social-economic development (SCBD-UNEP 2010). This prioritization is reflected in several European[3] and national[4] directives, policies and/or assessments that constitute the backbone of the European wide environmental protection strategy. Although many efforts were made, studies show that the targets that were set were not met (Leadley et al., 2010; Jones et al., 2011a; Pereira et al., 2012a) and that in some cases the main pressures and impact drivers are intensifying (Thuiller et al. 2008; Abbott and Le Maitre 2009; Lindner et al. 2010).

In the European context some of the most important drivers include: i) increasing human pressure in rural and peri-urban areas and the implementation of regional/high dimension projects (Swaffield and Primdahl 2010; Rounsevell et al. 2012); ii) intensification of productive strategies (Oñate et al. 2007; Dibden et al. 2009); iii) abandonment of productive activities and land, specially in rural, economically undeveloped and socially depreciated regions (Henle et al. 2008; Figueiredo and Pereira 2011); iv) the application of uncoordinated economic, social and environmental policies (Korhonen and Seager 2008; Pe'er et al. 2014); and v) the increase of environmental risks resulting from man made disturbances and/or climate change (Schröter et al. 2005; Scholze et al. 2006; Lindner et al. 2010; Shrestha et al. 2013). These and other still existing drivers raise the importance of the evaluation and impact assessment of policy design and implementation not only for current and past conditions but also for future policy and environmental scenarios (Maes et al. 2013b).

The need to establish causal relations between social/environmental variables and the resulting impacts, and to mitigate their social and ecological consequences puts a focus on having monitoring systems that provide information not only on the impacts themselves but also on their related

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[1] e.g. the *European Diploma of Protected Areas* [1965], the *Convention on the Conservation of European Wildlife and Natural Habitats* [1979], the *European Outline Convention on Transfrontier Co-operation between Territorial Communities or Authorities* [1980], the *European Landscape Convention* [2000], the *Convention on the Value of Cultural Heritage for Society* [2005], the *EU Biodiversity Strategy to 2020*.

[2] e.g. the *Convention concerning the Protection of the World Cultural and Natural Heritage* [1972], the *Ramsar Convention* [1971], the *Convention on Biological Diversity* [1992], the *UN Framework Convention on Climate Change* [1994], the *Global Strategy for Plant Conservation* [2002], the *Nagoya Protocol* [2010], the *Strategic Plan for Biodiversity 2011-2020 and the Aichi targets*.

[3] e.g. *Water Framework Directive*, *Habitats Directive*, *Soil Thematic Strategy*, *Biodiversity Information System for Europe*.

[4] e.g. *National Ecosystems Assessment* (UK), *National Environmental Policy Plan* (The Netherlands), *National Strategy on Biodiversity* (France), *National Strategy for the Conservation of Nature and Biodiversity* (Portugal).

processes. Although existing monitoring systems provide a significant and important pool of data on ecosystems' states and dynamics, they often fail to deliver useful information to monitor and to mitigate environmental risks. This results from the fact that they usually focus on impact indicators (e.g. soil erosion, flood area) rather than on indicators of mitigation capacity and/or effectiveness (e.g. soil erosion prevention, flood protection) (Gobin et al. 2004; Dawson et al. 2009).

One particular case is the monitoring of soil erosion effects/impacts (Fig. 1.1a) without a focus on monitoring soil erosion prevention (Fig. 1.1b). While the first is related to the impact itself, and thus dependent on broad scale variables (e.g. climate), the second is a key factor where land management can act to reduce or mitigate the impact that is being generated. This difference of monitoring focus can have a substantial effect in the success of environmental protection measures/policies by providing land managers and policy makers with relevant data to assess and monitor the implications of production and policy goals. In agricultural and forest areas land management determines the land use and consequently the land cover of a particular area (Fig. 1.1). This underlines the importance of the decisions/management choices taken by the land manager, making them the backbone of territorial shaping (Darnhofer et al. 2012). These vary according to a combination of regional, environmental, social, economic and policy dynamics and constrains (Box 1.1 gives a synthesis of the main drivers for Mediterranean Europe).

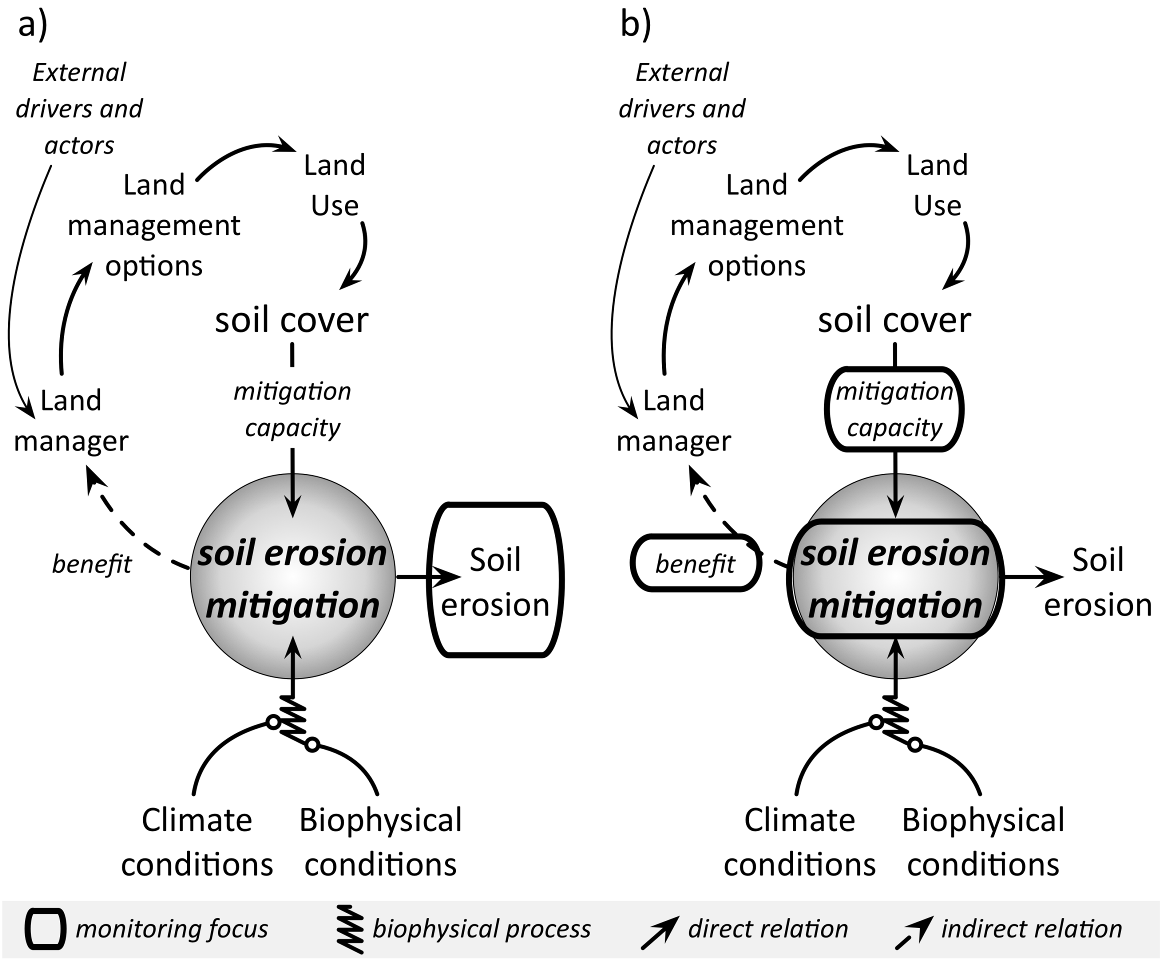

Figure 1.1 Focus of environmental monitoring within the erosion process: a) current environmental risk monitoring; and b) ecosystem service based approach.

Box 1.1

Land management is on of the main pillars of land transformation (Darnhofer et al. 2012). This is particularly relevant in Mediterranean conditions where environmental, social, economic and policy drivers and constrains are extremely present in the decision process of each land manager (Pinto-Correia et al. 2011). In these conditions, land management and the respective land use options are always tested against a fragile equilibrium of scarce environmental resources (Geeson et al. 2002), growing environmental risks (Boardman and Poesen 2006) and pressuring social, economic and policy dynamics (Darnhofer et al. 2012). Considering the contribution of soils to the farm system sustainability (Vanslembrouck and Van Huyenbroeck 2005) and their characteristics in Mediterranean Europe, soil erosion is presently one of the main drivers of soil degradation under these conditions (Van-Camp et al. 2004). At the same time, most cultivated areas are located in areas with sediment accumulated from past erosion and without it the soil would be too thin to allow for an intensive agricultural use (Grove and Rackam 2001). This dichotomy is exacerbated by the current and foreseen climatic trends, regional topography, water availability, but also by the political and social pressures felt by the land manager, configuring a complex web of decisions that ultimately results in very specific and multi-functional land use types.

At a larger scale, land management is therefore responsible for shaping the landscape and for having a significant role in maintaining its resources and ecological functions. In the case of soil resources and processes, soil erosion prevention is rarely the target of management options, mostly oriented to optimize production outputs. This often results in increased impacts mainly caused by land management practices with unsuitable environmental goals. Shifting the focus of environmental monitoring from the impact to the mitigation process (Fig. 1.1) allows to evaluate the implications of land managers' choices and eventually to steer their decision process to a more environmental friendly pathway. Therefore, monitoring the effectiveness of impact mitigation in specific land management systems could then provide important information to assess potential causal relations between land management and the associated impacts resulting from environmental risks. At a different scale, it would also be helpful in policy evaluation and to support future policy design in the context of scenario building assessments.

In order to interpret and assess existing causal relations it is necessary to describe the structure, relations and dynamics between the different social-ecological system elements (Folke et al. 2010). From this perspective, it is not possible to assess the different system elements by themselves, rather it is necessary to focus on the interactions and feedbacks between them in order to determine the relative influence of each element to the final system outcome (Bergengren et al. 2011; Galic et al. 2012). This implies that the focus of the analysis has to be on process related indicators that target the systems' interactions. Consequently it is necessary to introduce new concepts and tools that allow to produce direct information about the vulnerability of these systems and ecological functions.

At the same time, it is becoming more evident for policy makers that nature-based solutions, e.g. using wetlands for water purification, flood protection or carbon storage, may indeed be more cost-effective than implementing other type of technical solutions and infrastructures (Maes et al. 2012). As part of this process, the use of ecosystem service based approaches has transcended the academic sphere to reach the decision making process (Kumar 2010; EC 2011).

Pushing the ecosystem service agenda into policy-making has to ensure that the positive or negative impacts of specific policies on ecosystems and their services are considered during both the policy design and the policy implementation phase (Cowling et al. 2008; Daily et al. 2009). Doing this requires knowledge of the drivers and pressures on the systems under study, as well as an understanding of how the system is changing or might change from its current state (Fisher et al. 2008). Therefore, to monitor and improve the provision of ecosystem services at different scales, it is important to follow a standardized approach, in which clear indicators are identified and spatially explicit methods used to follow and explore current and future changes (Mäler et al. 2008; Tallis et al. 2008). In this context, information must be exchanged not only among diverging scientific disciplines but also between the scientific community and policy and decision-makers (Dunbar et al. 2013). This is also crucial to assess the impacts of policies and whether they need to be re-evaluated in the future (Carpenter et al. 2009; Braat and de Groot 2012).

1.2 The ecosystem service framework

Ecosystem services are the direct and indirect contributions of ecosystems to human well-being (Kumar 2010), corresponding to the outputs of ecological processes that can generate benefits to society. The concept of ecosystem services is attracting increased attention as a way to communicate society's dependence on ecological functions and processes (De Groot et al. 2002). In this context, the ecosystem service concept and general framework (MA 2005a; Carpenter et al. 2006) provide a starting point to generate relevant data for multiple purposes, objectives and scales to support the decision making process (MA 2005b; Tallis et al. 2012).

The origins of the modern history of ecosystem services are to be found in the late 1970s starting with the utilitarian framing of beneficial ecosystem functions as services in order to increase public interest in biodiversity conservation (Gómez-Baggethun et al. 2010). In the 1990s the concept was mainstreamed in the literature (Costanza and Daly 1992; Perrings et al. 1992; Rapport et al. 1998), followed by an increased interest on methods for their assessment and estimation of their economic value (Costanza et al. 1997). The Millennium Assessment (MA 2003) has contributed to great extent to mainstream the ecosystem services concept into the policy agenda, and since its release the literature on ecosystem services has grown exponentially (Fisher et al. 2009; Seppelt et al. 2011; Costanza and Kubiszewski 2012).

The Millennium Assessment (MA 2005a) distinguished provisioning, regulating, cultural and supporting services, and demonstrated that they are subject to natural and human-induced pressures at various spatial and temporal scales. It also provided and tested an ecosystem services framework for analysing social-ecological systems, which influenced policymakers and the scientific community (Maes et al. 2012). Generally speaking, the provision of ecosystem services depends on the local biophysical conditions, land cover, land use, the existing ecological processes and the climatic changes over space and time (Rounsevell et al. 2010; Burkhard et al. 2012). Changes in the state and trends of ecosystem service provision produce a system response that can induce adaptation or mitigation cycles, depending on the influence of the existing external and internal drivers.

In the case of regulating services, service provision depends on the interaction between the biophysical, climatic and land cover/use conditions and dynamics. This specific class of ecosystem services is related to the mitigation of natural hazards (e.g. floods, soil erosion, extreme climatic events) and environmental impacts (e.g. water quality). It includes (Kumar 2010): i) local climate and air quality; ii) carbon sequestration and storage; iii) moderation of extreme events; iv) waste-water treatment; v) soil erosion prevention and maintenance of soil fertility; vi) pollination; and vii) biological control. Due to its relation to specific ecological cycles, ecosystem regulating services are completely dependent on the ecological processes themselves rather than on particular landscape characteristics.

While other ecosystem services (e.g. food production) can have a direct relation to the land cover/use of a given area (i.e. the provisioning capacity is directly related to the actual service provision), in the case of regulating services there appears to be a mismatch between the capacity to provide the service, the actual service provision and the generated benefits to society. A clear example of this mismatch is flood regulation. In this particular case, the actual ecosystem service provision (e.g. water percolation or retention) is not only related to the capacity of a patch of vegetation to provide the service but also to the existence of the impact itself (e.g. water runoff). This means that there can be places where the capacity to provide the service exists but no ecosystem service is provided. Following the same example, the spatial and temporal relation between service provision and the benefit to society is also not clear. While water percolation or retention happens uphill, the benefit (i.e. reduced flooding) happens downstream. The oversimplification of these relations can induce incorrect assumptions and assessments and result in misleading conclusions (Eigenbrod et al. 2010b; Eigenbrod et al. 2010a).

Until recently, most case studies still rely on relatively simplistic approaches using land cover data as a proxy to estimate ecosystem services provision and their values (Schägner et al. 2013). The impact of these proxy-based methods has already been demonstrated for mapping a number of ecosystem services, including regulating services (Eigenbrod et al. 2010b). At the same time, one of the most recurrent limitations of the ecosystem service methodological approach is related to the drawbacks associated with most ways of inferring value (Daily et al. 2000). This stresses the need for novel and interdisciplinary approaches (Tress and Tress 2001; Abson and Termansen 2011; Antrop et al. 2013) that can, at the landscape scale, provide insights about the states and dynamics of ecosystem regulating services.

Since 2005 (MA 2005a; MA 2005b) on-going research has revealed new possibilities for measuring and projecting the effects of policy choices and human activities on the structure and processes of ecosystems, the services they provide, and human well-being (Carpenter et al. 2009). Monetary approaches like cost-benefit analyses, contingent valuations or willingness-to-pay assessments are useful attempts (Farber et al. 2002) but their outcomes are often unsatisfactory due to the economic focus and the lack of appropriate pricing methods, e.g. for non-marketed goods and services (Ludwig 2000; Spangenberg and Settele 2010). More recently the focus of these ecosystem service evaluations has been the mapping of ecosystem services (e.g. Naidoo et al. 2008; Burkhard et al. 2012; Crossman et al. 2013; Maes et al. 2014) in order to provide spatially explicit assessments that can be related to other landscape dynamics. From global (e.g. Naidoo et al. 2008; Costanza et al. 2014) to regional (e.g. Fitter et al. 2010; Maes et al. 2011) or local (e.g. Burkhard et al. 2009; Locatelli et al. 2010) assessments, there are a number of ecosystem services quantification, mapping and valuation methods being applied. Also a considerable volume of information on the state and trends of these services is being generated (Maes et al. 2014; Drakou et al. 2015). In this context, the increased use of the ecosystem services framework creates data integration and methodological challenges particularly in specific groups of ecosystem services like regulating services (Elmqvist and Maltby 2010; Kumar 2010).

1.3 Soil erosion in Mediterranean land use systems

Mediterranean landscapes are characterised by fragile natural ecosystems, long-term human exploitation and insufficient rainfall for fast vegetation recovery after each degrading event (Geeson et al. 2002; Bangash et al. 2013). These conditions have directly influenced the existing land use systems and thus shaped the Mediterranean landscape (Rigueiro-Rodríguez et al. 2009). In these land use systems, anthropogenic disturbances such as deforestation, grazing, agricultural development, and fire management have been factors influencing ecosystem dynamics for at least 10 000 years (Kosmas and Danalatos 1993; Rundel et al. 1998; Grove and Rackam 2001). Although problems of deforestation and erosion date back to Greek and Roman times, many land-use practices were considered relatively sustainable (Geeson et al. 2002). Many of the early forms of agro-pastoral use of Mediterranean woodlands and shrublands continued without significant modification from classic times up until the middle of the twentieth century (Caravello and Giacomin 1993). In contrast, in the past half-century, the frequency and dynamics of land use and land cover change produced dramatic modifications on the structure and diversity of existing ecosystems (Jones et al. 2011b). Strongly supported by the productivist philosophy of the Common Agricultural Policy – after the Second World War – agriculture in Europe has undergone a process of radical transformation through specialization, concentration and the intensification of production. The intensification process resulted in increased mechanization, use of fertilization, and the promotion and extension of irrigated areas (Woods 2005; Woods 2011). This processes occurred later in Mediterranean Europe, and particularly in the Iberian Peninsula, taking place mainly after the integration in 1986 of Portugal and Spain in the European Union (Jones et al. 2011b). In this region the extensive grazing of silvo-pastoral systems has suffered a process of intensification (e.g. with increasing cattle headage), and areas of extensive crop farming have been converted into fields of fast-growing, heavily fertilized and pest-treated crops (Pinto-Correia and Vos 2004; EEA 2007; Emanuelsson et al. 2009).

Considering these dynamics, pressure on Mediterranean soil resources has increased not only due to increasing soil/land demand and land cover change for more intensive land use purposes, but also due to increasing soil degradation risks, particularly soil erosion, desertification, organic matter depletion and contamination (Van-Camp et al. 2004). In the case of the former, by removing the most fertile topsoil, soil erosion reduces productivity and can lead to an irreversible loss of natural farmland and reduce local/regional adaptive capacity (Kirkby et al. 2004). At the same time, erosion rates are expected to rise during the twenty-first century due to global warming, which is related to a more vigorous hydrologic cycle particularly in Mediterranean regions (Zhang and Nearing 2005; Boardman 2006). Under these conditions soil erosion by water is the main process leading to land degradation (Kosmas et al. 2000) and one of the key indicators for long term assessments (Bosco et al. 2014).

Since the 1930s, but mainly after the 1960s, soil scientists and decision-makers have been developing and extensively using models to calculate soil loss from a field, a hill slope, or small watersheds (Wischmeier and Smith 1978). Despite these developments, in the 1970s the problem of land degradation due to soil erosion was considered of minor importance for most of the countries of the European Community. It was pointed out that the traditional agricultural systems were well capable of controlling soil degradation due to erosion (Chisci and Morgan 1986). As a consequence low priority was given to research projects on soil erosion and conservation even if the Mediterranean was considered to be at severe soil erosion risk due to the introduction of new agricultural practices, particularly in mid-slope areas (Morgan 1986). Since the 1980s there was a strong development in soil science with several papers (e.g. Vrieling 2006; Verheijen et al. 2009; Dunbar et al. 2013; Panagos et al. 2014a), books (e.g. Chisci and Morgan 1986; Schwertmann et al. 1988; Boardman and Poesen 2006) and reports (e.g. Coelho et al. 1995; Van Rompaey et al. 2003; Kirkby et al. 2004) being published. These publications highlight the potential problems resulting from soil loss caused by water in agricultural and forest landscapes and identify past and current research opportunities.

Soil erosion caused by water is the result of many processes, which influence each other through complex interactions and it proceeds at rates that vary in space and time (Driesen 1986). It corresponds to a multivariate phenomenon that results from the interaction and combination of (Panagos et al. 2011b) (Fig. 1.2): i) **rainfall erosivity**, which accounts for the erosive potential directly caused by rainfall intensity (Wischmeier and Smith 1978); ii) **soil erodibility**, which reflects the detachability (a function of soil infiltration rate, permeability, and total water capacity) and transportability (a function of soil dispersion, splashing, and abrasion) of a specific soil type (Wawer et al. 2005; Panagos et al. 2012); iii) **topography**, which affects the volume and velocity of water flow resulting from a complex interaction of several terrain parameters, such as the angle (or degree) of slope, slope length, curvature, and roughness of the surface (Desmet and Govers 1996; Van Remortel et al. 2004); iv) **land use**, which is associated with the anthropogenic acceleration of erosion discriminating actual from potential erosion (Schuler and Sattler 2010; Panagos et al. 2015); and finally v) **land cover**, particularly vegetation cover that prevents the process of soil detachment from rainfall and runoff (Wang et al. 2008; Zhongming et al. 2010).

The rate of soil degradation is dependent upon the rate of vegetation degradation, which in turn is influenced by both adverse climate (Langbein and Schumm 1958; Wang et al. 2008), biological invasions (Charles and Dukes 2007; Vicente et al. 2013a; Fernandes et al. 2014) and land use and management changes (Kosmas et al. 1997; Geeson et al. 2002). At the level of ecosystem service provision, vegetation is therefore the main ecosystem service provider related to soil erosion prevention (Faber and van Wensem 2012; Galic et al. 2012). Hence, in the framework of this dissertation, soil erosion prevention corresponds to the ecological process by which soil erosion is mitigated by vegetation cover in a given place and time.

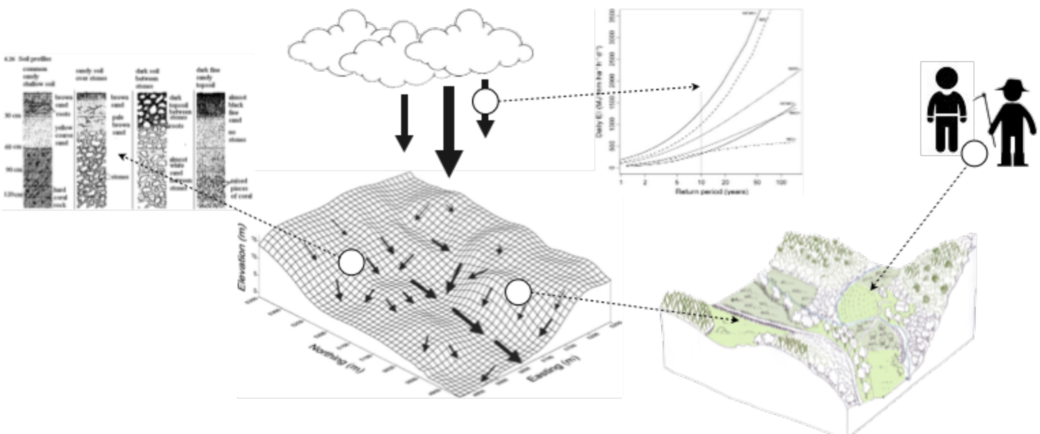

Figure 1.2 Main variables and interactions within the soil erosion process by water.

As highlighted before, land management plays a central role in maintaining and promoting vegetation cover, particularly in periods of heavy rain following a dry season (Giráldez et al. 1988). Management options depend on farm biophysical and structural characteristics and production goals, but are also affected by the individual profile of the land manager, his social context and by a set of external drivers

(Darnhofer et al. 2010; Sutherland et al. 2014; Darnhofer 2014). These external drivers are related to global and regional markets and technological developments, but in todays' Europe are strongly dependent on sectorial policies (e.g. the European Common Agricultural Policy) (Darnhofer et al. 2012; van Vliet et al. 2015). Due to these drivers (both internal and external) and to the local adjustments made according to the biophysical and climatic characteristics of each place, land management can vary through time and space (Almeida et al. 2013). This makes the evaluation of land management practices a significant challenge, especially in the context of monitoring their environmental impacts.

In order to use ecosystem service provision as a potential indicator to monitor the impacts of land management and policy implementation, the methodological approach has to be able to assess and show variations in space and time (Syrbe and Walz 2012), and also to be used across scales, i.e. allowing for upscaling and downscaling exercises with a high degree of confidence. This would allow to capture, at a representative scale of the process and/or of the driver, the associated impacts (positive and negative) (Schröter et al. 2005; Metzger et al. 2006). In the particular case of soil erosion prevention (and theoretically of all regulating services) this approach assumes an even more critical importance in social-ecological systems with fragile equilibriums like the land use systems present in the Mediterranean area (Fürst et al. 2013; Almagro et al. 2013). Its monitoring has to consider the different components of its particular ecological cycle and, at the same time, it has to be represented at a relevant spatial and temporal scale.

1.4 Research objectives and thesis outline

1.4.1 Research objectives

This thesis presents a novel assessment framework for assessing the spatial distribution of ecosystem regulating services provision according to a process-based approach that considers specific biophysical and climatic traits and different land use trajectories. The framework is sensitive to environmental and/or social-economic changes and can be adapted to different social-ecological contexts and different evaluation scales.

Overarching objective: To develop and test an ecosystem service assessment framework that provides a complementary approach to address modern conservation challenges in land management and policy design.

The framework will be demonstrated for soil erosion prevention with focus on the spatial and temporal mismatch between the evaluation of ecosystem service capacity and the evaluation of actual ecosystem service provision. At the same time the thesis describes the development of an indicator set that allows to monitor the effects of policy and land management dynamics on ecosystem service provision. Specifically, the Thesis will address the three research questions described below.

RQ 1: What are the impacts of historic and current policy implementation in the provision of soil erosion prevention?

Specific objectives:

- *Demonstrate the implications of agricultural policy implementation and change for soil erosion prevention provision in a particular land management system.*
- *Discuss the challenges for policy design deriving from the inclusion of ecosystem service based indicators in strategic impact evaluations.*

RQ 2: Can land management practices be monitored using ecosystem service indicators to determine the degree to which they are improving or hampering ecosystem service provision?

Specific objectives:

- *Test if ecosystem service indicators are sensitive to different land management practices and if specific land management practices reflect different soil erosion prevention considering their specific biophysical, climatic and topographic conditions.*
- *Discuss the application of using ecosystem service provision indicators to compare between land management practices in terms of environmental conservation outputs.*

RQ 3: Can the developed framework be up-scaled to produce relevant results at macro and sub-regional scales and what are the key challenges?

Specific objectives:

- *Describe the recent trends of soil erosion prevention in Mediterranean Europe.*
- *Use ecosystem service indicators in the frame of a strategic evaluation and test the potential to identify and typify vulnerable areas of ecosystem service provision.*
- *Discuss the potential of ecosystem services broad scale assessment for policy support.*

The research presented in this dissertation provides a common conceptual and modeling framework to quantify and map ecosystem regulating services. The application examples included demonstrate how the framework can help to increase and improve the available information and data for decision-makers and contribute to the understanding of the implications of complex policies. It further demonstrated the value of ecosystem service indicators to improve land management monitoring and decision support.

1.4.2 Thesis outline

The conceptual framework for quantifying and mapping ecosystem services provision is described in detail in Chapter 2 and sits at the core of the dissertation (Fig. 1.3). The validity and applicability of the framework is the described using three applied case studies, testing the framework at three different spatial scales and across a range of social and environmental conditions within the Mediterranean area (Chapters 3-5; Fig. 1.3).

Chapter 2 presents a novel ecosystem service provision assessment framework that was designed to focus on the "avoided impact" principle related to the provision of ecosystem regulating services. It assumes that ecosystem service is provided in the interface between the structural and social-ecological components of the system under evaluation, and it lays out a set of indicators to assess the distribution, magnitude and dynamics of ecosystem service provision. This framework resides at the core of this Dissertation from which opportunities for policy and management support and evaluation are recognized.

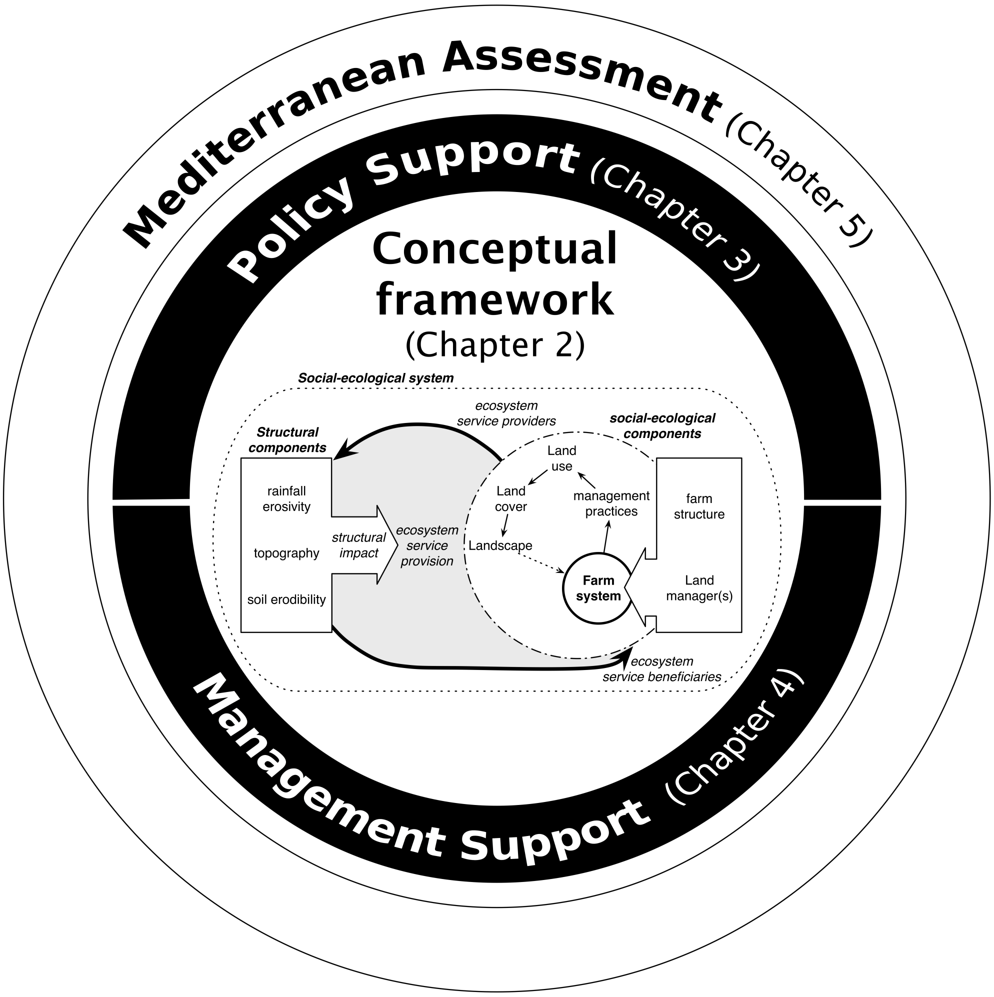

Figure 1.3 Outline of the Dissertation describing the research objectives and scales of application of each case study.

In Chapter 3 the sensitivity of the proposed framework to identify the influence of policy implementation is assessed. It includes a local example where policy measures related to soil prevention are described for the last 60 years and are related to the dynamics of soil erosion prevention in the area. This represents a key aspect to increase the range of applications of the proposed framework as well as to improve its relevance for environmental monitoring systems.

In the following assessment (Chapter 4), the influence of management options in the provision of soil erosion prevention is also evaluated. To do this a group of land management units was surveyed and relations were inferred regarding the estimated ecosystem service provision indicators.

These last two Chapters present and discuss the application opportunities of the proposed ecosystem service evaluation framework. They also provide evidence of the sensitivity of the proposed ecosystem service indicators of soil erosion prevention to changes in policy and management.

Chapter 5 illustrates how the proposed ecosystem service framework can be upscaled to the Mediterranean Europe macro region. Here, maps of the provision of soil erosion prevention are provided and discussed and regional ecosystem service profiles are used to compare between regions in an illustrative regional assessment.

Finally Chapter 6 gives a general discussion of the proposed ecosystem service evaluation framework and of the results from each assessment. Policy and management support and monitoring applications are described and future work is anticipated.

2

MAPPING SOIL EROSION PREVENTION USING AN ECOSYSTEM SERVICE MODELING FRAMEWORK FOR INTEGRATED LAND MANAGEMENT AND POLICY

Based on the manuscript:
Guerra, C.A., Pinto-Correia, T., Metzger, M.J., 2014. Mapping Soil Erosion Prevention Using an Ecosystem Service Modeling Framework for Integrated Land Management and Policy. Ecosystems 17, 878–889.

Abstract

Current spatially explicit approaches to map and assess ecosystem services are often grounded on unreliable proxy data based on land use/cover to derive ecosystem service indicators. These approaches fail to make a distinction between the actual service provision and the underlying ecosystem capacity to provide the service. We present the application of an integrative conceptual framework to estimate the provision of soil erosion prevention by combining the structural impact of soil erosion and the social-ecological processes that allow for its mitigation. The framework was tested and illustrated in the Portel municipality in Southern Portugal, a Mediterranean silvo-pastoral system that is prone to desertification and soil degradation. The results show a clear difference in the spatial and temporal distribution of the capacity for ecosystem service provision and the actual ecosystem service provision. It also shows that although the average actual ecosystem service provision in the region is sufficient to mitigate the existing structural impact, vulnerable areas can be identified where significant soil losses are not mitigated at present. This becomes more significant when comparing different land management intensities. Considering these results, we argue that the general assumption that there is an almost direct relation between the capacity for ecosystem service provision of a given area and the actual ecosystem service provision is wrong. We also discuss how the framework presented here could be used to support land management and policy, and how it can be adapted for other regulating services.

2.1 Introduction

The Millennium Assessment (MA 2005a) provided and tested an ecosystem services framework for analysing social-ecological systems, which influenced policymakers and the scientific community (Maes et al. 2012). It distinguished provisioning, regulating, cultural and supporting services, and demonstrated that they are subject to natural and human-induced pressures at various spatial and temporal scales. Despite its limitations (Wallace 2007; Kumar 2010) the ecosystem services concept provides an important framework for understanding and managing complex social-ecological processes at both local and regional scales (Williams and Kapustka 2000; Holling 2001; Daily and Matson 2008; Daily et al. 2009; Hauck et al. 2013).

Following the inclusion of ecosystem services in the CBD Aichi targets (CBD 2012; Pereira et al. 2013), the concept is now increasingly used in policy and decision-making, e.g. in the EU biodiversity strategy (EC 2011). For the appropriate design of policy tools and to target their implementation and allow for their monitoring, there is an increasing need for spatially explicit information on the state and trends of ecosystems and their services (Maes et al. 2012). Spatial and temporal representation of ecosystem services allows the identification of areas with particular importance in terms of the services provided, as well as vulnerable areas, and the possibility to explore consequences of current and future environmental and socioeconomic change (Müller and Burkhard 2012; Rounsevell et al. 2012). Therefore, research efforts are now underway to quantify, value and map ecosystem services at all scales, ranging from local (e.g. Nedkov and Burkhard 2011; Plieninger et al. 2013) regional (Bangash et al. 2013; Bagstad et al. 2013) national (e.g. van Wijnen et al. 2012) and international scales (e.g. Metzger et al. 2008b; Naidoo et al. 2008).

Current spatially explicit approaches are mostly based on land use/cover assessments (Vihervaara et al. 2010; Burkhard et al. 2012), assigning a static value, often based in expert opinion, to the capacity of a given land cover class to provide an ecosystem service (e.g. Burkhard et al. 2009). Although insightful, especially for national or international assessments, these maps form a major simplification of inherently multi-dimensional processes. They also provide limited support for developing regional land management strategies and policy (Eigenbrod et al. 2010b; Seppelt et al. 2011), which require information on both the *actual ecosystem services provision* and the *capacity for ecosystem services provision*.

These two concepts, *actual ecosystem services provision* and the *capacity for ecosystem services provision*, express different ecosystem components within a single land cover class, which often do not overlap in space and time. The later relates to the capacity of a given land cover type to provide soil protection (e.g. associated to the density of the canopy), while the first relates to the actual quantity of soil not eroded in a given environmental and biophysical context. For example, in the Mediterranean, maximum vegetation cover within a land cover class varies through the year, and consequently the *actual ecosystem services provision* of soil erosion prevention will show temporal differences. However, current mapping approaches (e.g. Burkhard et al. 2012; Haines-Young et al. 2012; Vicente et al. 2013b), focusing on the *capacity for ecosystem services provision,* only provide a static figure and do not consider the variation of environmental or biophysical traits. Although the latter is sufficient for broad overviews, the effectiveness of land management (e.g. to mitigate soil erosion) and policy incentives can only be measured if we can assess whether adequate vegetation cover is in place during critical periods, requiring a spatially and temporally explicit assessment of the *actual ecosystem service provision.*

This second Chapter presents a modelling framework to assess *actual ecosystem services provision,* illustrated for soil erosion prevention, a regulating service (Fu et al. 2011) that varies over time and space, especially in regions with high risk of desertification and soil degradation (Van-Camp et al. 2004). Soil erosion has both local (e.g. Vanwalleghem et al. 2010) and regional impacts (e.g. Martín-Fernández and Martínez-Núñez 2011), which can often be mitigated by adopting suitable land management practices (Presbitero et al. 1995). Although there are several papers describing the spatial distribution of soil erosion (e.g. Bou Kheir et al. 2006), there are important knowledge gaps regarding the spatial and temporal extent and distribution of soil erosion prevention.

Previous studies mapping soil erosion prevention used static approaches to determine the *capacity for ecosystem services provision* (e.g. Maes et al. 2011; van Wijnen et al. 2012), ignoring interactions between the structural ecosystem components such as topography, soil type and rainfall, and the social-ecological processes affecting vegetation cover (Fig. 2.1). Drivers affecting the social-ecological system can be multiple, including changes in commodity prices and consumer demand, changes in policy, and changes in land managers' preferences and priorities (Pinto-Correia and Vos 2004; Rounsevell et al. 2012). Combined, they influence land management decisions at the farm level, including livestock breeds, stocking density of livestock, grazing pressure, and shrub control. The social-ecological drivers therefore ultimately determine the overall landscape (Swaffield and Primdahl 2010; Benoît et al. 2012), within which farmers manage their land. For those reasons we argue that to understand, measure and map *actual ecosystem services provision*, both the structural and the social-ecological components of the social-ecological system need to be considered.

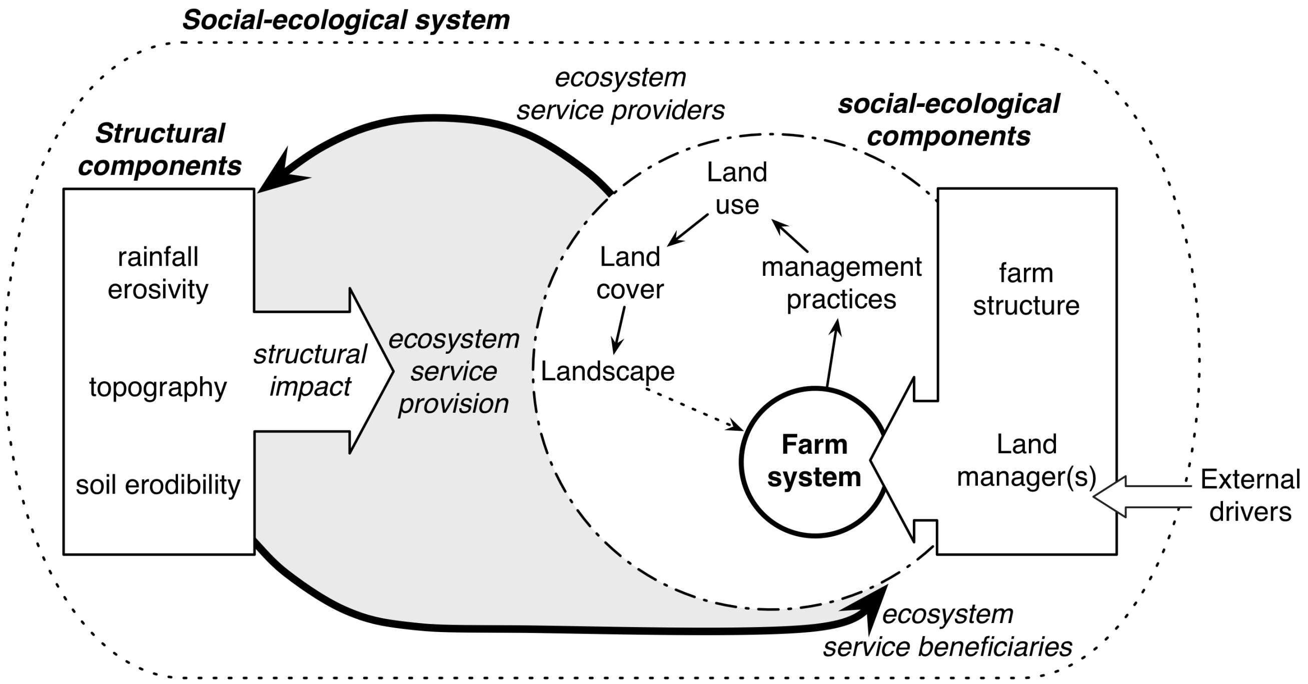

Figure 2.1 Conceptual framework of ecosystem service provision, as dependent on both the biophysical structure and land management, driven by a combination of multiple structural and social-ecological components.

The following section presents a conceptual framework for assessing *actual ecosystem services provision* of soil erosion prevention, incorporating the structural and social-ecological components of a given system. This framework is subsequently illustrated for a silvo-pastoral case study in Portugal (Section 2.3). Finally the results (Section 2.4) demonstrate how the presented approach can:

i) assess the impact of land management practices on *actual ecosystem services provision*;

ii) identify spatial and temporal trends in *actual ecosystem services provision*;

iii) identify vulnerability hot spots; and

v) support the development of sustainable land use policy.

2.2 Conceptual ecosystem service framework for assessing soil erosion prevention

To begin assessing the contribution of soil erosion prevention we need to identify the *structural impact* (Υ) of soil erosion, i.e. the erosion that would occur when vegetation is absent and therefore no ecosystem service is provided (Fig. 2.2a). It determines the potential soil erosion in a given place and time and is related to rainfall erosivity (i.e. the erosive potential of rainfall), soil erodibility (as a characteristic of the soil type) and local topography (Panagos et al. 2011b). Although external drivers can have an effect on these variables (e.g. climate change), they are less prone to be changed directly by human action.

The *actual ecosystem service provision* (*Es*) reduces the total amount of *structural impact* (Υ), and we define the remaining impact as the *ecosystem service mitigated impact (* β_e). We can then define the *capacity for ecosystem service provision (*e_s*)* as a key component to determine the fraction of the *structural impact* that is mitigated (Fig. 2.2b). This *capacity for ecosystem service provision (*e_s) is influenced by both internal and external socio-ecological drivers. Examples of internal drivers include management options, forest fires, and urban sprawl, whilst agricultural policy measures, spatial planning, and climate change are examples of external drivers affecting soil erosion prevention.

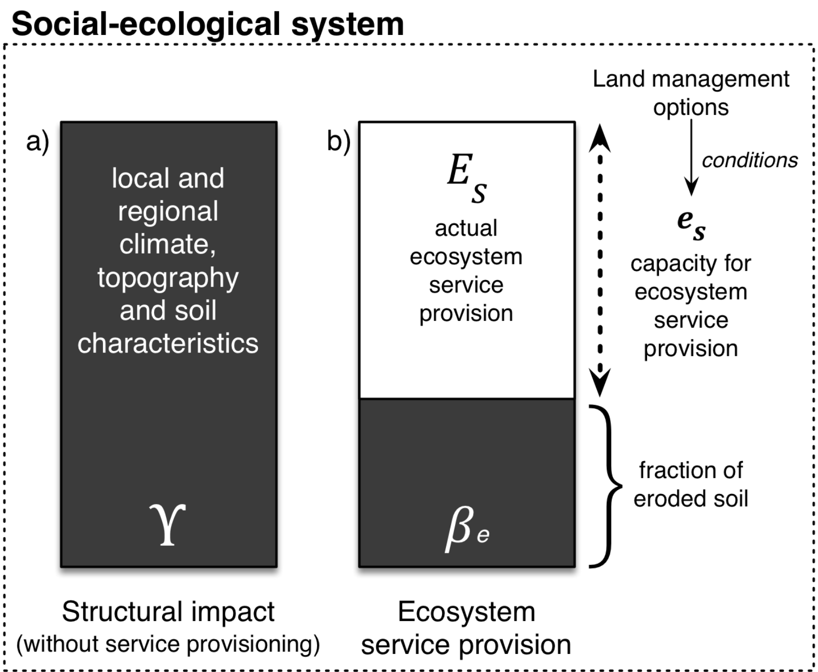

Figure 2.2 Conceptual framework where: a) presents the *structural impact* (Υ), i.e. the total soil erosion without ecosystem service provision (here vegetation cover); and b) distinguishes the *actual ecosystem service provision* (E_s), as a fraction of the structural impact and determined by the capacity for ecosystem service provision, and the remaining *ecosystem service mitigated impact* (β_e).

The four concepts, described in Table 2.1, can be used as the basis for assessing the impacts of land use practices on ecosystem service provision, and for the evaluation of land management and policy measures aimed at increasing soil erosion prevention. The challenge is to clearly identify and define the relevant processes within the social-ecological system (e.g. the soil erosion prevention provided by vegetation cover), which must then be mathematically formulated.

Table 2.1 Identification and description of the four concepts used in the proposed ecosystem service vulnerability framework.

Concept	**Variable**	**Description**
Structural impact	Υ	The total soil erosion impact when no ecosystem service is provided.
Capacity for ecosystem service provision	e_s	The fraction of the structural impact that is mitigated by the ecosystem service provider, it corresponds to an adimensional gradient ranging from 0 to 1.
Actual ecosystem service provision	E_s	The *actual ecosystem service provision* corresponds to the total amount of ecosystem service provided, measured in ecosystem service providing units (tons of soil not eroded). It varies from season to season and year-to-year depending on the variation of the structural impact.
Ecosystem service mitigated impact	β_e	The remaining soil erosion after ecosystem service provision.

Here, we provide a simple mathematical outline, which could be further elaborated depending on system knowledge and data availability. It is based on the premise that the *actual ecosystem service provision* is determined by the difference between the *structural impact* being evaluated and the *ecosystem service mitigated impact*:

$$E_s = \Upsilon - \beta_e \quad \text{(Eq.2.1)}$$

where, E_s corresponds to the *actual ecosystem service provision*, Υ to the *structural impact* from soil erosion, and β_e to the *ecosystem service mitigated impact*. This provides a simple conceptual background to the formulation of a more complex mathematical model.

In the case of soil erosion, the *structural impact* can be measured as a function of a set of key structural climate and physiographic components, given by:

$$\Upsilon = f(\eta) \quad \text{(Eq.2.2)}$$

where, Υ corresponds to the *structural impact* from soil erosion, and $f(\eta)$ to a function of the previously defined soil erosion variables considered as fundamental to determine a structural soil erosion (e.g. rainfall erosivity, soil erodibility, topography).

While structural soil erosion considers the main climate and physiographic components of soil erosion, to determine the *ecosystem service mitigated impact* it is necessary to study the mitigation capacity of the ecosystem service, i.e. the ecosystem capacity to provide a specific service (here soil erosion prevention). Mathematically, the *ecosystem service mitigated impact* is defined as:

$$\beta_e = \Upsilon \times \alpha \tag{Eq.2.3}$$

where, β_e corresponds to the *ecosystem service mitigated impact*, Υ to the *structural impact* from soil erosion, and a to the inverse gradient of the *capacity for ecosystem service provision* (e_s) measured between 0 and 1 and obtained as an estimation of the ecosystem functions and processes related to the ecosystem service provision, given that $\alpha = 1 - e_s$ and:

$$\begin{cases} E_s = \Upsilon \; if \; e_s = 1 \\ E_s = 0 \; if \; e_s = 0 \end{cases} \tag{Eq.2.4}$$

This mathematical quantification of *actual ecosystem service provision* can be related to specific land management practices by assessing ecosystem service provision in space and time. By describing the spatial and temporal distribution of ecosystem service provision it is possible to: i) compare two different land management strategies and assess their effectiveness in improving the ecosystem service provisioning; ii) evaluate the same management strategy with different impact intensities in order to identify acceptable impact thresholds; and iii) assess the temporal effect of land management strategies over specific ecosystem services or functions.

The analysis of land management impacts can then be implemented within a specific spatial or temporal context given the following relation:

$$\begin{cases} \Delta_{(LM_A - LM_B)} \rightarrow \Delta_{\left(E_{S(A)} - E_{S(B)}\right)} \\ \Delta_{t\,(LM_A - LM_A)} \rightarrow \Delta_{t\,\left(E_{S(A)} - E_{S(A)}\right)} \end{cases} \tag{Eq.2.5}$$

where, $\Delta_{(LM_A - LM_B)}$ corresponds to the variation in land management practices between evaluation sites A and B, $\Delta_{\left(E_{S(A)} - E_{S(B)}\right)}$ and $\Delta_{t\,\left(E_{S(A)} - E_{S(A)}\right)}$ to the variation of the *actual ecosystem service provision* resulting from each of the land management practices, and $\Delta_{t\,(LM_A - LM_A)}$ to the temporal variation between land management practices in site A. The first expression considers time invariant conditions as it proposes the evaluation of different sites with potentially variable environmental and management conditions, while the second expression considers space invariant conditions. This means that the latter considers the evaluation of the ecosystem service trends of a single region. These two expressions combined make it possible to identify the trajectories of ecosystem service provision in different spatial regions (i.e. a pixel or a county depending on the scale of the proposed evaluation) and to consider the comparison between different social-ecological systems.

2.3 Application of the framework in a case study in Portugal

2.3.1 Case study description

Soil erosion is one of the major environmental challenges for Mediterranean Europe (Van-Camp et al. 2004). Warm and dry summer are followed by periods of concentrated precipitation in autumn and spring, which easily cause erosion of the shallow poor soils, especially in areas with high grazing pressure, such as the montado in southern Portugal (Pinto-Correia et al. 2011). This silvo-pastoral system is an open savannah-like forest of cork (*Quercus suber*) and holm (*Quercus ilex*) oaks, with trees of different ages randomly dispersed in changing densities, and pastures in the under cover. The latter are mostly natural pastures in a mosaic with patches of shrubs, which differ in size and distribution and depend mainly on the grazing intensity.

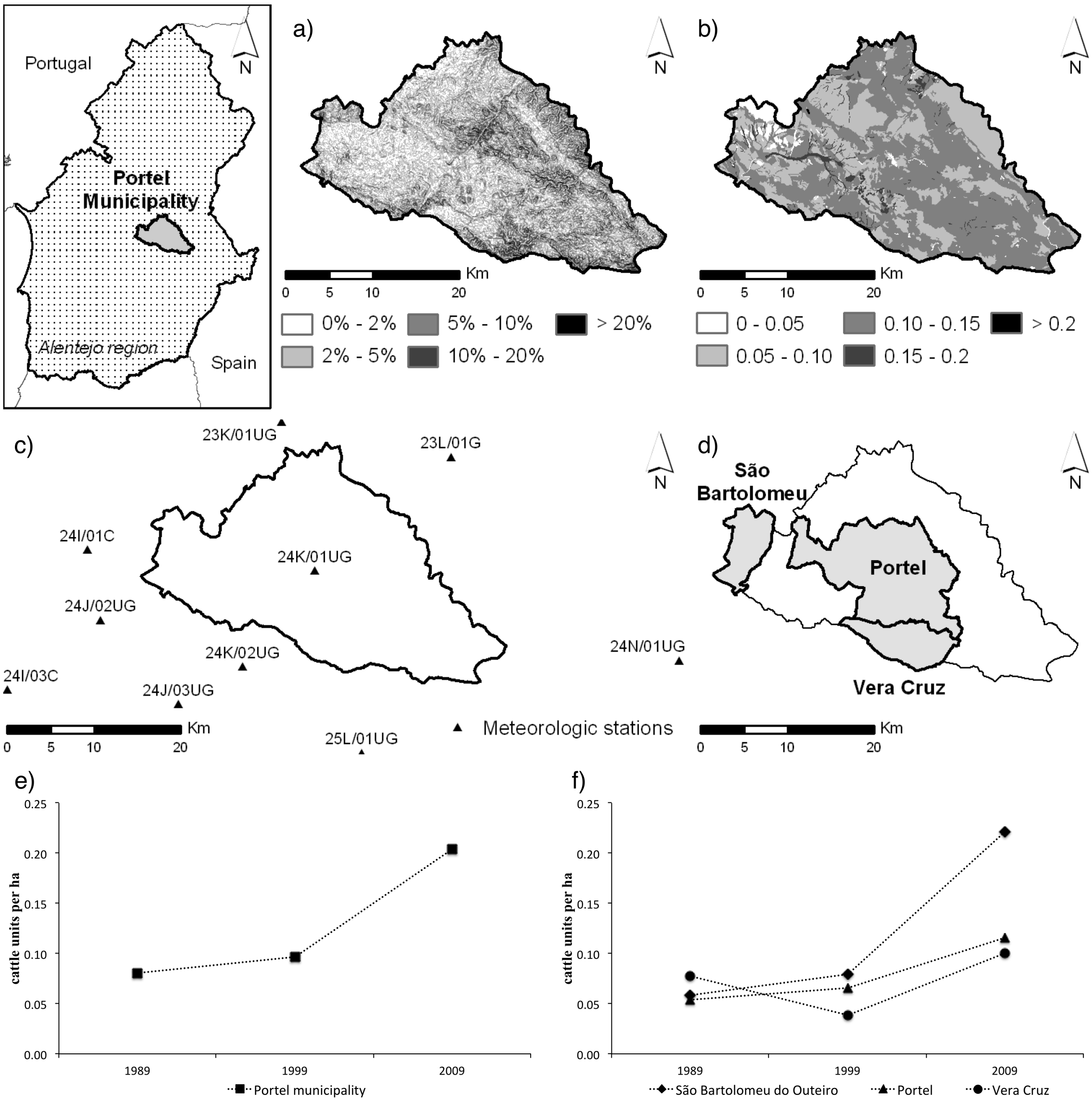

Figure 2.3 Spatial location of the selected case study and the distribution of the: a) identified slope classes; b) soil erodibility values (obtained from the application of the mathematical equation in section 2.3.2 with data collected from 98 soil profiles of the 25k soil map national series (Cardoso 1965)); c) meteorological stations used in the modelling process (obtained from the Portuguese national water resources information system); d) selected civil parishes used in section 2.3.3 to assess the influence of land management practices in ecosystem service provisioning; e) variation in cattle breed intensity (INE 2011) for the Portel municipality; and f) variation in cattle breed intensity (INE 2011) for the three selected civil parishes ("Portel", "Vera Cruz" and "São Bartolomeu").

In recent years, and despite its acknowledged natural and cultural values (Surová et al. 2013), the montado system has been threatened by a constant land use intensification (Pereira and Fonseca 2003; Almeida et al. 2013). The total area of Montado is under decay, and this is mostly due to a gradual degradation of the soil, pastures and tree stands (Pinto-Correia and Godinho 2013).

Although there are several land management factors that result in land cover changes (e.g. livestock breed, frequency of shrub mechanical control, and soil mobilization techniques) cattle density is currently the primary reason (Pinto-Correia and Mascarenhas 1999; Pinto-Correia et al. 2011). The number of grazing animals has increased along with a trend of replacing sheep by cattle in response to Common Agricultural Policy livestock payments (stopped in 2012) that were substantially higher for cattle than for sheep (Pinto-Correia et al. 2011; Pinto-Correia and Godinho 2013). Having more than 90% of montado cover coupled with the increased agricultural intensification (Fig. 2.3e), which coincided with decreased vegetation cover and high soil erodibility, makes the Potel Municipality (Fig. 2.3) an appropriate case study to test our framework.

2.3.2 Mathematical outline

Following the conceptual outline (section 2.2), we will estimate the soil erosion prevention provided by vegetation cover using an adaptation of the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978). Within USLE, soil erosion is represented by a set of critical factors given by (Panagos et al. 2011b):

$$A = R \times LS \times K \times C \times P \qquad \text{(Eq.2.6)}$$

where, A represents the amount of soil loss, R the rainfall runoff factor, LS the topographic factor, K the soil erodibility factor, C the vegetation cover factor and P the conservation practices factor.

An adaptation of this general expression was made to convey the outputs of the conceptual model. In this context, the structural soil erosion (Υ) was calculated using the expression $\Upsilon = R \times LS \times K$, and the gradient of ecosystem service mitigated soil erosion (β_e) will be determined by $\beta_e = \Upsilon \times \alpha$ (where $\alpha = C$ and $e_s = 1 - \alpha$). These two expressions will allow determining the *actual ecosystem service provision* (E_s). Although it will not allow to obtain an absolute measure of soil erosion, this mathematical formulation defines a spatially explicit gradient of potential soil loss and the correspondent gradient of ecosystem service provided by vegetation cover (E_s). To estimate each of the system components the model will be parameterized according to the mathematical equations described in Table 2.2.

The rainfall erosivity factor (see Table 2.2) was assessed based on the MedREM model proposed by Diodato and Bellocchi (2010) for Mediterranean conditions and a spatial interpolation of available daily rainfall data for the period between January and December of 2003, obtained from 16 meteorological stations of the Portuguese national water resources information system (INAG 2010) (Fig. 2.3a). Afterwards an inverse distance weighted (IDW) interpolation algorithm was applied to obtain a monthly rainfall erosivity surface.

Table 2.2 Soil erosion critical factors and corresponding mathematical expressions and variables.

Critical factor	Mathematical expression	Variables	Main references
Rainfall erosivity	$R_m = b_0 \times P_m \times \sqrt{d_m} \times (a + b_1 \times L)$	R_m ($MJ.mm.ha^{-1}.h^{-1}.month^{-1}$) = monthly erosivity factor for the month m b_0 ($MJ.ha^{-1}.h^{-1}$) = 0.117 b_1 ($d^{0.5}.mm^{-0.50-1}$) = -0.015 a ($d^{0.5}.mm^{-0.50}$) = 2 L (°) = site longitude P_m (mm) = total amount of precipitation in a given month m d_m (mm.d-1) = monthly maximum daily precipitation for month m	(Diodato and Bellocchi 2010)
Soil erodibility	$K = \frac{2.1M^{1.14}10^{-4}(12-a)+3.25(b-2)+2.5(c-3)}{100 \times 7.59}$	K = soil erodibility factor a = % of organic matter b = soil structure parameter c = profile permeability class $M = (\%_{silt} + \%_{very\ fine\ sand}) \times (100 - \%_{sand})$	(Wischmeier and Smith 1978; Morgan 2005; Prasannakumar et al. 2012)
Topography	$LS = \left(\frac{a \times p}{22.13}\right)^{0.4} \times \left(\frac{\sin(d)}{0.0896}\right)^{1.3}$	LS = topographic factor a = flow accumulation model p = pixel size of the DEM d = slope model in degrees	(Moore and Burch 1986)
Soil cover	$C = \exp\left[-a \times \frac{NDVI}{(b - NDVI)}\right]$	$a = 2$ $b = 1$	(Van der Knijff et al. 1999; Van der Knijff et al. 2000; Prasannakumar et al. 2012)

Vegetation cover was estimated using the relation between NDVI (calculated from MODIS 250 meters pixel images) and the USLE C Factor proposed by Van der Knijff et al. (1999, 2000). A monthly average of the NDVI (e.g. Purevdorj et al., 1998; Kouli et al., 2008) was calculated from collection 5 MODIS images (250 meters pixel) (Fensholt and Proud 2012; Fritsch et al. 2012). Both the topographic factor and the soil erodibility factor were calculated using previously existing datasets from the Portuguese national geographic information system. All outputs were provided at a pixel resolution of 250 meters that we consider sufficient for illustration purposes within these extensive silvo-pastoral systems.

2.3.3 Land management

Appropriate land management is one of the most important factors in preventing soil erosion (Burger and Kelting 1999; Herrick 2000; Carter 2002). Almeida et al. (2013) have showed that in montado areas, the grazing pressure on vegetation increases as the summer progresses, reducing, according to grazing intensity, the capacity for soil erosion prevention and therefore increasing the difficulty of re-establishing significant levels of soil protection. Assessing the impacts of land management practices on soil erosion prevention can help define and evaluate sustainable land management strategies.

To illustrate the potential of the proposed framework, a comparison was made between three civil parishes in the Portel municipality with different livestock densities (Fig. 2.3f). For these three civil parishes (Portel, São Bartolomeu and Vera Cruz) all ecosystem service indicators were calculated and their temporal variation was evaluated and compared. Using this procedure, management thresholds could eventually be identified by assessing the spatial and temporal variation in the amount of *actual ecosystem service provision* between the three civil parishes.

2.4 Results

2.4.1 Ecosystem service assessment

Monthly estimates of *actual ecosystem service provision* were calculated for the entire study area between January and December of 2003. The results (Fig. 2.4a) show a marked difference between the temporal variation of the *capacity for ecosystem service provision* and the *actual ecosystem service provision* (Fig. 2.4b). While the *capacity for ecosystem service provision* is influenced by the seasonal variations in vegetation growth, and is therefore lower in the dry Mediterranean summer months, the *actual ecosystem service provision* corresponds to the interaction between vegetation and the *structural impact* (Fig. 2.4a).

The results show the significant influence of vegetation in the *actual ecosystem service provision*, causing an important reduction of the existing structural impact to more acceptable impact levels (i.e. *ecosystem service mitigated impact*) (Fig. 2.4a). The highest *actual ecosystem service provision* (i.e. October) does not correspond to the period of highest *capacity for ecosystem service provision* (January to April). In sensitive Mediterranean agricultural and silvo-pastoral systems, which are highly water dependent in the summer and have important decays in soil cover through the season (Fig. 2.4b), this mismatch is particularly important as it can drastically increase the amount of eroded soil in a given area. Also, the projected climate change will result in a decrease of the total amount of precipitation but with an increase of heavy rain periods in late summer and in the beginning of autumn. This will increase water dependency and soil cover decay, decreasing the *capacity for ecosystem service provision* and result in higher levels of impact related to soil erosion.

For these types of social-ecological systems, our results clearly reject the assumption that there is a straightforward relation between the *capacity for ecosystem service provision* of a given area, often obtained from land cover maps (cf. Burkhard et al. 2012), and the *actual ecosystem service provision*.

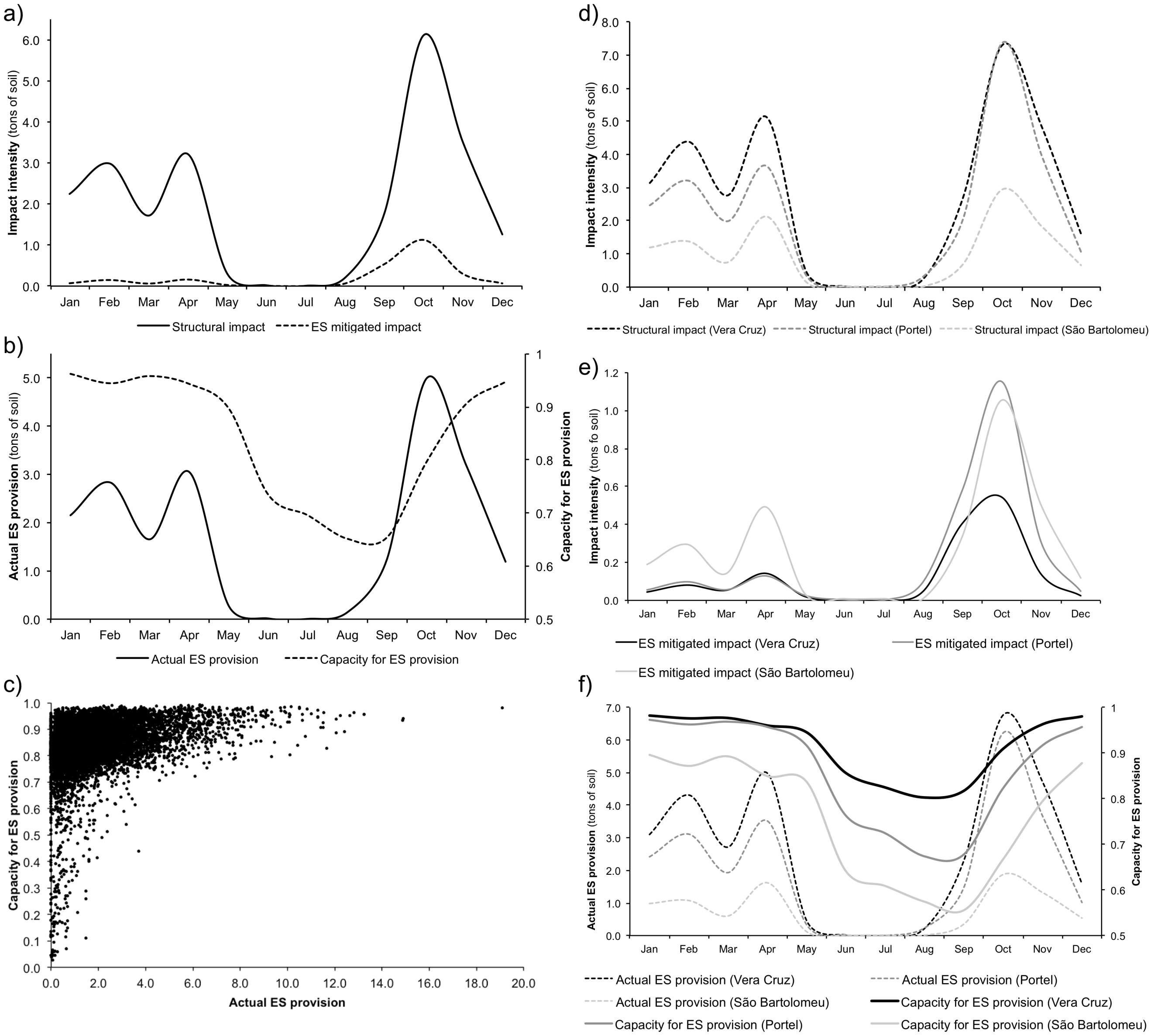

Figure 2.4 Illustration for the Portel municipality of (with exception of Fig. 4c all values represent monthly values for the entire study area): a) temporal variation of the *structural impact* and the *ecosystem service mitigated impact* (average values for the municipality); b) temporal variation of the *actual ecosystem service provision* and the *capacity for ecosystem service provision* (average values for the municipality); and c) relation between the average *actual ecosystem service provision* and the average *capacity for ecosystem service provision* (each point represents a pixel value); and for the three selected civil parishes of: d) temporal variation of the modeled *structural impact* (average values for each civil parish); e) temporal variation of the ecosystem service *mitigated impact* (average values for each civil parish); and f) comparison between the average *actual ecosystem service provision* and the average *capacity for ecosystem service provision*.

In fact, they show that there is a significant mix between areas with high *capacity for ecosystem service provision* with rather low *actual ecosystem service provision* (Fig. 2.4c). This is especially relevant when we consider the spatial distribution of both system components (Fig. 2.5a and Fig. 2.5b) and their temporal variation (Fig. 2.4b). In this case, it is clear that although the *capacity for ecosystem service provision* shows seasonal variation, the *actual ecosystem service provision* only occurs when the climatic and biophysical conditions are favourable to the occurrence of a *structural impact*. This implies that with no rainfall (like in the period between June and July) there is no ecosystem service provision, although there is a latent *capacity for ecosystem service provision* that allows to prevent potential peak situations.

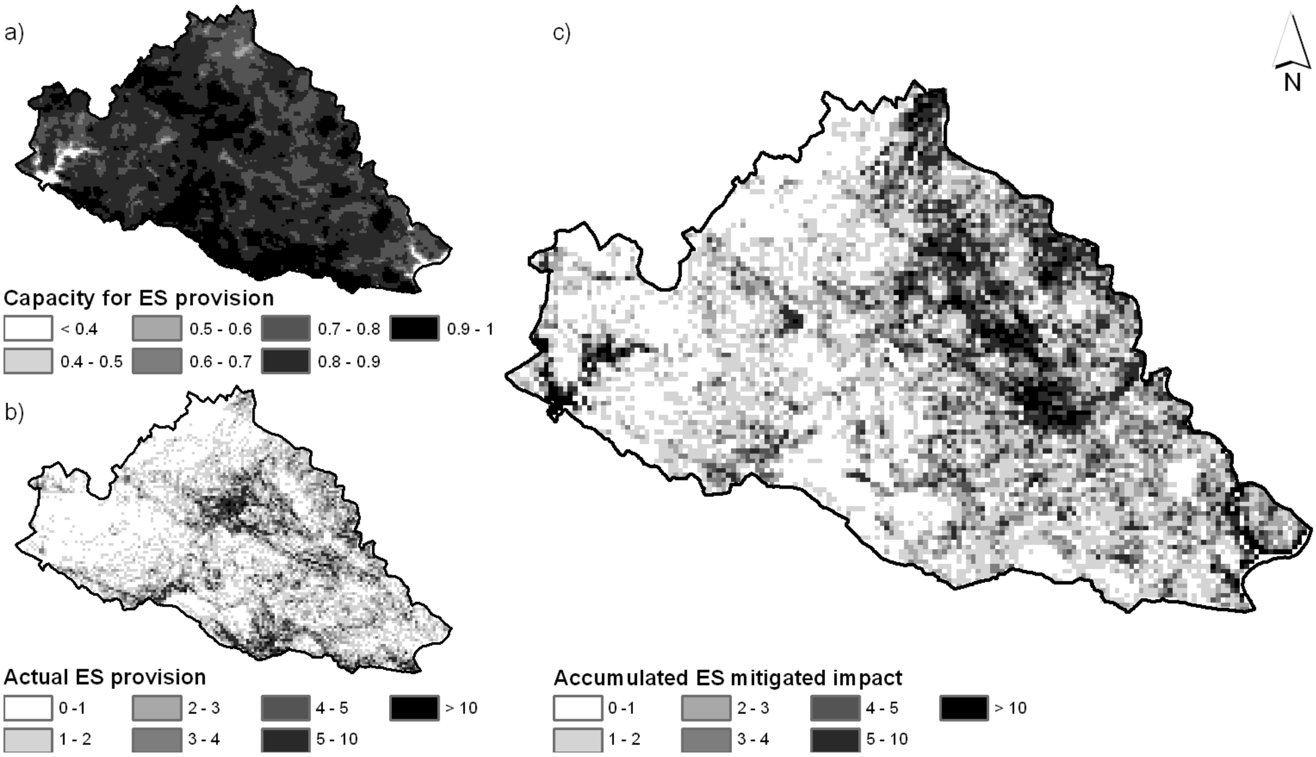

Figure 2.5 Spatial distribution of the: a) annual average of the *capacity for ecosystem service provision*; b) annual average of the *actual ecosystem service provision* (tons of soil per ha per month); and c) the accumulated ecosystem service *mitigated impact* (tons of soil per ha per year).

The range and distribution of accumulated ecosystem service *mitigated impact* is summarized in Fig. 2.5c. It shows that although the *actual ecosystem service provision* is in average sufficient to mitigate the existing *structural impact* (Fig. 2.4a and Fig. 2.4b), the spatial distribution of the ecosystem service *mitigated impact* is very heterogeneous, varying from areas with very high ecosystem service *mitigated impact* (> 10 tons of soil/ha [monthly average]) to areas where the remaining fraction of eroded soil is comparable to the soil formation rates (<1 ton of soil/ha [monthly average]). This makes it possible to identify areas that are vulnerable to the provision of soil erosion prevention and to target corrective measures e.g. reducing cattle density, implement buffer areas, among others.

2.4.2 Land management effects

The three selected civil parishes display a range of farming intensity related to cattle breeding intensity (Fig. 2.3f) that are translated in different *capacity for ecosystem service provision* and *actual ecosystem service* *provision* temporal profiles (Fig. 2.4f). One important finding is that these profiles manifest an inverse gradient in relation to the cattle breeding intensity, i.e. high intensity farming areas like "São Bartolomeu" have a lower ecosystem service provision due to the grazing pressure on the vegetation, while low intensity areas like "Vera Cruz" have a higher ecosystem service provision. This illustrates the possibility to quantify and map the effects of different farming intensities in the provision of soil erosion prevention.

Between the three sites it is not only important to observe and calculate the differences between the ecosystem service provision, but also the effects of this service in the resulting ecosystem service *mitigated impact* (Fig. 2.4e). By observing the temporal patterns of both the structural and the ecosystem service *mitigated impact* (Fig. 2.4d and Fig. 2.4e respectively), it is possible to identify that although the structural impact is smaller in the "São Bartolomeu" site the ecosystem service *mitigated impact* is the highest in most of the year (i.e. January-May, November and December), and even in the month of highest impact (October) it presents a peak close to the one obtained in "Portel", which has a higher *structural impact*. This shows the relevance of the intensity and distribution of the *capacity for ecosystem service provision* in the provision of ecosystem service and how it determines the ecosystem service *mitigated impact*. In this context, the results show a clear relation between land management practices and the effects of these practices on the vegetation cover and consequently in the provision of soil erosion prevention.

2.5 Discussion

With this Chapter we presented a framework to evaluate the provision of soil erosion prevention based on the interaction between the *structural impact* from soil erosion and the *capacity for ecosystem service provision*. Its implementation in Portel illustrated a significant difference between the *capacity for ecosystem service provision* and the *actual ecosystem service provision*, supporting the suggestion that generalized or oversimplified ecosystem service provision models can produce incorrect estimates and thus misinform decision-making by identifying areas with high *capacity for ecosystem service provision* as areas with high *actual ecosystem service provision*.

The framework helped to identify vulnerable areas where concrete measures can and should be implemented by considering the spatial and temporal variability of both the *actual ecosystem service provision* an the ecosystem service *mitigated impact*. In the Portel municipality, although the ecosystem service *mitigated impact* was generally low, the results show that there are vulnerable areas where the provision of the service is not sufficient to successfully mitigate the *structural impact*. In this sense, at the regional level, policy and regulation strategies can be defined to target key aspects of the system that have in consideration its dynamics and main social-ecological potentials, allowing to change the attainable ecosystem service provision of a given area and/or management type.

Likewise, by observing the variations in ecosystem structure as a response to different land management practices it is possible to define management thresholds by determining acceptable levels of the ecosystem service *mitigated impact* (e.g. Verheijen et al., 2009) and compare them with different land management intensities. In this case, considering the normal soil formation rates (Morgan 2005; Verheijen et al. 2013), only the "Vera Cruz" site presents an acceptable management intensity due to the identified ecosystem service provision and the high ecosystem service *mitigated impact* registered in the other two sites in October.

Although the results were appropriate for the selected area, the use of MODIS images (Fritsch et al. 2012) to improve the temporal frequency of the assessment has important limitations (i.e. in terms of spatial resolution) in small scale landscapes where management and habitat conditions may vary substantially within small areas (Ortega et al. 2013). Another important limitation is related to the availability of datasets with high spatial and temporal coverage to infer each system component, though work has been done to improve the quality and comparability of available datasets (e.g. Haylock et al. 2008; Panagos et al. 2011a; Fritsch et al. 2012).

While this framework was applied for soil erosion prevention, the same principles apply to other regulating services (e.g. flood control, carbon fixation, water regulation, and other natural hazards regulation). Correspondingly, this can be done by determining the relevant system components (e.g. runoff, roughness and topography for flood control) and estimating the spatial and temporal variation of each component. Therefore studies addressing the identification and quantification of other regulating services should not be based only on the capacity of specific areas or land cover classes to provide a given service. Also, it is important, following some recent examples (c.f. Wallace 2007), that in the future we are able to distinguish and estimate the contribution of each ecosystem service and reduce the double counting of ecosystem services.

In conclusion, this Chapter shows the limitations for decision making of using the *capacity for ecosystem service provision* as a proxy of the *actual ecosystem service provision* and highlighted the potential misleading conclusions that can result from this analysis. We present a geographic model that can cope with these limitations and produce a spatially and temporally distributed representation of *actual ecosystem service provision*. This model allows to reduce the bias of estimating ecosystem service provision by only using the *capacity for ecosystem service provision*, and increases the accuracy of ecosystem service estimates. At the regional scale (e.g. National or European) the spatial identification of *actual ecosystem service provision* and its temporal variation can improve policy design and allow the definition of land management thresholds for specific areas.

3

POLICY IMPACTS ON REGULATING ECOSYSTEM SERVICES: LOOKING AT THE IMPLICATIONS OF 60 YEARS OF LANDSCAPE CHANGE ON SOIL EROSION PREVENTION IN A MEDITERRANEAN SILVO-PASTORAL SYSTEM

Based on the manuscript:
Guerra C.A., Metzger M.J., Maes J., Pinto-Correia T., 2015. Policy impacts on regulating ecosystem services: looking at the implications of 60 years of landscape change on soil erosion prevention in a Mediterranean silvo-pastoral system. Landscape Ecology. DOI:10.1007/s10980-015-0241-1

Abstract

Policy decisions form a major driver of land use change, with important implications for socially and environmentally susceptible regions. It is well known that there can be major unintended consequences, especially where policies are not tailored to regionally specific contexts. In this Chapter we assess the implications of 60 years of agricultural policies on soil erosion prevention by vegetation, an essential regulating ecosystem service in Mediterranean Europe. To assess these implications we produced and analysed a time series of land cover/use and environmental conditions datasets (from 1951 to 2012) in relation to changing agricultural policies for a specific region in the southern Portugal. A set of indicators related to soil erosion prevention allowed us to identify that land use intensification as increased soil erosion in the last 60 years. Particularly in the last 35 years, as a consequence of headage payments for cattle, the agricultural policy had a significant effect in the density and renewal of the tree cover, resulting in drastic effects for the provision of the soil erosion prevention service. These are more significant after 1986, coinciding with the implementation of several Common Agricultural Policy instruments focused on increasing the modernization and productivity capacity of farm systems. The results show some unintended effects of agricultural policy mechanisms on ecosystem service provision and highlight the need for context-based policies, tailored to the environmental constrains and potentials of each region.

3.1 Introduction

Land use change has dramatically increased in recent decades due to rapid technological, social and economic changes at the global level (Ojima et al. 1994; Rabbinge and van Diepen 2000; Renwick et al. 2013). Policy shifts, e.g. in production support mechanisms, are one of the main drivers influencing the way landowners manage their farms and therefore also the land use of the farm (Brouwer and van der Heide 2009; Swaffield and Primdahl 2010; Primdahl et al. 2013; Ribeiro et al. 2014). Policy shifts can therefore lead to radical transformations at the landscape level, resulting in unintended changes in the provision of ecosystem goods and services (MA 2005b; Braat and de Groot 2012; Rounsevell et al. 2012). Consequently, there is a growing interest among policy-makers, practitioners and scientists to develop indicator-based approaches to assess the impacts of policies on the ecological sustainability of agricultural land use systems (Brouwer and van Ittersum 2010).

The use of ecosystem services as indicators of regional sustainability offers several advantages as they are affected by changes in land use/cover (Metzger et al. 2006; Müller and Burkhard 2012; Haines-Young et al. 2012) and represent the direct and indirect contributions of ecosystems to human well-being (Chiesura and de Groot 2003; Chapin et al. 2010). Ecosystem service indicators are therefore seen as promising policy-relevant tools to identify gaps and communicate trends about recent, past or potential future states of a given social-ecological system (Müller and Burkhard 2012).

In the past few years, several studies have related ecosystem services with key ecosystem functions (e.g. Schäfer 2012; Crossman et al. 2013), but the link between policy measures and their effects on ecosystem functions and services is rarely made explicit (van Meijl et al. 2006). Despite significant policy interest (Brink et al. 2012), there is often a lack of empirical information about the spatial and temporal flows of ecosystem services and their rates of change (Fisher et al. 2009; Scholes et al. 2013). Haines-Young et al. (2012) argue this is due to the complexity of obtaining direct measures of ecosystem service outputs and from the fact that existing monitoring systems were not designed to deliver such information. Further study is therefore required to develop efficient methods to assess the impact of policies in the environment (Hauck et al. 2013; Crossman et al. 2013) as recognised by the European Union (EU) within the Mapping and Assessment of Ecosystems and their Services (MAES) initiative (Maes et al. 2013b; Maes et al. 2013a).

This is especially important when addressing sensitive social-ecological systems like the extensive Mediterranean silvo-pastoral systems, (e.g. the montado in Portugal and the dehesa in Spain) which support many highly valued ecosystem services (Pinto-Correia and Godinho 2013). These social-ecological systems cover an area of about 3.5–4.0 M ha in the south-western Iberian Peninsula (Olea and Miguel-ayanz 2006) under severe limiting conditions for agriculture: a dry Mediterranean climate with strong inter-annual variability and shallow soils with low organic matter content (Pinto-Correia and Vos 2004). In order to cope with the scarcity of the environmental conditions, these systems are characterised by complex interactions and a sensitive balance between tree cover and the different and complementary uses of the under cover (i.e. between forestry and agricultural uses). The different uses need to be balanced or the systems will collapse: if the agricultural use is too extensive, there will be shrub encroachment and forest densification; if the agricultural use is too intensive, the trees cover will be reduced and the forestry component will likely disappear (Bugalho et al. 2011; Costa et al. 2011). Within this system, regulating services (e.g. climate regulation or soil erosion prevention) are strongly affected by changes in the land use intensity, which in turn strongly depend on changes in the mechanisms of public policies (Primdahl et al 2013). For the Iberian silvo-pastoral systems, the most documented example are the livestock payment schemes under the Common Agricultural Policy, which so far in Iberia have been kept coupled to headage. At the same time as competing in a global market has become more difficult for these extensive, family farms, dependency on agricultural policy payments has increased (Costa et al 2011; Ribeiro et al 2014). These schemes are thus heading to a progressive but constant increase in the intensity of livestock pressure, due to an increased number of animals using the grazing areas (Pinto-Correia et al. 2013; Ribeiro et al. 2014).

The spatial and temporal mismatch between service provision (e.g. flood regulation) and the benefit (e.g. reduced flood risk) highlights the need to assess regulating services (Nedkov and Burkhard 2011; Guerra et al. 2014). To establish an indicator framework to support decision-making it is therefore necessary to identify ways to assess the implications of policies on the provision of regulating services, and to establish possible cause-effect relations. This indicator framework is dependent on the availability of spatially explicit information on the state and trends of ecosystems and their services (Maes et al. 2012).

In this Chapter, we calculate and map 24 state and impact indicators to characterize the provision of the soil erosion prevention (SEP) service, assessing how agriculture policy measures implemented over the past 60 years have affected the ecosystem service provision in the montado. This specific service was selected as it (a) corresponds to a regulating service and therefore enables conditions for the flow of other services (Power 2010), (b) regulates a key impact in the selected social-ecological system (Pinto-Correia et al. 2011), and (c) has been subject to targeted international, national and regional policy measures over the past decades (e.g. CAP agri-environmental measures).

The following section presents the conceptual approach, describes the evolving agricultural policy affecting the montado, and explores the modelling methods. The results section then demonstrates how the presented approach was applied to produce a long-term spatially explicit assessment of ecosystem service based indicators relating to SEP, and how ecosystem provision was influenced by changes in the agricultural policy. Finally the discussion and conclusions sections discuss the findings and their implication on current and future policy design.

3.2 Methods

3.2.1 Conceptual approach

In this Chapter we implement the conceptual model recently described by Guerra et al. (2014), and summarised in Fig. 3.1, which identified a set of process based indicators to calculate, map and describe the provision of regulating services. The analysis is carried out for a 60 years time-series (between 1951 and 2012) and is linked to the changing agricultural policy in a montado based case-study region in southern Portugal.

SEP is provided at the interface between the biophysical and climatic components (i.e. structural) of the social-ecological system and its land use-land cover dynamics (Fig. 3.2), as the ecosystem service providers (in this case vegetation cover) mitigate the impact from soil erosion (Guerra et al., 2014). To begin assessing SEP we need to identify the *structural impact* (Υ) of soil erosion, i.e. the erosion that would occur when vegetation is absent and therefore no ecosystem service is provided (Fig. 3.1a). It determines the potential soil erosion (tons of soil per pixel area in a given year) in a given place and time and is related to variables such as rainfall erosivity (i.e. the erosive potential of rainfall), soil erodibility (as a characteristic of the soil type) and local topography (Panagos et al. 2011b). Although external drivers, e.g. climate change, can have an effect on these variables, they are less prone to be changed directly by human action.

The *actual ecosystem service provision* (*Es*) reduces the total amount of *structural impact* (Υ), and we define the remaining impact as the *ecosystem service mitigated impact (β_e)* (i.e. the remaining soil erosion after the provision of the ecosystem service). We can then define the *capacity for ecosystem service provision (e_s)* as a key component to determine the fraction of the *structural impact* that is mitigated (Fig. 3.1b). This *capacity for ecosystem service provision (e_s)* is influenced by both internal and external social-ecological drivers. Examples of internal drivers include management options, forest fires, and urban sprawl, whilst agricultural policy measures, spatial planning, and climate change are examples of external drivers affecting SEP.

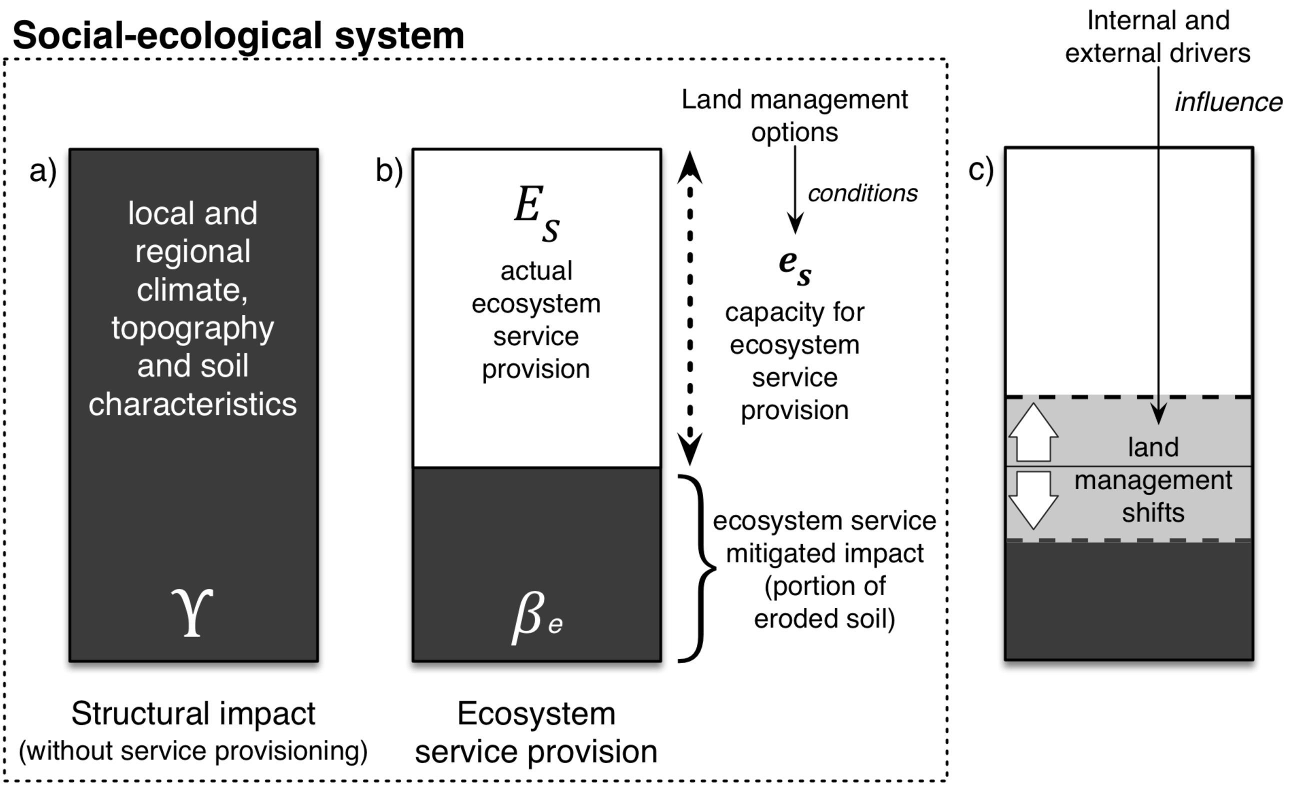

Figure 3.1 Conceptual framework for assessing the provision of regulating services (adapted from Guerra et al., 2015), where (a) presents the *structural impact* (Υ) (i.e. the total soil erosion impact in the absence of soil erosion prevention); (b) distinguishes the *actual ecosystem service provision* (*Es*) as an avoided portion of the *structural impact* (measured in tons of soil not eroded) and determined by the *capacity for ecosystem service provision* (e_s) (i.e. the fraction of the *structural impact* that is mitigated by the ES, corresponding to an adimensional gradient ranging from 0 to 1), and the remaining ecosystem service *mitigated impact* (β_e) (i.e. the remaining soil erosion that is not regulated by soil erosion prevention); and (c) considers the variations in the *actual ecosystem service provision* resulting from changes in land management that occur at the local level although influenced by internal and external drivers.

To understand the relation between policy and ecosystem service provision, it is necessary to translate the dynamics of the social-ecological system in a set of process relevant indicators that express the system responses to specific policy measures (Fig. 3.2). One of the main issues to establish these relations is the site specificity of service provision in relation to the broader spatial and sectoral scope of policy measures. This is particularly relevant for agricultural policies as they are applied to land use systems that can be found in areas with significantly diversified social, economic and environmental conditions. It is therefore necessary to (a) identify relevant policy measures for each specific social-ecological system, (b) describe the social system responses to selected policies (e.g. in terms of land use and land cover change), and (c) identify and measure their implications in the ecosystem service provision (e.g. in terms of the influence in the dynamics of ecosystem service provision) (Fig. 3.2).

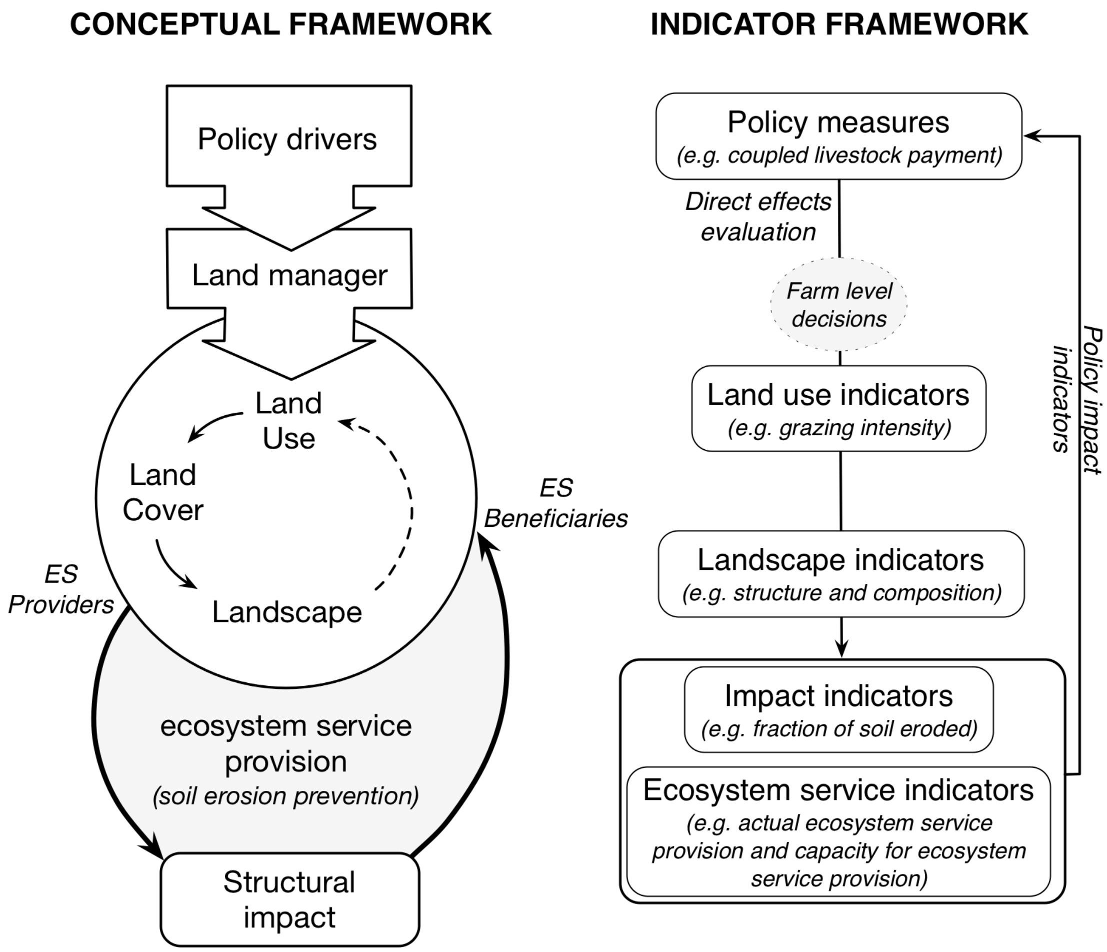

Figure 3.2 Diagram indicating how the conceptual framework (Fig. 1) can be converted in to an indicator based approach to assess the impact of specific policy measures in the provision of ecosystem services (adapted from Guerra et al. (2014)).

To identify relations between policy measures and ecosystem service indicators for our Portuguese case-study we have (a) identified the relevant agricultural policies and policy measures that are related to impacts on the provision of SEP, (b) calculated and described the dynamics of key land use and landscape indicators, and (c) calculated and described the dynamics of key environmental impact and ecosystem service indicators and assess their relation to the selected policies, through the evaluation of their response to the dynamics characterized in step two.

3.2.2 Sixty years (1951-2012) of policy change in the montado

Despite its particularities, the montado system has hardly been considered as a distinct land use system in policy, and policy measures have not considered the sensitivity of the system (Table 3.1) (Pinto-Correia 2000; Ferreira 2001; Pinto-Correia and Godinho 2013). Consequently, agricultural policies with a production target have had severe unintended effects on the system over the past decades (Schröder 2011; Costa et al. 2011; Godinho et al. 2014).

From 1929 until the early 1960s the dominant policy affecting land use in Alentejo was the Wheat Campaign, which aimed at increasing national cereal production by improving crop production and minimizing imports. During the 1950s financial incentives supporting agricultural mechanisation resulted in wide-scale clearing of the montado for cereal production (Pinto-Correia and Mascarenhas 1999; Pinto-Correia 2000; Ferreira 2001). During the 1960s its effect lessened due to (a) a decrease in the protectionist mechanisms, (b) the high rate of emigration and reduced labour force in the region, and (c) the introduction of an afforestation policy that subsidised 50% of the planting costs. As an effect, the total cultivated area decreased and the montado regained shrub and some tree regeneration (Jones et al. 2011b).

From 1975 to 1979 the Agrarian Reform introduced a new phase of aggressive intensification, with increasingly heavier machinery used to clear montado for cereal production and livestock grazing (Ferreira 2001). With the output price support both collective and privately managed farms invested heavily in cereal production, at the expense of the montado (Avillez et al. 1988).

Table 3.1 Main policy drivers of land use/cover change in the montado system between 1940 and 2012.

Policy	**Relevant measures**	**Scope of change**	**Temporal scope**
Wheat campaign	Minimum prices for wheat Land clearing subsidies Protectionist policy	Until 1963 the support scheme includes minimum prices, land clearing and fertiliser's subsidies. From then on land clearing subsidy is suspended.	1928-1963
Private forestation policy		The support includes loans covering 50% of installation costs.	1965-1983
Wheat policy reform	Support for the afforestation of marginal lands	The reform includes more incentive to marginal land rehabilitation through afforestation and structural measures	1970
Agrarian reform	Output price supports Input price subsidies Land market regulations Agricultural credit programmes	After 1974 output price supports, input price subsidies, land market regulations, and agricultural credit programmes are implemented. Cereals and milk/meat sectors are among the most supported.	1975-1982
Agricultural Transitory Measures	Access to various EC financial support programs to develop the agriculture and forestry sector, strength production capacity and prepare the country for international markets	Modernization of agricultural infrastructures and investments in agriculture intensification. In some areas, support to regionally adapted crops and breeds. This corresponded to the CAP transition period.	1980-1986

Table 3.1 Main policy drivers of land use/cover change in the montado system between 1940 and 2012 (cont.).

Policy	Relevant measures	Scope of change	Temporal scope
PEDAP – specific programme for the Portuguese Agriculture	Investment support Production support	Intensification of production. Livestock increased due to direct payments.	1986-1992
Common Agricultural Policy reform (MacSharry's)	Direct payments and coupled supports Compensation measures as forestation support Agri-environment measures Early retirement Set-aside	The 1992 CAP reform introduced direct payments per animal, reducing intervention prices. This was an incentive to livestock intensification. Parallel, measures were created to plant new forest areas and to protect extensive pastures (agri-environmental schemes) – but the budget of these agri-environmental and afforestation measures is much smaller than production payments and the impact is mostly on marginal and follow-up measures aimed at reducing the negative impacts of agriculture in Natura 2000 and improve agro-ecosystems services.	1992-1996
Common Agricultural Policy	Compensation payments	This period resulted in a continuation of the previous reform with direct payments and intensification incentives.	1996-2000
Common Agricultural Policy	Compensation payments Environmental and Cultural landscape payments (2nd pillar)	In this period the rural development share of the CAP was separated into the second pillar although with a small budget. Direct area payments were implemented. The 2003 mid term review introduced the decoupled support, although in Portugal livestock payments were kept coupled.	2000-2006 2006-2013

From 1975 to 1979 the Agrarian Reform introduced a new phase of aggressive intensification, with increasingly heavier machinery used to clear montado for cereal production and livestock grazing (Ferreira 2001). With the output price support both collective and privately managed farms invested heavily in cereal production, at the expense of the montado (Avillez et al. 1988).

From 1980 to the present the Portuguese agricultural policy has been increasingly aligned with the European Common Agricultural Policy (CAP). First, under the Agricultural Transitory Measures, there was direct financial support for investments infrastructure and machinery. After 1986, when Portugal and Spain entered the European Economic Community, Portuguese agriculture received production support in line with the EU (Buckwell et al. 1994; Pinto-Correia and Mascarenhas 1999). In 1992 the CAP was reformed to reduce agricultural overproduction in Europe (Buckwell et al. 1994). This caused lower guaranteed prices along with set-aside and other compensation schemes (including agri-environmental measures, afforestation of agricultural land and early retirement) to provide income support (Buller et al. 2000). In Portugal the effects of the compensation schemes included the afforestation of marginal areas, but these reforms did not reduce the intensification on more favourable land (Jones et al. 2011b).

The Agenda 2000 reform of the CAP was mainly a reinforcement of the 1992 reform, with further decreases in prices and a clear policy of compensation through direct area payments to farmers. Rural development payments, including payment for environmental measures, were separated into a 'second pillar'. This included measures to preserve the montado (e.g. subsidies to maintain and increase tree densities), but the effects were limited as the combined measures in the second pillar received less than 10% of the total CAP budget (Brouwer and van der Heide 2009; Pinto-Correia and Primdahl 2009).

The 2003 Mid-Term review of the CAP brought a further change: to avoid extreme intensification, payments were decoupled from production and calculated based on the historical payments received by area in production. However, Member States could opt for keeping part of the payments coupled. This happened in Portugal, where livestock subsidies have been kept entirely coupled (Pinto-Correia and Primdahl 2009), i.e. support is given per headage. Furthermore, cattle payments increased sharply while sheep payments remained more or less unchanged. This decision, which has remained unchanged until today, forms one of the strongest pressures on the montado in recent years (Pinto-Correia et al. 2011). The existing policy mechanisms thus continue to lead to the intensification of cattle production in the montado, weakening tree regeneration and resulting in tree clearance and increased openings in the tree cover (Pinto-Correia and Godinho 2013; Pinto-Correia et al. 2014).

3.2.2.1 Description of the case study region

To study the effects of the described policy changes on SEP we selected a relatively small and homogeneous area in southern Portugal, approximately 64 km^2 in size. The area is located near Montemor-o-Novo (Fig. 3.3), a region where the property structure is dominated by large estates. Its montado landscape is composed of open tree cover in changing densities of *Quercus suber* or *Quercus ilex*, and pastures with shrub patches in the under cover where extensive grazing occurs (Pinto-Correia 1993; Almeida et al. 2013).

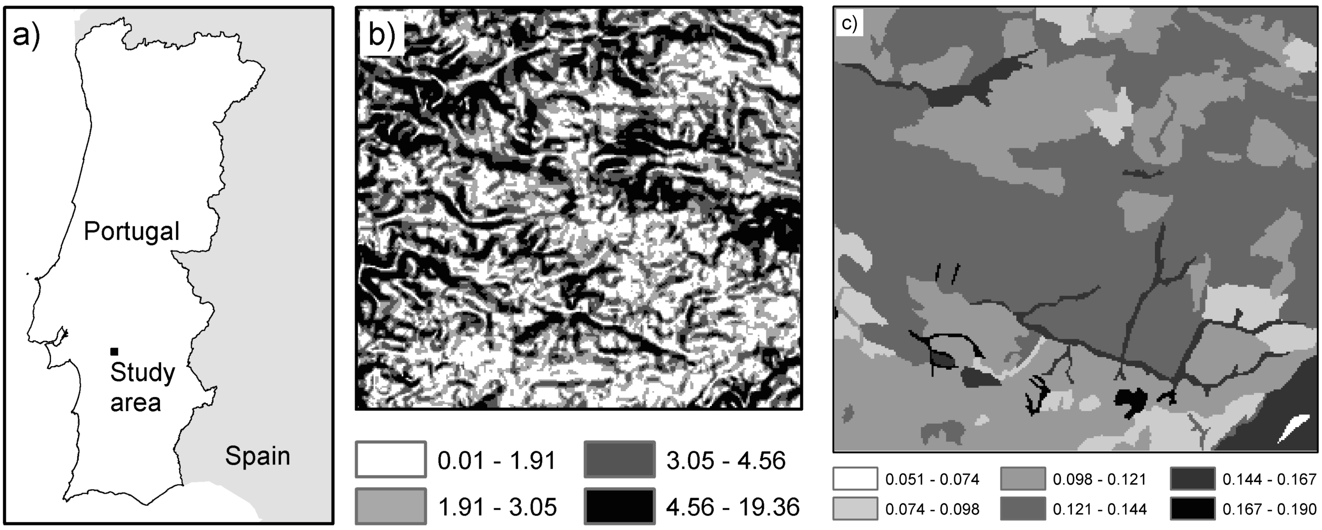

Figure 3.3 Location of the selected case study area and main environmental and social traits: (a) spatial location; (b) slope in degrees; and (c) soil erodibility.

This case-study area was chosen because it reflects the main traits and trends of the montado social-ecological system, and the required spatial and temporal explicit datasets were available to carry out the long-term evaluation of the potential relations between SEP provision and the described policy measures.

3.2.3 Indicators and methodological application

3.2.3.1 Indicator description

Following the indicator framework in Fig. 3.2, we selected a range of indicators that discriminate between significant land use, land cover and ecosystem service provisioning dynamics. These indicators include grazing intensity, land use states and dynamics, land cover fragmentation and dynamics and ecosystem service provision as well as ecosystem mitigated impacts and the capacity for ecosystem service provision. A full description of each selected indicator is provided in Table 3.2.

3.2.3.2 Data gathering and mathematical implementation

Following Guerra et al. (2014) SEP provided by vegetation cover from 1951 to 2012 was calculated using an adaptation of the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978). The USLE forms a commonly used empirical model for the determination of potential soil losses (Fistikoglu and Harmancioglu 2002; Amore et al. 2004), and for assessing mitigation measures at the farm system level (e.g. Erskine et al. 2002). Within USLE, soil erosion is represented by a set of critical factors given by (Panagos et al. 2011b):

$$A = R \times LS \times K \times C \times P \qquad \text{(Eq.3.1)}$$

where, A represents the amount of soil loss, R the rainfall erosivity factor, LS the topographic factor, K the soil erodibility factor, C the vegetation cover factor and P the conservation practices factor.

Table 3.2 Identification and description of the indicators used to describe the land use, land cover and ecosystem service provision dynamics (all spatial indicators were calculated using a 30x30 meters pixel resolution).

Indicator type	Indicator	Description	Units
Land Use	Grazing intensity	Total number of grazing cows (estimations made based on data available at the civil parish level and at the municipality level)	Number of grazing cattle
	Variation in grazing intensity	% variation in the total number of grazing cows considering the previous reference date	%
	Total arable land and permanent crops	Total area classified as arable land and as permanent crops	ha
	Total silvo-pastoral area	Total area classified as silvo-pastoral area	ha
	Total forest area	Total area classified as forest area	ha
	Total area of permanent pastures	Total area classified as permanent pastures	ha
	% of changing area	% of area that changed in land use class between periods	%
	% of maintenance area	% of area that maintained the land use class between periods	%
Land Cover	% of total area	% of total area per land cover class	%
	% of changing area	% of area that changed in land cover class between periods	%
	% of maintenance area	% of area that maintained the land cover class between periods	%

Table 3.2 Identification and description of the indicators used to describe the land use, land cover and ecosystem service provision dynamics (all spatial indicators were calculated using a 30x30 meters pixel resolution) (cont.).

Indicator type	Indicator	Description	Units
Land Cover	Number of patches	Total number of patches	nr.
	Total edge	Total patches perimeter	km
	Mean patch size	Average patch size	ha
	Shannon's diversity index	Measure of relative patch diversity. The index will equal zero when there is only one patch in the landscape and increases as the number of patch types or proportional distribution of patch types increases.	-
Impacts and ecosystem service provision	Structural impact	Total annual soil erosion impact when no ecosystem service is provided	tons of soil per pixel
	Ecosystem service mitigated impact	Total annual of the remaining soil erosion after the ecosystem service provision	tons of soil per pixel
	Actual ecosystem service provision	Total annual of the actual ecosystem service provision corresponds to the total amount of ecosystem service provided, measured in ecosystem service providing units (tons of soil not eroded). It varies from season to season and year-to-year depending on the variation of the structural impact	tons of soil per pixel
	Capacity for ecosystem service provision	Average annual fraction of the structural impact that is mitigated by the ecosystem service, it corresponds to an adimensional gradient from 0 to 1	-
	Variation in structural impact	% variation in the total amount of structural impact considering the previous reference date	%
	Variation in ecosystem service provision	% variation in the total amount of actual ecosystem service provision considering the previous reference date	%
	Rate of effective ecosystem service provision	% variation in the total amount of actual ecosystem service provision corrected by the structural impact fluctuations for a given region using the following expression $100\times\left(\left[\frac{E_{S(t+1)}}{E_{S(t)}}-1\right]-\left[\frac{\Upsilon_{(t+1)}}{\Upsilon_{(t)}}-1\right]\right)$, where ecosystem services is the total annual actual ecosystem service provision, Υ is the total annual structural impact, and t corresponds to the temporal frame.	%
	Variation in the capacity for ecosystem service provision	% variation in the total amount of ecosystem service provision capacity considering the previous reference date	%
	Variation in ecosystem service mitigated impact	% variation in the total amount of ecosystem service mitigated impact considering the previous reference date	%

In this context, the structural impact (Υ) was calculated using the expression $\Upsilon = R\times LS\times K$ (Prasuhn et al. 2013), and the gradient of ecosystem service mitigated impact will be determined by $\beta_e = \Upsilon\times\alpha$ (where $\alpha = C$ and $e_s = 1-\alpha$). These two expressions allow to determine the actual ecosystem service provision (E_s). Although not allowing an absolute measure of soil erosion, this mathematical formulation defines a spatially explicit gradient of potential soil loss and the correspondent gradient of ecosystem service provided by vegetation cover (E_s).

The rainfall erosivity factor was estimated based on the MedREM model proposed by Diodato and Bellocchi (2010) for Mediterranean conditions. An available precipitation dataset from a meteorological station located within the area was used to obtain daily precipitation data from 1931 to 2010. Afterwards, this dataset was segmented into six temporal segments according to six defined temporal frames ([1931-1952]; [1949-1970]; [1966-1987]; [1975-1996]; [1984-2005]; [1989-2010]). For each temporal frame, the rainfall erosivity factor was calculated using the following expression (Diodato and Bellocchi 2010):

$$R_m = b_0 \times P_m \times \sqrt{d_m} \times (a + b_1 \times L) \qquad \text{(Eq.3.2)}$$

where, R_m (MJ.mm.ha^{-1}.h^{-1}.month^{-1}) corresponds to the monthly erosivity factor for the month m, b_0 (MJ.ha^{-1}.h^{-1}) is a constant equal to 0.117, b_1 (d$^{0.5}$.mm$^{-0.50-1}$) is a constant equal to -0.015, a (d$^{0.5}$.mm$^{-0.50}$) is a constant equal to 2, L (º) corresponds to the site longitude, P_m (mm) to the total amount of precipitation in a given month m, and d_m (mm.d-1) to the monthly maximum daily precipitation for month m averaged over a multi-year period. Considering the geographic extent, it is assumed that the rainfall erosivity factor is spatially homogeneous for the entire area.

To estimate soil erodibility an available dataset for the region was used (Cardoso 1965), in which 98 soil profiles were collected, and the expression (Wischmeier and Smith 1978; Morgan 2005; Prasannakumar et al. 2012):

$$K = \frac{2.1 M^{1.14} 10^{-4} (12 - a) + 3.25 (b - 2) + 2.5 (c - 3)}{100 \times 7.59} \qquad \text{(Eq.3.3)}$$

where, K corresponds to the soil erodibility factor, a is the percentage of organic matter, b the soil structure parameter, c the profile permeability class, and $M = (\%_{\text{silt}} + \%_{\text{very fine sand}}) \times (100 - \%_{\text{sand}})$. For the topographic factor a 30 meters resolution topographic dataset provided by the Portuguese National Geographic Institute was used to compute the following expression (Moore and Burch 1986):

$$LS = \left(\frac{a \times p}{22.13} \right)^{0.4} \times \left(\frac{\sin(d)}{0.0896} \right)^{1.3} \qquad \text{(Eq.3.4)}$$

where, LS is the topographic factor, a refers to the flow accumulation model obtained from the topographic dataset, p to the pixel size, and d to the slope model in degrees.

Vegetation cover was estimated using a land cover time series for six dates (i.e. 1951, 1969, 1986, 1995, 2004, and 2012) covering 60 years of land cover dynamics. These land cover maps were produced by photointerpretation of multiple aerial photographs ranging from 1951 to 2012, following a specific hierarchical land cover classification (Table 3.3).

For each of the six considered time slices, a complete coverage was obtained and ground validation data from 2004 and 2012 was used to assist in the land cover interpretation and classification. The remaining land cover maps (1951, 1969, 1986, and 1995) were made based on these late land cover maps (i.e. 2004 and 2012) by only registering changes in land cover between time slices (cf. Liu et al. 2007; Arnaez et al. 2011; Otero et al. 2011). After these land cover maps were produced, a set of 37 Landsat images ranging from January of 2011 to December 2013 was compiled and transformed into the normalized difference vegetation index (NDVI; Seto et al. 2004; Jiang et al. 2006).

Table 3.3 Hierarchical land cover classification and the respective ecosystem service provisioning capacity, calculated in a monthly basis from NDVIs obtained from Landsat images (period: 2011-2013) and averaged for each land cover class.

Land cover classification				**Capacity for ecosystem service provision (e_s)**											
L 1	**L 2**	**L 3**	**Description**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Set**	**Oct**	**Nov**	**Dec**
1			**Artificial surfaces**	-	-	-	-	-	-	-	-	-	-	-	-
2			**Agricultural areas**												
2	1		Arable land												
2	1	1	Irrigated arable land	0,981	0,973	0,943	0,983	0,890	0,673	0,568	0,562	0,528	0,531	0,889	0,975
2	1	2	Non-irrigated arable land	0,947	0,959	0,913	0,961	0,818	0,612	0,549	0,549	0,488	0,540	0,853	0,941
2	2		Permanent crops												
2	2	1	Olive groves	0,949	0,961	0,925	0,974	0,847	0,647	0,618	0,611	0,551	0,580	0,933	0,946
2	2	2	Orchards	0,938	0,964	0,906	0,946	0,808	0,658	0,589	0,606	0,540	0,578	0,903	0,939
2	4		Silvo-pastoral areas (“montado”)												
2	4	1	Silvo-pastoral areas with >50% of tree cover	0,936	0,967	0,931	0,954	0,889	0,823	0,763	0,763	0,739	0,751	0,913	0,940
2	4	2	Silvo-pastoral areas with 30-50% of tree cover and >50% of shrubs	0,889	0,927	0,870	0,913	0,838	0,792	0,713	0,711	0,682	0,668	0,832	0,888
2	4	3	Silvo-pastoral areas with 30-50% of tree cover and <50% of shrubs	0,940	0,965	0,925	0,961	0,871	0,756	0,681	0,683	0,636	0,658	0,911	0,944
2	4	4	Silvo-pastoral areas with <30% of tree cover and >50% of shrubs	0,948	0,971	0,947	0,954	0,883	0,811	0,744	0,734	0,697	0,732	0,929	0,939
2	4	5	Silvo-pastoral areas with <30% of tree cover and <50% of shrubs	0,942	0,963	0,921	0,959	0,866	0,717	0,621	0,619	0,570	0,599	0,889	0,937
2	5		Permanent pastures	0,955	0,967	0,933	0,961	0,875	0,685	0,571	0,568	0,520	0,573	0,914	0,947
3			**Forest areas**												
3	1		Mixed forest	0,933	0,956	0,934	0,986	0,905	0,831	0,747	0,741	0,674	0,689	0,901	0,930
3	2		Production forest	0,923	0,945	0,861	0,916	0,820	0,749	0,671	0,735	0,720	0,738	0,893	0,904
3	3		Shrubs and/or herbaceous vegetation associations	0,781	0,841	0,707	0,748	0,617	0,565	0,488	0,512	0,463	0,439	0,600	0,661
4			**Water bodies**	-	-	-	-	-	-	-	-	-	-	-	-

Each NDVI image was converted in vegetation cover (C) using the expression proposed by Van der Knijff et al. (1999, 2000) (Prasannakumar et al. 2012):

$$C = \exp\left[-a \times \frac{NDVI}{\left(b - NDVI\right)} \right] \qquad \text{(Eq.3.5)}$$

where, a=2 and b=1.

Finally, zonal statistics were calculated for the mean value of C present in each patch present in the land cover map of 2012. Afterwards, all values were grouped by land cover class and the average value of C was calculated for each month. This procedure made it possible to obtain a within-year profile of the variation of vegetation cover per land cover class (Table 3.3). The 2012 template of vegetation cover variation was reproduced for the remaining dates to overcome the diminished data availability for early dates.

3.2.4 Data integration and statistical analysis

A comparative analysis was made to establish consistent relations between the identified policy measures and the resulting landscape and ecosystem service changes. The final group of indicators was divided into static indicators (e.g. % of silvo-pastoral areas, total actual ecosystem service provision) and dynamic indicators (e.g. % of changed area, rate of effective ecosystem service provision) to better grasp the relationships between type and rate of changes.

The Pearson correlation coefficient (r) was used separately to static and dynamic indicators to observe the relationships between indicators, particularly between the provision of ecosystem services and the intensity and frequency of landscape changes. Based on previous studies (e.g. Schröder 2011; Costa et al. 2011), here we assume that most changes in land management in this region result from social-economic changes derived from policy incentives and measures. All the values were computed considering the results for the entire study area (Table 3.4).

3.3 Results

3.3.1 Land use and land cover change

Fig. 4 provides a comprehensive picture of the main land use and land cover changes in the area. Between 1951 and 2012 montado was the dominant land use in the case-study area, covering in average 67.7% of the area, with an absolute variation of -3.5%. Although this absolute variation is not significant, the spatial variation tells a different story: 26.4% of the original montado area changed land use over the 60 years period. During this period cropland areas have decreased by 57.7% (from 21.9% to 9.3%) at the same time that permanent pastures increased by 119.2% (from 8.8% to 19.3%). This happened mainly between 1969 and 2012, following the end of the protectionist policy and the beginning of the cattle subsidies. This trend becomes more obvious after 1986 and results from a reduction of the agricultural areas by 12.6 percentage points between 1951 (21.9%) and 2012 (9.3%). Although this is the general trend, it is relevant to observe that these agricultural areas register an increase of 8.9% between 1969 and 1986 (mainly related with annual crops) and again between 2004 and 2012 (+15% mainly related with permanent crops) (Table 3.4).

Between 1986 and 1995 the rate of change (1.4% per year) was almost twice as high compared to the 1969-1986 period (0.71% per year). It decreased after 1995 resulting in relative stable land use, with change occurring in less than 3.5% of the region. One key driver of change is the grazing intensity and its effects on soil degradation. It shows a strong increase on the number of grazing cows from 1986, a trend that continues until 2012 (Table 3.4).

Table 3.4 Quantification of the selected land use, land cover, impact and ecosystem service provision indicators.

Type	Indicator	1951	1969	1986	1995	2004	2012
Land use	Grazing intensity[(a)]	23[1]	-	864[2]	1788[3]	-	3 489[4]
	Variation in grazing intensity	-	-	+3 619	+106	-	+95
	Total arable land and permanent crops	1 288	774	843	414	474	545
	Total silvo-pastoral area	4 017	4265	3 888	3 939	3 894	3 877
	Total area of permanent pastures	519	620	886	1 275	1 199	1 139
	Total forest area	27	153	193	163	213	215
	% of changing area	-	18.7	12.1	12.3	3.5	2.1
	% of maintenance area	-	81.3	87.9	87.7	96.5	97.9

Table 3.4 Quantification of the selected land use, land cover, impact and ecosystem service provision indicators (cont.).

Type	Indicator	1951	1969	1986	1995	2004	2012
Land cover	% of artificial surfaces	36	46	49	58	66	68
	% of irrigated land	13	82	104	44	100	96
	% of non-irrigated land	1 219	557	600	224	183	187
	% of olive groves	16	64	64	58	127	174
	% of orchards	40	71	75	88	64	89
	% of silvo-pastoral areas with >50% of tree cover	2 458	2 687	2 177	1 971	1 890	1 849
	% of silvo-pastoral areas with 30-50% of tree cover and >50% of shrubs	7	25	23	49	63	126
	% of silvo-pastoral areas with 30-50% of tree cover and <50% of shrubs	694	669	782	851	779	784
	% of silvo-pastoral areas with <30% of tree cover and >50% of shrubs	52	8	5	7	83	18
	% of silvo-pastoral areas with <30% of tree cover and <50% of shrubs	806	876	902	1 061	1 079	1 100
	% of permanent pastures	519	620	886	1 275	1 199	1 139
	% of mixed forest	18	26	28	34	37	36
	% of production forest	8	127	164	130	172	171
	% of shrubs and/or herbaceous vegetation associations	0	0	0	0	4	7
	% of water surfaces	0	28	29	38	41	44
	% of changing area	-	31.5	20.5	19.6	10.1	6.8
	% of maintenance area	-	68.5	79.5	80.4	89.9	93.2
	Number of patches	425	531	575	655	706	710
	Total edge	910	1 018	1 069	1 144	1 185	1 205
	Mean patch size	13.85	11.09	10.24	8.99	8.34	8.29
	Shannon's diversity index	1.60	1.71	1.84	1.80	1.911	1.938
Impacts and ecosystem service provision	Structural impact	435 879	529 064	418 562	406 726	390 345	349 166
	Ecosystem service mitigated impact	44 936	53 637	45 943	47 990	52 791	52 505
	Actual ecosystem service provision	390 943	475 427	372 619	358 736	337 554	296 661
	Capacity for ecosystem service provision	0.822	0.830	0.824	0.814	0.825	0.824
	Variation in structural impact	-	+21.38	-20.89	-2.83	-4.03	-10.55
	Variation in ecosystem service provision	-	+21.61	-21.62	-3.73	-5.90	-12.11
	Rate of effective ecosystem service provision	-	0.23	-0.74	-0.90	-1.88	-1.57
	Variation in the capacity for ecosystem service provision	-	+1.00	-0.66	-1.22	+1.33	-0.12
	Variation in ecosystem service mitigated impact	-	+19.36	-14.34	+4.46	+10.00	-0.54
	Structural impact	435 879	529 064	418 562	406 726	390 345	349 166
	Ecosystem service mitigated impact	44 936	53 637	45 943	47 990	52 791	52 505
	Actual ecosystem service provision	390 943	475 427	372 619	358 736	337 554	296 661
	Capacity for ecosystem service provision	0.822	0.830	0.824	0.814	0.825	0.824
	Variation in structural impact	-	+21.38	-20.89	-2.83	-4.03	-10.55
	Variation in ecosystem service provision	-	+21.61	-21.62	-3.73	-5.90	-12.11
	Rate of effective ecosystem service provision	-	0.23	-0.74	-0.90	-1.88	-1.57
	Variation in the capacity for ecosystem service provision	-	+1.00	-0.66	-1.22	+1.33	-0.12
	Variation in ecosystem service mitigated impact	-	+19.36	-14.34	+4.46	+10.00	-0.54
	Structural impact	435 879	529 064	418 562	406 726	390 345	349 166

(a) Data proportionally estimated according to the available datasets from the Portuguese national statistics institute at the municipality and civil parish levels.

[1] Reference period 1955

[2] Reference period 1989

[3] Reference period 1999

[4] Reference period 2009

Land cover dynamics (Fig. 3.4) are more pronounced than the trends identified for land use change. For example, the most dominant land cover class (i.e. silvo-pastoral areas with >50% of tree cover) decreased by almost half between 1951 and 2012, from 41.7% of the area to just 24.8%, maintaining only 48.8% of the original area. This change resulted from a reduction of tree cover densities and from the substitution of these silvo-pastoral areas by permanent pastures. By contrast, there was also an increase of silvo-pastoral areas with dense tree cover (i.e. with >50% of tree cover) in 11.3% of the changed area, which occurred mostly between 1951 and 1969.

Between 1951 and 1995 the rate of land cover change is nearly double the one resulting from land use change, and it triples between 1995 and 2012. This reveals an impressive change dynamic, which manifests itself in all land cover classes at different rates. These landscape dynamics appear to be related to an intensification and increased frequency of the agricultural policies that pushed for more intensive land uses.

At the landscape scale, there is an increase in fragmentation, reflected in a 67% increase in the number of patches and a 36% decrease in mean patch size between 1951 and 2012 (Table 3.4). For both indicators the rate of change was more intensive between 1951 and 1969 and between 1986 and 2004, with a peak between 1986 and 1995. After 2004 it seems to have halted, which is also manifested in the land use and land cover dynamics.

3.3.2 Impacts and ecosystem service provision

The selected impact indicators illustrate two different realities. Drier climate conditions between 1986 and 2012 lead to a reduced rainfall erosivity and a 19.9% decrease in the total soil erosion impact when no ecosystem service is provided (Υ). By contrast, the remaining soil erosion after the ecosystem service provision (β_e) increased by 16.8% due to a decrease in tree cover (and consequently in the capacity for ecosystem service provision) in areas with high structural impact (Υ).

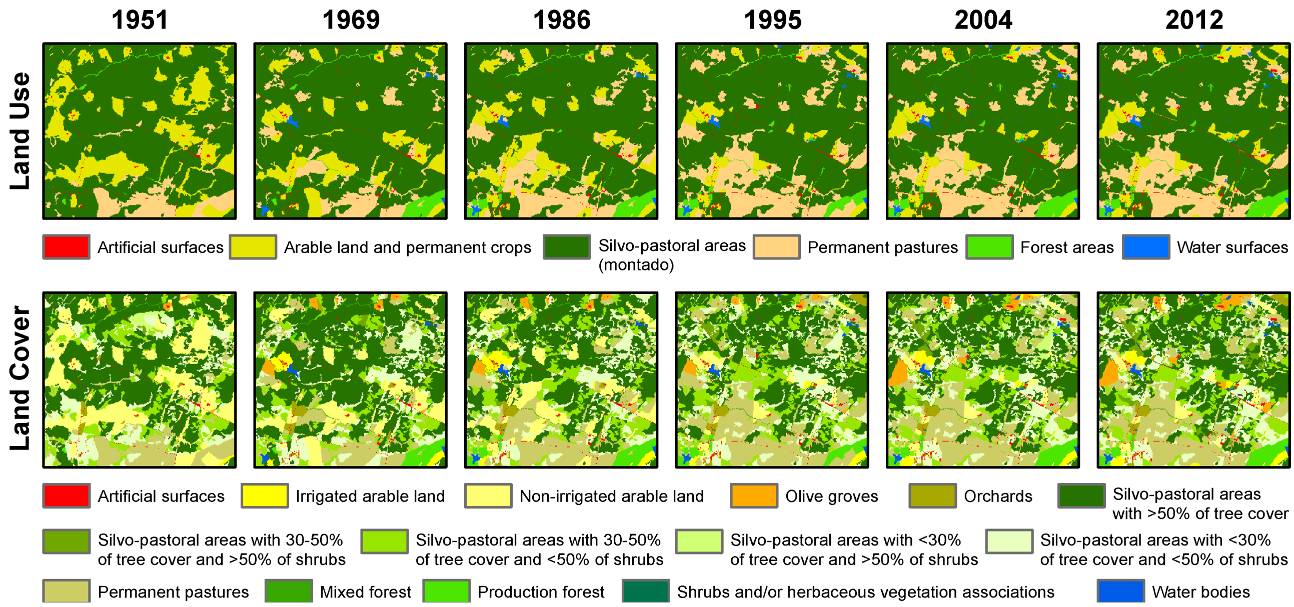

Figure 3.4 Spatial and temporal representation of the land use and land cover dynamics in the selected case study area (the maps are provided at the 1:320 000 scale).

This divergent trend between the structural impact and the ecosystem service mitigated impact reflects not only the shift in peak precipitations from spring months to October, but also the spatial and temporal landscape dynamics that resulted in a spatial rearrangement of the capacity for ecosystem service provision followed by a decrease in the effective soil erosion prevention provision (Fig. 3.5). This results in a reduction of the rate of effective ecosystem service provision, revealing the extent by which the land use intensification, with disregard for the local environmental conditions, is diminishing the capacity for ecosystem service provision in late summer, allowing it to be surpassed by increasing rainfall erosivity values in September and October.

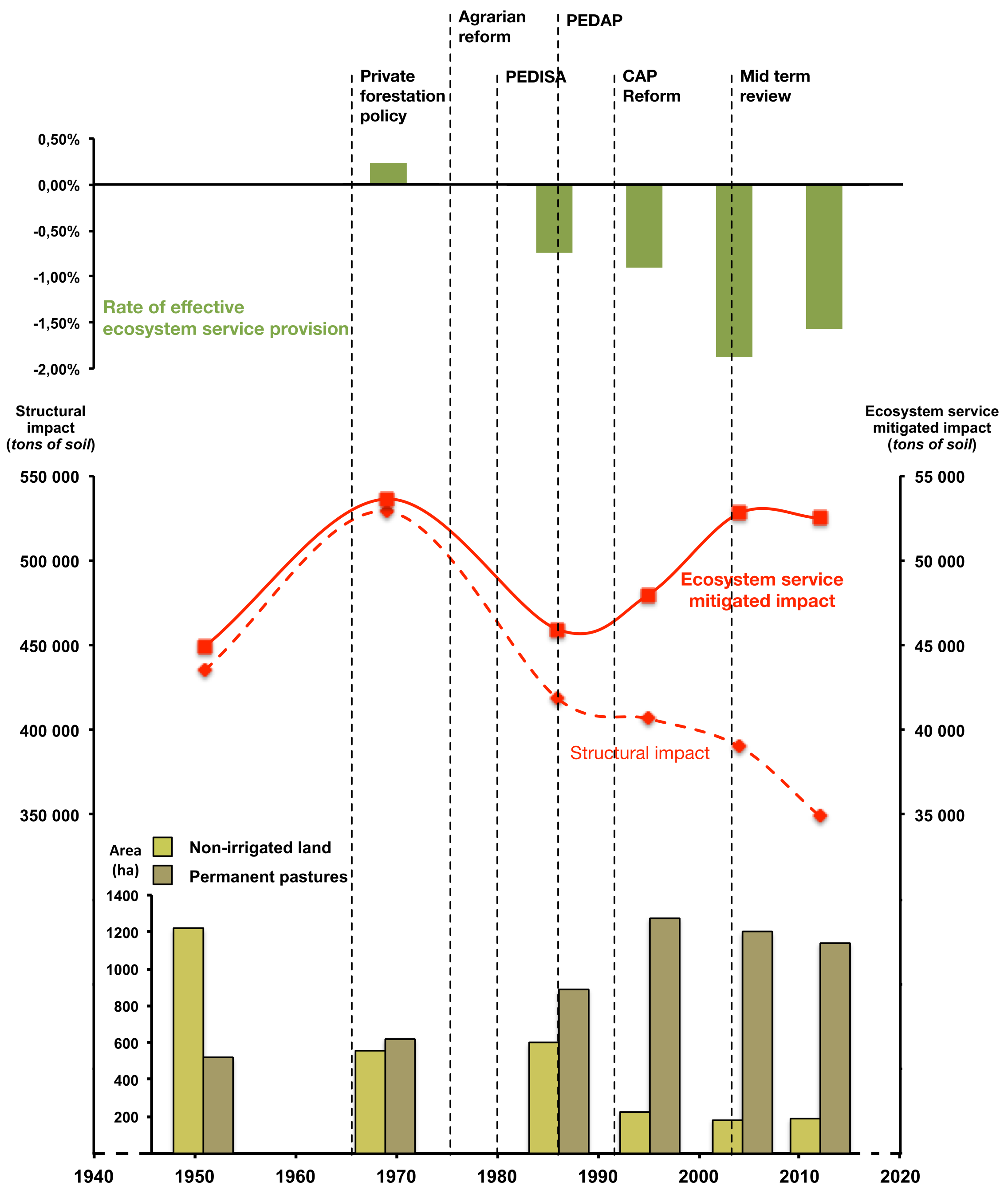

Figure 3.5 Representation of the temporal dynamics of the total sum of the structural impact and of the ecosystem service mitigated impact in relation to the implemented policies, variations in significant land cover classes and to the rate of effective ecosystem service provision.

While the capacity for ecosystem service provision ultimately determines the amount of ecosystem service provision, the results show that this indicator, when calculated at the landscape scale (Table 3.4), can have very low sensitivity to the changes that are occurring in the area. It also presents little relation to the values of the actual ecosystem service provision (r=0.433, p=0.391), i.e. areas with high capacity for ecosystem service provision often have a small effective provision and vice versa. This is backed up by the fact that although the capacity for ecosystem service provision has a very small variation over time, the actual ecosystem service provision has very dynamic values due to its correlation (r=0.999, p=0.000) to the variations in the structural impact indicators.

Despite this correlation, the rate of effective ecosystem service provision indicator shows that the ecosystem service provision is decreasing in the region, particularly after 1986. This coincides with the implementation of more productivist-oriented policies and with a more rapidly changing policy environment (Fig. 3.5). In fact, the rate of effective ecosystem service provision is highly correlated with changes in the land cover (r=0.966, p=0.008) and use (r=0.963, p=0.009). These results illustrate how this indicator provides a more clear representation of the dynamics of ecosystem service provision, which presents lower correlation values (with land cover [r=0.651] and land use [r=0.592]) changes, resulting in a suitable indicator to detect impact changes.

3.4 Discussion

3.4.1 Discussion of results

The results obtained in this Chapter confirm those described by Guerra et al. (2014), which stated that the spatial relation and configuration of the capacity for ecosystem service provision are more important than the capacity value itself. They illustrate how ecosystem service provision is dependent on the spatial and temporal arrangement of the interacting factors (Nelson et al. 2009; Eigenbrod et al. 2010b) by showing a significant correlation between the ecosystem service indicators and the identified drivers of change (Appendix 1). Finally they highlight how agricultural policy induces change both in land cover and in land use with significant implications for the actual ecosystem service provision and the mitigation of important impacts. An example of this relation is the rapid increase of permanent pastures and the decrease in tree density in most silvo-pastoral areas after the introduction of grazing intensification support schemes in the 1980s. This resulted in an increase of the *ecosystem service mitigated impact* and consequent decrease in the *rate of effective ecosystem service provision*.

The results show that service provision and the remaining impacts accompanied the same trend registered for the structural impact following a temporally stable agricultural policy and the implementation of private forestry promotion measures between 1951 and 1980. The same does not appear to occur in recent years, particularly between 1986 and 2012 (Fig. 3.5). While early policy measures implied significant changes in land use/cover (i.e. with an increase of cropland areas), the more gradual land cover change in recent years appears to reflect the policy decision to support cattle production and therefore indirectly support an increasing grazing intensity and pressure on the pasture areas. This results in little changes in land use patterns but with unintended policy impacts related to dramatic effects on ecosystem service provision.

In fact, the effect of these policy options seems to be very significant for soil erosion prevention (Pinto-Correia and Godinho 2013). This is reflected in the decay in tree regeneration (e.g. due to harder growth conditions to the young tree shoots), low tree replacement rates for adult trees, the presence of progressively larger openings in the montado tree cover and thus a higher fragmentation and a progressive decrease of the montado total area. Even if dramatic in the long run, these cattle support payments result in a slower landscape change dynamics rather than measures that produce more drastic effects in the land cover (e.g. subsidizing cereals production or the installation of new forest areas). This affects people's perception and reaction to landscape change and results in more critical situations as the increasing agriculture intensity demands for better and more stable soils that can cope with high productivity rates (Jose 2009; Podmanicky et al. 2011). In this case, the SEP service becomes more important as it substantially contributes to prevent soil degradation in immediate and future conditions (Shakesby 2011; Rickson 2014).

3.4.2 Discussion of methods

The implemented methodological approach resulted in a significant volume of land cover, land use, and ecosystem service indicators. This allowed to establish relations between the identified policy measures/drivers and the dynamics of ecosystem service provision and land cover/use change.

The aim of the Chapter is not to give an estimate of soil erosion but rather to give an estimate of the service being provided by vegetation cover. To avoid adding unnecessary complexity to the proposed framework we opted for a more simplified model to serve as the basis for ecosystem service provision calculations (Guerra et al., 2014). This had in consideration its successful application for soil erosion modelling in the Mediterranean (e.g. Diodato and Bellocchi 2007; Capolongo et al. 2008; de Vente et al. 2008), and the desire to use a parsimonious model providing spatially and temporally explicit results. This stylised and simple approach can be further expanded with more sophisticated models and better data if available.

There are still some caveats related to the assumption that soil conditions and vegetation profiles within the same land cover class remain stable through time. This was mainly due to data constrains related to the lack of an updated national soils database with consistent laboratorial methods (the existent one dates back to the 1960s (Cardoso 1965) and recent ones do not follow the same methodological procedures) and to insufficient satellite images covering the entire temporal scope presented. In both situations we acknowledge that they can introduce variation in the final values, but we do not consider that these differences influence the overall trends and outcomes.

By describing an extensive set of indicators for the different steps of the conceptual model it was possible to successfully determine important interactions between the identified policy measures and the provision of soil erosion prevention. Also, the use of several indicators rather than only one (commonly the capacity for ecosystem service provision) proved to be more effective to describe the systems dynamics. As results show, the use of only one indicator (i.e. the capacity for ecosystem service provision) would show a slight increase in ecosystem service provision, while the use of an extensive set of indicators showed a very different result. By assuming causal effects between policy mechanisms and soil erosion prevention provision, the implemented framework also revealed to be effective in detecting small variations resulting from the intensification or maintenance of the same policy measures trough time. Regarding the identified relations between policies and landscape/ecosystem service provision change, we are aware that there are other interactions that are not accounted for but considering the social, ecological and environmental characteristics of the area, we are certain that agricultural policy mechanisms are still the main driver for this area as shown by other studies (cf. Costa et al. 2009; Santana et al. 2013; Ribeiro et al. 2014).

3.4.3 Future applications

This Chapter emphasizes the need for place-based or at least context based policy design with a strong consideration for the environmental constrains and potentials of each region. Considering the agricultural policy as the main driver of change in rural areas (Swaffield and Primdahl 2010) the lack of system-driven policies that consider the whole montado system and not just specific productions is evident.

The approach used here demonstrated how unintended changes in the overall land use and impacts in ecosystem service provision are a clear result of the existing policy scheme (Pinto-Correia et al. 2013; Almeida et al. 2013; Ribeiro et al. 2014; Schröder 2011). Although some policy measures related to environmental sustainability were introduced in the last decade (e.g. the introduction of the second pillar in the CAP, the effects of the 2004 mid term review, or the current greening measures), they are often minimized by the way and extent to witch they are implemented (Pe'er et al. 2014). Therefore agricultural policy design should use integrated approaches as the one proposed in this Chapter coupled with prospective scenario building, in order to apply ex-ante assessments of policy measures and better target the implementation of environmental protection schemes.

These results show promise in the use of ecosystem service indicators as policy indicators, not only to monitor but also to evaluate their effective impacts. The use of soil erosion prevention in the evaluation of Mediterra-nean agricultural and environmental policies is urgent not only considering climate change (i.e. with the shift of precipitation concentration from winter and spring to the end of the summer) (Zhang and Nearing 2005; Metzger et al. 2008a), but also current and future social-economic realities (e.g. aging population, abandonment of rural areas, and the introduction of new land uses) (Pinto-Correia and Mascarenhas 1999; Pinto-Correia and Godinho 2013). Soil erosion prevention Pan-European assessments focused in Mediterranean areas are therefore needed to overcome the challenges stated by the European Thematic Strategy for Soil Protection (CEC, 2006).

Likewise, other regulating services can and should be considered to develop tradeoff scenarios for policy design. This could be made with an *a priori* identification of the relevant ecosystem service providers for each ecosystem service considered. Afterwards, appropriate methods to determine the structural impact had to be identified, and the spatial distribution of the capacity for ecosystem service provision correctly determined for each ecosystem service. This allows to avoid double counting situations and an integrated calculation of ecosystem service provision. Coupled with an effective characterization of the social-ecological dynamics, this would permit a better identification of current demand and also to improve ecosystem service provision awareness among ecosystem service beneficiaries (e.g. local and regional land managers).

4

LINKING FARM MANAGEMENT AND ECOSYSTEM SERVICE PROVISION: CHALLENGES AND OPPORTUNITIES FOR SOIL EROSION PREVENTION IN MEDITERRANEAN SILVO-PASTORAL SYSTEMS

Based on the manuscript:
Guerra, C.A., Pinto-Correia, T., 2015. Linking farm management and ecosystem service provision: challenges and opportunities for soil erosion prevention in Mediterranean silvo-pastoral systems. Land Use Policy. [Accepted]

Abstract

At both local and landscape levels, farm management is the main driver of land cover change influencing ecosystem functions, processes and traits. In Mediterranean large-scale silvo-pastoral systems these changes can have serious implications in the provision of valuable ecosystem services. Current ecosystem service assessment, mapping and valuation are still focussed in representing the state and trends of ecosystem service provision, often missing the link to actual farm management and farm management systems. We propose an approach that, at the farm level, combines the classification of farm management systems with indicators of ecosystem service provision. This is illustrated for soil erosion prevention, a key ecosystem service in mitigating current and future impacts in Mediterranean regions and the proposed approach is tested in Southern Portugal. We characterize thirty-eight large-scale farm management units (FMU) regarding their management system and environmental traits. Each FMU was then classified according to their management system and a set of ecosystem service indicators was calculated. To classify the FMU, data on livestock composition and grazing density, pastures, and soil mobilization practices were object of a cluster analysis and the result was tested against a set of ecosystem service indicators. The results highlight the implications and challenges for the provision of soil erosion prevention under different farm management systems and draw a clear relation between more intensive management practices and the degradation of service provision. Our results can also be used to support land management and policy design through the definition of intensity thresholds that consider the local environmental and ecological conditions.

4.1 Introduction

Mediterranean silvo-pastoral systems like the *montado* (Portugal) and the *dehesa* (Spain) are dynamic social-ecological systems with a high diversity of management and ecological conditions (Pinto-Correia 2000). They are characterized by open formations of cork and holm oaks in varying densities, combined with a rotation of cereal crops and fallow, and pastures which can be natural, improved or cultivated (Castro et al. 2010). These social-ecological systems are estimated to cover an area of about 3.5–4.0 M ha in the South-western Iberian Peninsula, thus being of large relevance in the Southern European context (Pinto-Correia et al. 2011).

Both socially and economically these are key social-ecological systems that significantly contribute for the regional food production (i.e. meat based outputs; Nunes 2007) and goods extraction (e.g. wood, cork; Ribeiro et al. 2004; Costa et al. 2014a), as well as the production of externalities as mushrooms and honey. Due to their extensive character and heterogeneous land cover, these systems are rich in game, and traditionally they have been used as highly valued hunting ground (Bugalho et al 2011). Recently many other leisure and touristic activities are taking place in the *montado*, as bird watching, hiking, bicycle tours, horse riding, landscape painting, etc. (Surová and Pinto-Correia 2008; Acácio and Holmgren 2012). From the ecological point of view, these systems play an important role on the maintenance of most fundamental ecosystem functions, and are considered as hotspots of both the regional and European farmland biodiversity (Pinto-Correia et al. 2011; Bugalho et al. 2011; Pereira et al. 2012b). The extensive land use result in a heterogeneous mosaic creating specific habitats that support a high species diversity (Pinto-Correia 2000; Castro et al. 2010), as well as important ecosystem services (Bugalho et al. 2011; Almeida et al. 2013; Ribeiro et al. 2014; Guerra et al. 2014). The adaptive human intervention that allows to maintain the balance between the different components of the system is fundamental in preserving this multifunctionality (Moreno et al. 2014; Almeida et al. 2015). The *montado* is thus a system that cannot be fully understood if not conceived as a socio-ecological system (McGinnis and Ostrom 2014). Its diversified combination of management and ecological traits also justifies the classification of the *montado* as High Nature Value (HNV) farming system (Paracchini et al. 2008; Almeida et al. 2013; Lomba et al. 2014).

As a consequence of a progressively globalized market, policy mechanisms that only target specific system components (e.g. livestock production support), and of the industrial and social-demographic changes of the last decades (Schröder 2011; Jones et al. 2011b), the *montado* multifunctional land use practices are being gradually transformed into more specialized systems (Pe'er et al. 2014; Ribeiro et al. 2014). This has been a gradual change in recent years and the implications of these transformations for ecosystems, for the landscape, but also for the long term sustainability of the system are still little understood (Castro et al. 2010; Costa et al. 2011; Acácio and Holmgren 2012). In these silvo-pastoral systems the most significant transformations are related to: a) an increase of livestock density, even when keeping livestock outside the all year round; b) changes in the type of cattle breed, from endogenous light and little demanding breeds to widely used more heavy and demanding breeds; c) the introduction of new, often more intensive, shrub cleaning methods (e.g. soil disking, soil mobilization); and d) the expansion of fodder crop areas and irrigated pastures (Pinto-Correia and Vos 2004; Almeida et al. 2015).

The consequences of the changing farm management practices are reflected in the increase of several environmental degradation processes (Costa et al. 2014b; Godinho et al. 2014). This is for example reflected in a higher fragmentation at the landscape level, in the reduction of vegetation cover in late summer critical periods, and in an increase of soil degradation processes, including increasing desertification, compaction, soil erosion, and decreasing soil fertility (Costa et al. 2010; Costa et al. 2014b; Godinho et al. 2014). These changes ultimately result in less resilient ecosystems that are more sensitive to changes in environmental conditions (e.g. an increase on rainfall intensity; Alpert et al. 2002).

In this context, to produce information that can support management decisions in these complex systems, the assessment of ecosystem services (ES) provision, particularly regulating services, is key to evaluate and monitor their sustainability (Acácio and Holmgren 2012; Almeida et al. 2013). Regulating services are related to the main ecosystem processes associated to impact mitigation from current and future environmental hazards (Brink et al. 2012; Santos-Martín et al. 2013).

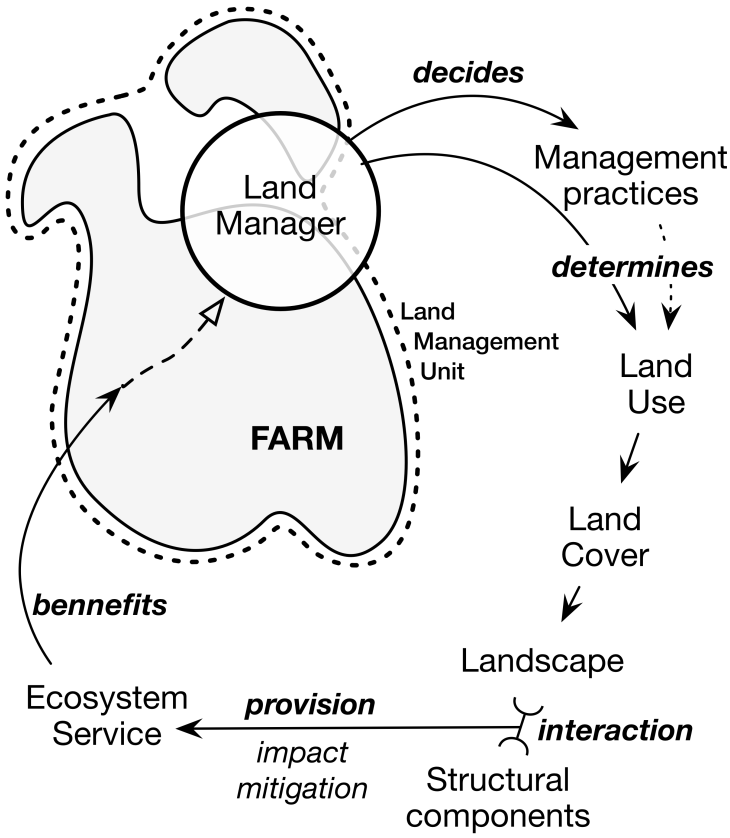

Figure 4.1 Conceptual framework of ecosystem regulating service provision, as dependent on both the local biophysical structure and the farm management (adapted from Guerra et al., 2014). The farm manager takes decision as to the orientation of its farm management practices and these options therefore determine the land use. In turn the land use shapes the local land cover and contributes, at a larger scale, to the local landscape. The interaction between this landscape and the local biophysical and climatic conditions ecosystem services are provided, and ultimately they benefit the farm and consequently the farm manager that produces new management decisions.

At the same time, they contribute to local biodiversity and habitat conservation (Fitter et al. 2010; Mace et al. 2012). The provision dynamics of this group of ecosystem services can be influenced by a number of factors, namely the presence and intensity of environmental impacts, the existence and extension of the ecosystem service providers, and the farm management system implemented, which in turn influences and it is influenced by the ecosystem service providers (Pinto-Correia and Godinho

2013; Pinto-Correia and Kristensen 2013) (Fig. 4.1). The farm management system can determine the effective ecosystem service provision by minimising or potentiating the *capacity for ecosystem service provision* in a given place and time. These changes are particularly relevant for the *montado* silvo-pastoral systems where the overall production system can be the same, while there are significant differences in its level of intensity and specialisation (Pinto-Correia et al. 2011; Almeida et al. 2015). This is the result of more or less integration of environmental mitigation and conservation awareness and practices in management options (e.g. in the case of soil erosion prevention by promoting a better vegetation cover in heavy rain periods).

At the same time, the main beneficiary of the ecosystem service provision is the farm manager, who benefits from better conditions (e.g. better soil fertility, increased pollination) for its options of land use (e.g. agricultural production) (Darnhofer 2014). This cycle of benefits is often interrupted or degraded by farm management practices directed specially at increasing short-term production, which in turn are most often driven by sectoral policies and increased globalized markets pushing for short term competitiveness.

Considering the importance of soil erosion prevention (SEP) in European Mediterranean systems (c.f. Van-Camp et al. 2004) we will critically examine the influence of farm management on the provision of SEP by vegetation cover in selected Mediterranean silvo-pastoral social-ecological systems. Our objective is to demonstrate and quantify the relations between the ecosystem service provision and farmers management options. This will allow to discuss the main management and policy challenges associated with the provision of the SEP in Mediterranean silvo-pastoral social-ecological systems.

4.2 Methods

The relation between farm management and ecosystem service provision can only be correctly attained at the farm level allowing to grasp the effects of the decisions made by each farmer (Darnhofer et al. 2010; Pinto-Correia and Kristensen 2013; Darnhofer 2014). This implies that farm management data has to be collected and compiled at this level, allowing to define the whole farmland area under the same farm manager as a farm management unit (FMU), which can then be related to the provision of specific ecosystem services. Based on a survey of *montado* farm managers in the Alentejo region, Portugal (e.g. Almeida et al. 2013; Almeida et al. 2015), a set of thirty-eight FMU belonging to similar environmental conditions was selected to perform the analysis (Fig. 4.2; Table 4.1).

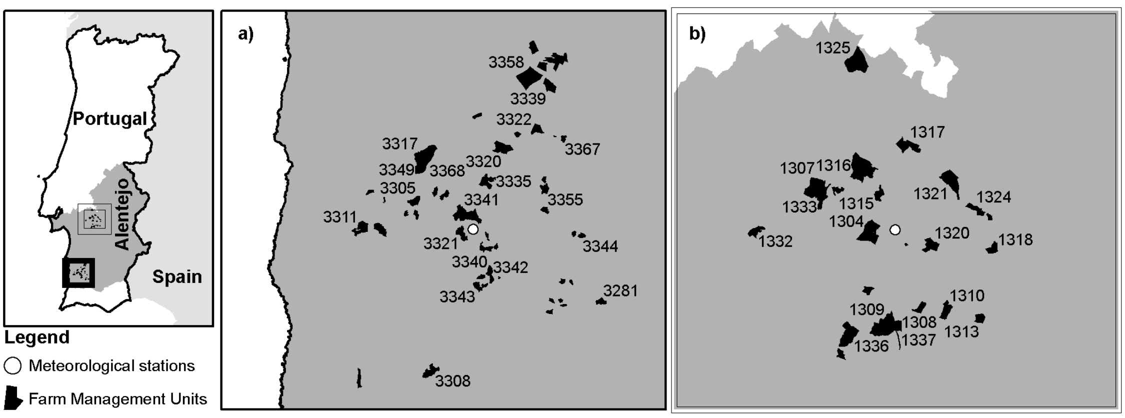

Figure 4.2 Geographic location of the selected thirty-eight FMU.

These are all FMU with more than 50 ha and are characterized by having a predominance of *montado* land cover. They are differentiated by the type and intensity of grazing (mainly related to cattle and sheep), shrub control practices (mainly shrub cutting and soil disking), and the type of soil fertilization implemented (mainly organic fertilization or with bio-chemical inputs). The FMU may contain more than one plot but all results were computed at the FMU level to better grasp the relation between farm management and ecosystem service provision.

Our general approach was to first classify the FMU according to different farm management types and then to compute for the *montado* areas a set of ecosystem service indicators related to SEP. For this step the *montado* areas were mapped through photointerpretation of aerial photographs. This allowed to relate both the farm management types and the corresponding provision of SEP and to determine if more intensive and less diversified farm management practices relate to less efficient SEP provision in *montado* areas.

4.2.1 Farm management classification

According to the management practices used in each FMU a hierarchical cluster analysis was performed based on Ward's method and a squared Euclidean distance measure. The identified farm management clusters were described in order to define the related farm management type. With this procedure we intended to separate FMU with more or less intensive farm management practices in order to relate them with the ecosystem service indicators produced for SEP.

4.2.2 Ecosystem service provision: estimation and classification

The conceptual approach for mapping and assessment of regulating services used in this Chapter has recently been described by Guerra et al. (2014), and it is summarised here. Ecosystem regulating services are provided at the interface between the structural components of the social-ecological system and its landscape dynamics (Guerra et al. 2014) (Fig. 4.1). Therefore, the provision of SEP results in the mitigation of the potential soil erosion generated in the context of a specific ecosystem. A complete review of the methodological and conceptual frameworks is given in Guerra et al. (2014) and more details are provided in the following sections.

Table 4.1 Main characteristics of the selected farm management units (FMU).

ID	FMU area (ha)	Montado (%)	Stocking density (heads/ha)	Proportion of cattle (%)	Proportion of sheep (%)	Type of grazing animals	Shrub control techniques	Fertilization	Food Supplements (%)
1304	445.25	85.34	0.53	100.0	0	C	Grazing	Organic	0
1307	74.88	85.31	1.30	100.0	0.0	C	Shrub cutting	Chemical	>50
1308	162.81	52.62	1.21	100.0	0.0	C	Shrub cutting	Chemical	0
1309	285.04	61.68	0.87	100.0	0.0	C	Chemical	Organic	<25
1310	179.11	61.50	1.74	67.8	32.2	CS	Shrub cutting	No fertilization	0
1311	502.30	73.55	1.39	70.7	0.0	CO	Shrub cutting	Chemical	<25
1313	209.63	64.40	0.86	100.0	0.0	C	Soil disking	Organic	<25
1315	116.93	68.41	0.98	100.0	0.0	C	Grazing	Chemical	<25

Table 4.1 Main characteristics of the selected farm management units (FMU) (Cont.).

ID	FMU area (ha)	Montado (%)	Stocking density (heads/ha)	Proportion of cattle (%)	Proportion of sheep (%)	Type of grazing animals	Shrub control techniques	Fertilization	Food Supplements (%)
1316	654.83	61.84	0.91	100.0	0.0	C	Grazing	Organic	<25
1317	269.09	82.76	0.00	0.0	0.0	-	Soil disking	Organic	25-50
1318	125.54	68.71	1.24	93.7	6.3	CS	Chemical	Chemical	<25
1320	208.78	76.77	1.62	76.9	23.1	CS	Chemical	Chemical	<25
1321	375.11	59.34	1.34	100.0	0.0	C	Grazing	Organic	<25
1324	166.28	91.08	0.70	100.0	0.0	C	Soil disking	Chemical	<25
1325	544.25	94.21	0.52	71.7	0.0	CO	Soil disking	Organic	0
1330	1030.33	63.56	0.96	90.4	9.6	CS	Soil disking	Chemical	>50
1332	138.66	62.70	2.41	40.6	59.4	CS	Shrub cutting	Chemical	<25
1333	605.64	83.27	0.32	98.8	1.2	CS	Soil disking	Chemical	>50
1336	547.63	74.56	0.50	97.8	2.2	CS	Shrub cutting	Chemical	<25
1337	222.21	49.02	1.71	100.0	0.0	C	Shrub cutting	Organic	<25
3281	192.14	61.27	0.15	0.0	100.0	S	Shrub cutting	No fertilization	>50
3305	183.06	47.87	0.61	83.1	0.0	CO	Soil disking	Chemical	<25
3308	231.02	53.68	0.15	100.0	0.0	C	Soil disking	No fertilization	<25
3311	314.37	56.32	0.27	100.0	0.0	C	Soil disking	Organic	0
3317	202.34	72.81	1.06	100.0	0.0	C	Soil disking	Chemical	25-50
3320	221.61	64.19	0.23	0.0	90.9	S	Soil disking	Chemical	<25
3321	124.81	58.52	0.91	100.0	0.0	C	Soil disking	Organic	<25
3322	110.45	59.25	0.34	0.0	100.0	S	Soil disking	Organic	<25
3335	168.15	53.32	0.39	100.0	0.0	C	Soil disking	Chemical	25-50
3339	302.85	65.04	0.00	0.0	0.0	-	Soil disking	No fertilization	0
3340	118.94	59.95	0.29	0.0	100.0	S	Soil disking	No fertilization	25-50
3341	436.05	50.90	0.27	44.1	5.1	CO	Soil disking	Organic	<25
3342	101.67	51.64	0.98	82.5	14.6	CO	Soil disking	No fertilization	>50
3343	127.30	56.28	0.38	100.0	0.0	C	Soil disking	No fertilization	<25
3344	60.31	54.81	0.00	0.0	0.0	-	Soil disking	No fertilization	0
3349	72.57	66.93	0.62	100.0	0.0	C	Soil disking	No fertilization	>50

Note: C: Catle ; S: Sheep; O: Other (goats or pigs)

This conceptual framework is based on a set of functional components, here also seen as indicators of ecosystem service provision (Table 2): i) the *structural impact* (ϒ); ii) the *capacity for ecosystem service provision* (e_s); iii) the *actual ecosystem service provision* (E_s); and iv) the ecosystem service *mitigated impact* (β_e)). In total we used a set of eight indicators that describe the different processes that contribute to SEP and allow to compare the ecosystem service provision of *montado* areas between FMU and farm management types (Table 4.2).

4.2.2.1 Structural impact

Following this assessment framework, in order to map and to quantify SEP it is necessary to first identify the *structural impact* (ϒ) related to soil erosion, i.e. the erosion that would occur when vegetation is absent and therefore no ecosystem service is provided (Guerra et al. 2014). It determines the potential soil erosion in a given place and time and it is considered to be related to rainfall erosivity (i.e. the erosive potential of rainfall), soil erodibility (as a characteristic of the soil type) and local topography (Ribeiro et al. 2004; Panagos et al. 2011b).

These parameters were calculated based on the approach implemented for the Universal Soil Loss Equation (Wischmeier and Smith 1978) to estimate rainfall erosivity (R Factor), soil erodibility (K Factor), and local topography (LS Factor). The *structural impact* (ϒ) was then calculated using the following expression (Prasuhn et al. 2013; Guerra et al. 2014):

$$\Upsilon = R \times K \times LS \qquad \text{(Eq.4.1)}$$

The rainfall erosivity parameter was calculated using the rainfall erosivity model for complex terrains (REM_{DB}) proposed by Diodato and Bellocchi (2007), which was found to give good results using monthly rainfall data. It gives suitable results when compared to other simplified models that do not account for the interaction with elevation and latitude and it follows a non-linear equation to estimate monthly rainfall erosivity (R_m) (Diodato and Bellocchi 2007):

$$R_m = 0.207 \times \left[p_m \times \left(f(m) + f(E,L) \right) \right]^{1.561} \qquad \text{(Eq.4.2)}$$

where, p_m (mm month^{-1}) is the monthly average precipitation amount for month m, $f(m)$ is a sinusoidal monthly function, $f(E, L)$ is a parabolic function reflecting the influence of site elevation (E) in meters and latitude (L) in degrees on the erosivity. The $f(m)$ function varies with month (m: 1 (January), ...,12 (December)) (Yu et al. 2001; Davison et al. 2005) following the expression:

$$f(m) = 0.3696 \times \left[1 - 1.0888 \times \cos\left(2\pi \times \frac{m}{2.9048 + m} \right) \right] \qquad \text{(Eq.4.3)}$$

To calculate the f(E, L) function we used the following expression (Diodato and Bellocchi 2007):

$$f(E,L) = 0.3024 + e \times \sqrt{E} \times (41 - L) - g \times \left[\sqrt{E} \times (41 - L) \right]^2 \qquad \text{(Eq.4.4)}$$

where, E is the elevation in meters, L is the latitude in degrees, e is equal to 0.0013848, and g is equal to 0.0000138092. Available monthly rainfall data was obtained from reference meteorological stations of the Portuguese national water resources information system (INAG 2010).

For the soil erodibility parameter we used the high resolution (500 meters resolution) European soil erodibility map (Panagos et al. 2014b) calculated from data collected in the Land Use/Cover Area frame Survey (LUCAS) soil survey for 2009. This was calculated based on the equation proposed by Wischmeier and Smith (1978) and Renard et al. (1997) (Panagos et al. 2014b):

$$K = \frac{2.1 \times 10^{-4} M^{1.14} (12 - a) + 3.25 (s - 2) + 2.5 (p - 3)}{100} \times 0.1317 \qquad \text{(Eq.4.5)}$$

where, K corresponds to the soil erodibility factor, a is the percentage of organic matter, b the soil structure parameter, c the profile permeability class, and $M = (\%_{\text{silt}} + \%_{\text{very fine sand}}) \times (100 - \%_{\text{sand}})$.

Table 4.2 List of calculated indicators to describe the provision of soil erosion prevention.

Indicator	Description	Units
Structural impact	Total soil erosion impact when no ecosystem service is provided	tons of soil per FMU
Ecosystem service mitigated impact	Total of the remaining soil erosion after the ecosystem service provision	tons of soil per FMU
Actual ecosystem service provision	Total of the actual ecosystem service provision corresponds to the total amount of ecosystem service provided, measured in ecosystem service providing units (tons of soil not eroded). It varies from season to season and year-to-year depending on the variation of the structural impact	tons of soil per FMU
Ecosystem service provision capacity	Average fraction of the structural impact that is mitigated by the ecosystem service, it corresponds to an adimensional gradient from 0 to 1	-
Variation in structural impact	% variation in the total amount of structural impact between farm management unit	%
Rate of effective ecosystem service provision	% variation in the total amount of actual ecosystem service provision corrected by the structural impact fluctuations for a given time slice using the following expression: $100\times\left(\left[\frac{Es_{eval}}{Es_{ref}}-1\right]-\left[\frac{\Upsilon_{eval}}{\Upsilon_{ref}}-1\right]\right)$, where *Es* is the total actual ecosystem service provision, Υ is the total structural impact, *ref* corresponds to the reference farm management unit type, and *eval* to the evaluation farm management unit type	%
Variation in ecosystem service provision capacity	% variation in the total amount of ecosystem service provision capacity between farm management unit	%
Variation in ecosystem mitigated impact	% variation in the total amount of ecosystem service mitigated impact between farm management unit	%

FMU – farm management unit

For the topographic factor it was used a 30 meters resolution topographic dataset provided by the Portuguese National Geographic Institute and the expression proposed by Moore and Burch (1986):

$$LS=\left(\frac{a\times p}{22.13}\right)^{0.4}\times\left(\frac{\sin(d)}{0.0896}\right)^{1.3} \qquad \text{(Eq.4.6)}$$

where, *LS* represents the topographic factor, *a* refers to the flow accumulation model obtained from the topographic dataset, *p* to the pixel size, and *d* to the slope model in degrees.

4.2.2.2 Capacity for ecosystem services provision

The *capacity for ecosystem service provision* corresponds to the idea that ecosystems provide a certain potential for service provisioning based on their functioning (van Oudenhoven et al. 2012). This determines the fraction of the structural impact that is mitigated by the ecosystem service in a given place and time, and corresponds to an adimensional gradient from 0 to 1 (Guerra et al. 2014). To consistently map the capacity for soil erosion prevention in Mediterranean Europe, we will estimate the *capacity for ecosystem service provision* by computing vegetation cover. Vegetation cover was computed using the relation between the Normalized Difference Vegetation Index (NDVI; calculated from MODIS 16 days NDVI composites with a 250 meters pixel resolution) and the USLE C Factor (Wischmeier and Smith 1978) proposed by Van der Knijff *et al.* (1999, 2000):

$$C = \exp\left[-a \times \frac{NDVI}{(b - NDVI)}\right] \quad \text{(Eq.4.7)}$$

where, a=2 and b=1.

Finally, the *capacity for ecosystem services provision* was estimated using the following expression:

$$e_s = 1 - \alpha \quad \text{(Eq.4.8)}$$

where, e_s is the *capacity for ecosystem service provision* and α corresponds to the C Factor from USLE (Guerra et al. 2014).

4.2.2.3 Actual ecosystem services provision and ecosystem services mitigated impact

The *actual ecosystem service provision* (*Es*) is the fraction of the total potential soil erosion (i.e. *structural impact*: ϒ) that is regulated by the ecosystem service providers (in this case vegetation). It is determined by the intensity of the *structural impact* and by the *capacity for ecosystem service provision* (e_s) in a given place and time following the expression:

$$E_s = \Upsilon - \beta_e \quad \text{(Eq.4.9)}$$

where, E_s is the *actual ecosystem service provision*, ϒ is the *structural impact*, and β_e is the *ecosystem service mitigated impact*.

Due to this relation, studies show (Guerra et al. 2014; Guerra et al. 2015) that, for regulating services, there is a significant overlap between the magnitude and the spatial and temporal distribution of the *structural impact* and of the *actual ecosystem service provision*. Considering this overlap, calculating the variation of the *actual ecosystem service provision* by itself does not provide relevant information for an ecosystem service assessment, as increases or decreases in ecosystem service provision largely depend on the variations of the *structural impact*. To overcome this problem Guerra et al. (2014) proposed the *rate of effective ecosystem service provision* to estimate variations of the *actual ecosystem service provision*. The *rate of effective ecosystem service provision* corresponds to the % variation in the total amount of *actual ecosystem service provision* corrected by the % variation of the total amount of *structural impact* for a given region. The expression was originally proposed to compare between time slices for the same region (Guerra et al. 2015), although here we will adapt it to compare between regions (i.e. FMU) for the same time slice using the following expression:

$$100 \times \left(\left[\frac{Es_{\text{eval}}}{Es_{\text{ref}}} - 1 \right] - \left[\frac{\Upsilon_{\text{eval}}}{\Upsilon_{\text{ref}}} - 1 \right] \right) \quad \text{(Eq.4.10)}$$

where, *Es* is the total *actual ecosystem service provision*, ϒ is the total *structural impact*, *ref* corresponds to the reference region, and *eval* corresponds to the evaluation region.

Finally, the *ecosystem service mitigated impact* corresponds to the remaining soil erosion after the regulation provided by the ecosystem service provider. It was determined by:

$$\beta_e = \Upsilon \times \alpha \qquad \text{(Eq.4.11)}$$

where, β_e is the *ecosystem service mitigated impact*, Υ is the *structural impact*, and α corresponds to the C Factor from USLE as determined in the previous Section.

Although it does not allow for an absolute measure of soil erosion, this mathematical formulation will generate a spatially explicit gradient of the potential soil loss and the related gradient of ecosystem service provided by vegetation cover (Guerra et al. 2014).

4.3 Results

4.3.1 Farm management units classification

The original dataset was classified into eight clusters. These clusters separate characteristics like the predominance of *montado* or pasture areas, the type and percentage of animal breed, and the type of shrub control practices or soil fertilization (Fig. 4.3). The multiple cluster divisions highlight the complexity of the farm management systems present in the FMU dataset. These range from no animal breed and crop production oriented FMU (Cluster 3), to high headage and meat oriented FMU (e.g. Cluster 6). It is important to state that all FMU have a predominance of *montado* areas (between 47.9% and 94.2%), which depending on headage and other management practices (e.g. soil fertilization, shrub control practices, existence of other crops) have higher or lower pressure from grazing.

The eight clusters (Cl) show a high array of farm management practices that illustrate different farm systems and can be summarized as (Table 4.3):

- Cl1. Cluster 1 represents one of the clusters with lower *montado* area in the total land cover (median=58.5%). Farm units here have no fodder production and low cereal production areas (median=6.4%). This cluster is also characterized by an above average headage (median=0.859) mostly composed of cattle. Within this cluster a mix of chemical and organic fertilization is promoted, although soil disking is used predominantly as a shrub control practice.
- Cl2. This cluster presents three major characteristics. The predominance of cattle, followed by no fertilization and the implementation of soil disking practices in all farm management units. This is also related to the absence of fodder crops and to the existence of higher levels external food supplements for animals, although headage is bellow average when compared with the total dataset.
- Cl3. This cluster is characterized by farm management units with no livestock (cattle and sheep) and a relatively high predominance of *montado* areas in the total farm area (median=65%). In terms of shrub control the FMU included in this cluster use only soil disking accompanied by low levels of soil fertilization. These farm management units are oriented to the production of food supplements for other farm management units, and cork.
- Cl4. From the farm management units that have livestock (cattle and sheep), this cluster presents the lowest headage values (median=0.288) with 100% of sheep breeding. Land cover it is predominantly characterized by *montado* areas (median=60%) and open pastures (median=23.7%). Soil disking and no fertilization also prevail as management practices in the tree under cover.

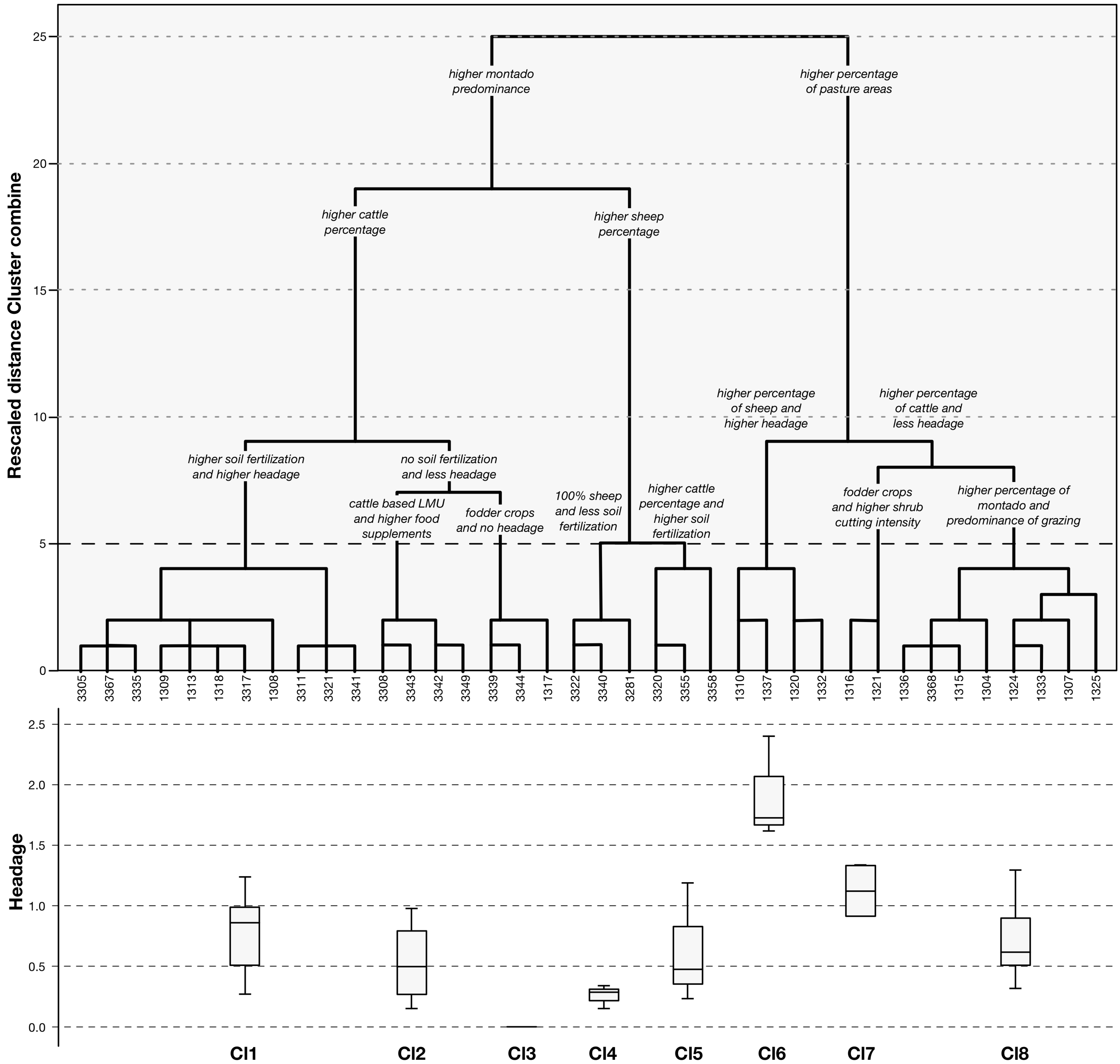

Figure 4.3 Hierarchical cluster classification of the farm management units identifying the main cluster separations in each cluster division. Cluster (Cl) numbering is identified in the bottom of the figure.

Cl5. This cluster is characterized by farm management units that have mixed livestock with a predominance of sheep and a below average headage (median=0.474). Soil disking is used in all farm management units and chemical fertilization is favoured. This, together with the absence of fodder crops, promotes the use of external food supplements that are predominantly between 25 and 50% of the total food available for animals.

Cl6. Cluster 6 aggregates the farm management units with the highest headage of all clusters (median=1.724) with a mixed livestock (cattle and sheep) with predominance of cattle. This cluster also presents the smallest pastures areas of all clusters. Although the headage is high, shrub control is made predominantly with shrub cutting practices and a mix of chemical and organic fertilization is also used.

Cl7. The farm management units present in this cluster show a relatively high headage (median=1.257) and are 100% cattle oriented. In this case a higher headage is related to the presence of fodder crops and also grazing and organic faming are the predominant management practices.

Cl8. The farm management units included in this cluster present a relatively small headage (median=0.617), mainly of cattle, although varying between 0.319 and 1.299. Mainly chemical fertilization is used with a mix of shrub cutting practices being applied, predominantly grazing and shrub cutting.

4.3.2 Soil erosion prevention response to farm management practices

The ecosystem service provision indicators were calculated for the *montado* areas of each FMU and then were aggregated at cluster level. The results show that ecosystem service provision indicators translate differences in farm management practices. This is evident not only for the management practices (Fig. 4.4) but also at the farm system level (Fig. 4.5).

As shown in Figure 4.4 different farm management practices combinations have a direct effect in the *rate of effective ecosystem service provision*. It is possible to observe that concerning shrub management practices, independently of the fertilization practices, more intensive shrub management practices (e.g. soil disking) always result in a degradation of the ecosystem service provision in the FMU. While organic fertilization seems to be related with higher ecosystem service provision efficiency, no fertilization always results in more negative outputs than other fertilization practices.

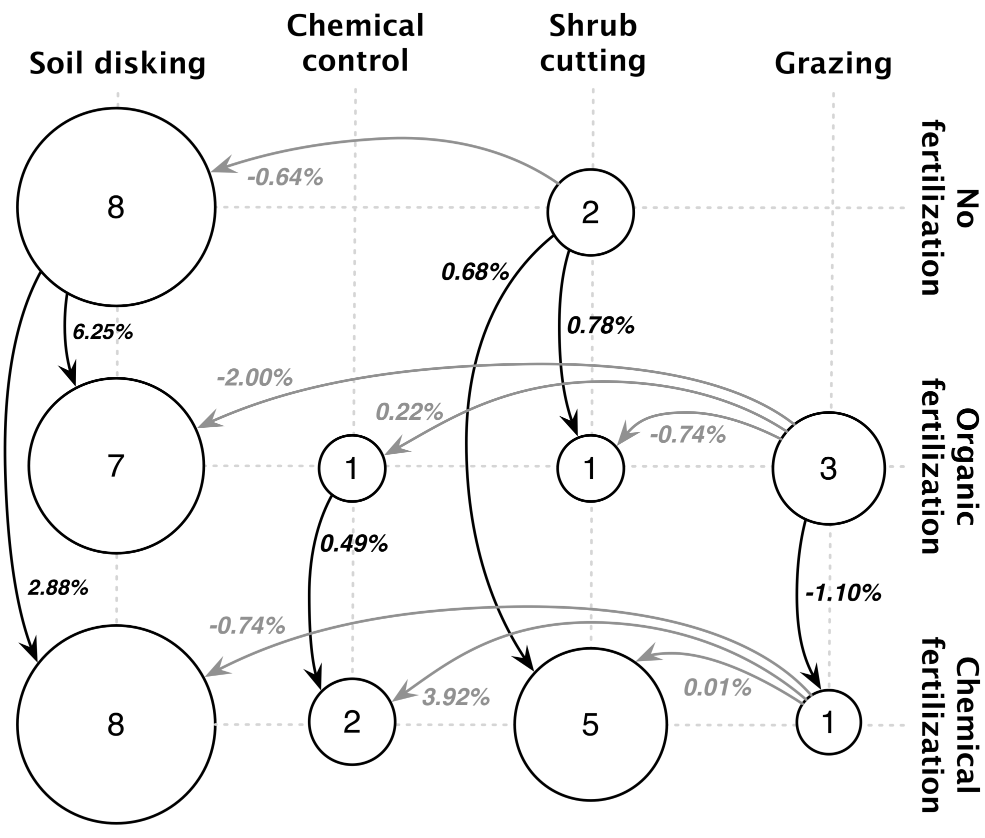

Figure 4.4 Effects on the rate of effective ecosystem service provision of a combination of different farm management practices: shrub management practices (on top) and fertilization type (on the right).

Table 3 Main characteristics of each identified cluster.

Clusters	Statistics	% of Montado	% of Cereal	% of Fodder crops	% of Pastures	Shrub control	Fertilization
Cluster 1	Minimum	47.9	0.0	0.0	3.5	Shrub cutting	Organic
	Maximum	72.8	14.4	0.0	38.9	Soil disking	Chemical
	Median (a)	58.5	6.4	0.0	24.6	Soil disking	Organic
	Std. Deviation	7.7	4.9	0.0	11.4	-	-
Cluster 2	Minimum	51.6	1.8	0.0	19.7	Soil disking	No fertilization
	Maximum	66.9	19.3	0.0	34.6	Soil disking	No fertilization
	Median (a)	55.0	9,9	0.0	26.9	Soil disking	No fertilization
	Std. Deviation	6.8	8.3	0.0	6.4	-	-
Cluster 3	Minimum	54.8	0.0	0.0	16.8	Soil disking	No fertilization
	Maximum	82.8	0.3	3.2	23.0	Soil disking	Organic
	Median (a)	65.0	0.0	0.0	19.2	Soil disking	No fertilization
	Std. Deviation	14.2	0.2	1.8	3.1	-	-
Cluster 4	Minimum	59.2	0.0	0.0	19.4	Shrub cutting	No fertilization
	Maximum	61.3	6.8	0.0	24.0	Soil disking	Organic
	Median (a)	60.0	6.1	0.0	23.7	Soil disking	No fertilization
	Std. Deviation	1.1	3.7	0.0	2.6	-	-
Cluster 5	Minimum	54.9	15.1	0.0	4.4	Soil disking	No fertilization
	Maximum	64.2	30.0	0.0	11.2	Soil disking	Chemical
	Median (a)	58.3	24.0	0.0	8.0	Soil disking	Chemical
	Std. Deviation	4.7	7.5	0.0	3.4	-	-
Cluster 6	Minimum	49.0	12.9	0.0	0.0	Shrub cutting	No fertilization
	Maximum	76.8	46.4	0.0	4.6	Chemical	Chemical
	Median (a)	62.1	27.3	0.0	3.5	Shrub cutting	Chemical
	Std. Deviation	11.4	14.0	0.0	2.1	-	-
Cluster 7	Minimum	59.3	1.0	0.0	19.5	Grazing	Organic
	Maximum	61.8	7.7	9.3	29.7	Grazing	Organic
	Median (a)	60.6	4.4	4.7	24.6	Grazing	Organic
	Std. Deviation	1.8	4.7	6.6	7.2	-	-
Cluster 8	Minimum	68.4	0.0	0.0	2.7	Grazing	Organic
	Maximum	94.2	5.3	0.0	21.4	Soil disking	Chemical
	Median (a)	84.3	0.9	0.0	10.8	(b)	Chemical
	Std. Deviation	8.4	1.8	0.0	6.0	-	-

(a) In the case of shrub control, fertilization and food supplements the statistic values correspond to the Mode (in grey).

(b) Multiple Modes exist.

Table 3 Main characteristics of each identified cluster (Cont.).

Clusters	Statistics	Food Supplements	Headage	% of Cattle	% of Sheep
Cluster 1	Minimum	0%	0.266	44.1	0.0
	Maximum	50-25%	1.238	100.0	14.3
	Median	<25%	0.859	100.0	0.0
	Std. Deviation	-	0.349	16.9	4.6
Cluster 2	Minimum	<25%	0.155	82.5	0.0
	Maximum	>50%	0.979	100.0	14.6
	Median	(b)	0.498	100.0	0.0
	Std. Deviation	-	0.353	8.8	7.3
Cluster 3	Minimum	0%	0.000	0.0	0.0
	Maximum	50-25%	0.000	0.0	0.0
	Median	0%	0.000	0.0	0.0
	Std. Deviation	-	0.000	0.0	0.0
Cluster 4	Minimum	<25%	0.152	0.0	100.0
	Maximum	>50%	0.344	0.0	100.0
	Median	(b)	0.288	0.0	100.0
	Std. Deviation	-	0.099	0.0	0.0
Cluster 5	Minimum	0%	0.232	0.0	60.3
	Maximum	50-25%	1.193	39.7	100.0
	Median	(b)	0.474	0.0	90.9
	Std. Deviation	-	0.500	22.9	20.8
Cluster 6	Minimum	0%	1.623	40.6	0.0
	Maximum	<25%	2.408	100.0	59.4
	Median	<25%	1.725	72.4	27.7
	Std. Deviation	-	0.362	24.6	24.6
Cluster 7	Minimum	<25%	0.914	100.0	0.0
	Maximum	<25%	1.339	100.0	0.0
	Median	<25%	1,127	100.0	0.0
	Std. Deviation	-	0.301	0.0	0.0
Cluster 8	Minimum	0%	0.319	71.7	0.0
	Maximum	>50%	1.299	100.0	2.2
	Median	<25%	0.617	99.4	0.0
	Std. Deviation	-	0.316	9.9	0.8

Looking at the FMU types level (Fig. 4.5) it is possible to observe these relations between farm management practices and SEP by comparing FMU types with specific differences between them. For example, by comparing Cl1 to Cl2 although differences exist both are cattle based farm systems with comparable headage (although Cl2 has a slightly lower headage) and similar shrub management practices. Their main difference resides in the existence of fertilization measures (Cl1 with a mix of chemical and organic fertilization and Cl2 where no fertilization is predominant). The results reveal that overall Cl2 has a lower *rate of effective ecosystem service provision* (-0.4%) when compared with Cl1. This illustrates the reduced ecosystem service provision efficiency particularly in autumn and early spring. Cl2 also presents a higher ratio between the *ecosystem service mitigated impact* and the *structural impact*, which indicates proportionally higher soil erosion and less ecosystem service provision efficiency.

Another example can be found when comparing Cl4 to Cl6. Both are sheep based farm systems with very different headage (Cl4 [median=0.288] and Cl6 [median=1.724]) but also have very different farm management practices (Cl4 with a predominance of no fertilization and soil disking while Cl6 has a predominance of a mix of chemical and organic fertilization and shrub cutting). This comparison shows that overall Cl4 has a less effective performance than Cl6 in terms of ecosystem service provision with a comparison *rate of effective ecosystem service provision* of -16.17%, although it presents a significantly lower headage. This can be explained by two different reasons.

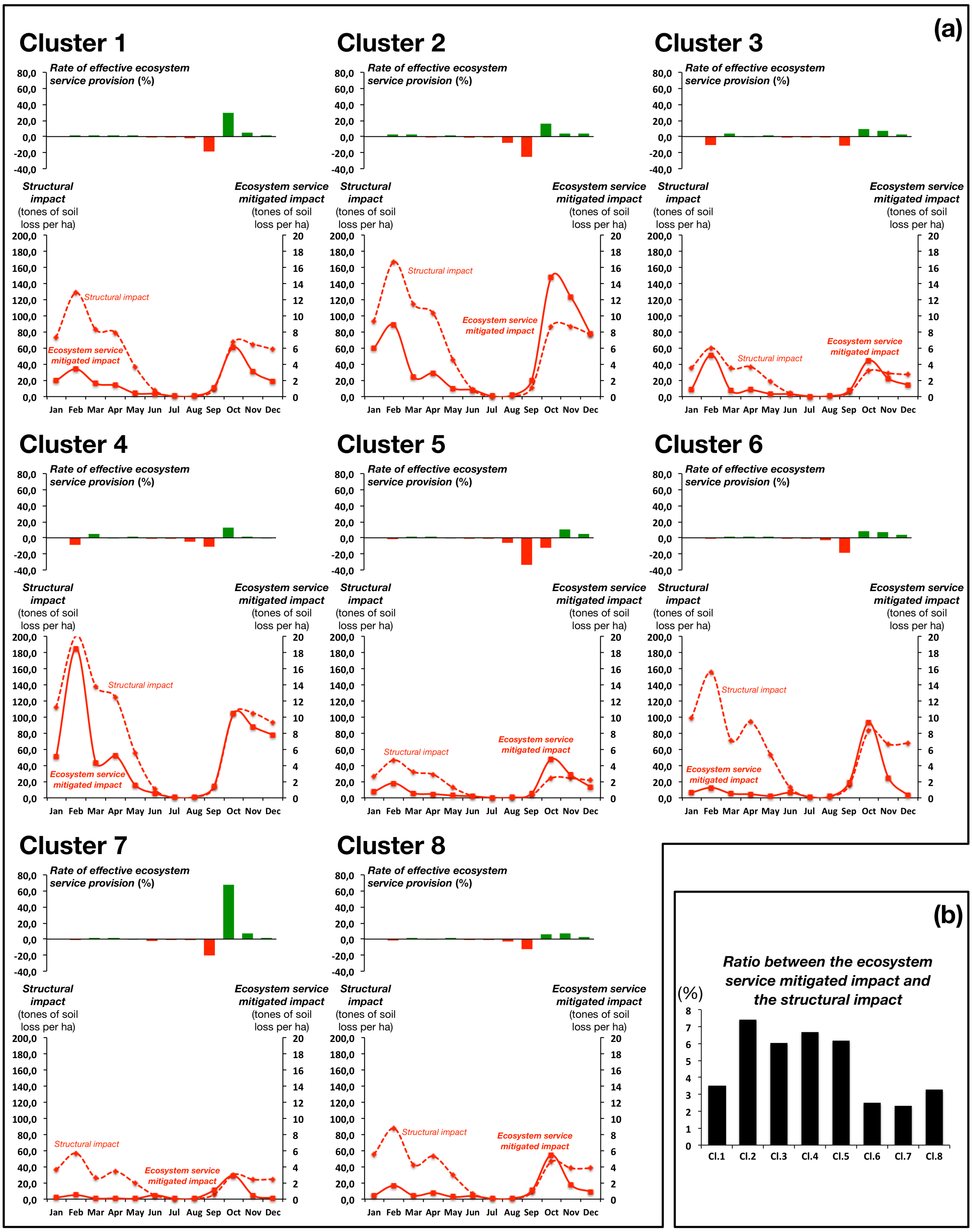

Figure 4.5 Ecosystem service provision indicators: (a) ecosystem service provision profiles; and (b) ratio between the ecosystem service mitigated impact and the structural impact calculated per cluster considering the yearly total.

The first is that the existence of a very small headage in Cl4 promotes the implementation of more intensive shrub management practices applied mostly in early Spring (i.e. soil disking), which in return increase the amount of ecosystem service *mitigated impact* in this period (Fig. 4.5a). In the case of Cl6 the higher headage reduces the need for more intensive shrub management practices, which reduces soil erosion especially in early spring. The differences in fertilization can also help to explain the divergence found between the ecosystem service provision efficiency. The combination of these different farm management practices resulted in a decrease of 62.6% in the ratio between the ecosystem service *mitigated impact* and the *structural impact* when comparing Cl4 with Cl6 (Fig. 4.5b).

4.4 Discussion

4.1. Data and methodological constrains

Even if the major components of the system are maintained, the examples used show that the *montado* is subject to different farm management practices, which result in very different ecosystem service provision conditions. The results are consistent and show the main line of relations between farm management and ecosystem service provision. Nevertheless, the reduced number of fine scale farm system data is a limitation that potentially limits the scope of the conclusions drawn from the present study. This lack of detailed data showed to be particularly important when assessing the relation between the farm management types and the provision of SEP. For example, soil disking is identified as the main shrub management practice in most farm units analysed, but we have not been able to get exact data on how frequently it is applied and which is the proportion of the *montado* farm land subject to this practice.

At the same time, the absence of higher resolution datasets with similar temporal and spatial coverage (e.g. NDVI data) also limits the level of detail of the conclusions. In the future there is an opportunity to overcome this limitation with the use of the outputs from the recently launched Sentinel-2A satellite (Johansen et al. 2008; Drusch et al. 2012; Pettorelli et al. 2014). Also the use of systematic survey data could overcome the limited data regarding the farm system dynamics. For example, in Europe, a solution could be a slightly more detailed farm survey within the existing European Common Agricultural Policy control system for support payments, and by making the resulting data systematically available.

From a methodological point of view, the number of and the specificity of the FMU also does not allow for a generalization of the results to other regions and/or management systems. Despite this, the consistent results that were obtained allow us to at least signal the relation between specific management practices and the provision of SEP, as illustrated in section 3.2. Also, the non-utilization of daily data, particularly regarding precipitation events, can underestimate the total amount of service provided. Although it falls out of the scope of this Chapter, the precise determination of the actual ecosystem service provision is a key step for valuation exercises and should be taken in consideration in future case studies addressing this issue.

4.4.2 Main findings

Despite these limitations the results illustrate the link between farm management and ecosystem service provision and exemplify the potential use of a combined set of ecosystem service provision indicators to monitor variations in farm management systems and their impact on the system sustainability. First, although with a highly simplified description of the farm system, the results allowed the classification of different variants of farm management types, acknowledging the high management complexity of these systems (Pinto-Correia et al. 2011). This complexity has significant implications when evaluating ecosystem service provision making explicit the need for more detailed and updated information on management practices and on the ongoing ecological processes (Almeida et al. 2013).

The relation between the eight identified FMU clusters and the provision of SEP was not clear. But when the comparison was made at the management practices level (Fig. 4) this relation emerges more clearly. The reduced data pool available to describe the farm system, which in turn limited the characterization and discrimination of the farm management types, can justify these results.

SEP seems to be sensitive to different farm management practices that go beyond headage or beyond a simple characterization of the farm system. The combined use of several ecosystem service provision indicators to describe the same ecosystem service, allowed to identify a direct effect between the combination of management practices and the variation of SEP. In this context, the rate of effective ecosystem service provision proved to be a good indicator to compare between land management practices, obtaining relevant indications about the impacts related to specific management practices (i.e. shrub management and fertilization type). By combining shrub control practices with fertilization practices it is possible to identify the farm practices that result in a lower ecosystem service provision performance, as soil disking and no fertilization. This also allowed to explain some of the results obtained at the FMU level, particularly the unclear relation between ecosystem service provision and farm management in clusters that result from a mix of shrub control and/or fertilization practices.

Simultaneously, the ratio between the ecosystem service mitigated impact and the structural impact also revealed to be a suitable indicator to compare between FMU types. It allowed to discriminate between highly intensive FMU and low intensity FMU. This distinction needs to take in to consideration the reduced number of FMU that was included in this analysis. Thus, further research should be conducted to better support these conclusions and validate the approach. Overall the results highlight the role of farm management in the provision of SEP in Mediterranean conditions. They also underline the importance of understanding the effects of specific farm practices on the sustainability and future viability of the FMU.

4.4.3 Management support opportunities

The description of SEP ecosystem service profiles for each FMU type (Fig. 5a) allowed to identify critical periods where farm management practices could be improved to increase SEP and to reduce soil erosion. It also provided evidences that could be used to establish farm system improvement opportunities. As an example, the FMU present in Cl4 have the ability to increase their production outputs (e.g. by increasing the headage) without damaging the ecosystem service performance if farm management changes were to be implemented (e.g. increasing fertilization and adopting less aggressive shrub management practices).

Although the management practices present in each FMU were simplified due to the limitations of data availability, the results show promise in the use of SEP indicators to monitor these practices and their effects on the functioning of the system. This offers opportunities to support decision-making towards sustainability, with high accuracy potential. As shown here, the use of remote sensing datasets can also aid in streamlining this type of ecosystem service based monitoring systems (c.f. Tallis et al. 2012; Pettorelli et al. 2014).

4.4.4 Implications for policy and compensation payments

From a policy point of view, the results of this assessment are promising. There are many attempts to improve the sustainability of the *montado,* and accurate supporting information is highly needed (Aronson et al. 2009; Sales-baptista et al. 2015). Due to the complexity of this social-ecological system adequate indicators have been difficult to define and the precise impact of farm management practices has so far been hard to assess (Cooper et al. 2007; Lomba et al. 2014). It is also acknowledged today that not all *montado* can be classified as having high natural value due to the impact on the system of different management practices at the farm level (Plieninger and Bieling 2013; Almeida et al. 2013; Almeida et al. 2015). Also, the challenge of the precise identification of thresholds in such practices (e.g. livestock grazing intensity, grazing breeds, frequency and depth of soil mobilization, etc.) in relation to the system's sustainability and resilience has not been met yet (Bugalho et al. 2011; Moreno et al. 2014; Almeida et al. 2015).

The use of ecosystem service based monitoring systems could be a step forward in developing and implementing effective policy impact evaluations as they provide evidence about the outcomes in the field of specific policy measures (Fancy et al. 2009; Tallis et al. 2012). It can also make possible the identification of conflicting or competing policy measures that can hamper or even improve the provision of ES. With this, it can allow to support the design, implementation and monitoring of both market mechanisms and public policy schemes grounded on the provision of multiple ES.

5

AN ASSESSMENT OF SOIL EROSION PREVENTION BY VEGETATION IN MEDITERRANEAN EUROPE: CURRENT TRENDS OF ECOSYSTEM SERVICE PROVISION

Based on the manuscript:
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Abstract

The concept of ecosystem services has received increased attention in recent years, and is seen as a useful construct for the development of policy relevant indicators and communication for science, policy and practice. Soil erosion is one of the main environmental problems for European Mediterranean agro-forestry systems, making soil erosion prevention a key ecosystem service to monitor and assess. Here, we present a spatially and temporally explicit assessment of the provision of soil erosion prevention by vegetation in Mediterranean Europe between 2001 and 2013, including maps of vulnerable areas. We follow a recently described conceptual framework for the mapping and assessment of regulating ecosystem services to calculate eight process-based indicators, and an ecosystem service provision profile. Results show a relative increase in the effectiveness of provision of soil erosion prevention in Mediterranean Europe between 2001 and 2013. This increase is particularly noticeable between 2009 and 2013, but it does not represent a general trend across the whole Mediterranean region. Two regional examples describe contrasting trends and illustrate the need for regional assessments and policy targets. Our results demonstrate the strength of having a coherent and complementary set of indicators for regulating services to inform policy and land management decisions.

5.1 Introduction

Soil erosion is one of the main environmental problems in European Mediterranean agro-forestry systems (García-Ruiz 2010) and for the sustainability of important ecosystems (Arnaez et al. 2011; Almagro et al. 2013). Several legislative and scientific initiatives have focussed on this issue since the late 1950s and recently the Thematic Strategy for Soil Protection (TSSP) defined a coherent framework for the assessment of European soils (CEC 2006). It pointed out the concentration of soil related risks in Southern Europe and the absence of a standardized approach to obtain policy relevant indicators (Gobin et al., 2004; Panagos et al., 2014a; Van-camp et al., 2004).

The ecosystem service (ES) concept is an effective communication tool to bridge knowledge between science and policy (Maes et al. 2012; Viglizzo et al. 2012). In the case of soil erosion prevention (SEP), the TSSP recognizes the importance and knowledge gaps related to the contribution of specific ecosystems and ecosystem functions to the mitigation of soil erosion. The ecosystem service concept also supports guidelines for the development of policy relevant indicators for international monitoring systems (Tallis et al. 2012; Reyers et al. 2013) because ecosystem service indicators that are sensitive to changes in land use, calculated using standardised methods (e.g. Maes et al., 2015), provide critical sources of information for agro-forestry systems under pressure from policy, environmental or climatic drivers (Hill et al. 2008; Navarra and Tubiana 2013).

Several studies (e.g. Martinez and Balvanera, 2012) and international initiatives (e.g. the Common International Classification of Ecosystem Services (Haines-young and Potschin 2013)) are contributing to the development of a coherent indicator set for the mapping and assessment of ES. Under Action 5 of the European Union (EU) Biodiversity Strategy to 2020 (EC 2011) the Working Group on Mapping and Assessment of Ecosystems and their Services (MAES) was set up to develop an assessment approach to be implemented by the EU and its Member States (Maes et al., 2014; 2013b). Supported by a growing scientific literature (Seppelt et al. 2011; Costanza and Kubiszewski 2012), this working group identified the need for more consistent methodological approaches to quantify and map ecosystem services and underlined the importance of finding indicators of ecosystem service provision (Müller and Burkhard 2012) that are sensitive to measure policy impacts (Maes et al. 2012; Dunbar et al. 2013).

Vegetation regulates soil erosion and thereby provides a major contribution to Mediterranean agro-forestry system's sustainability (Olesen et al. 2011; Iglesias et al. 2011). However, the regulation of soil erosion is projected to decrease in the coming decades in the region due to overgrazing, forest fires, land abandonment, climate change, urbanization or the combination of these drivers, (Shakesby 2011; López-Vicente et al. 2013a). And the intensity of these drivers has increased in the last decade (Llasat et al. 2010; Otero et al. 2011; García-Ruiz and Lana-Renault 2011; Hoerling et al. 2012; Bangash et al. 2013). Vegetation acts as an ecosystem service provider by preventing soil erosion and therefore mitigating the impact that results from the combination of the erosive power of precipitation and the biophysical conditions of a given area. Consequently, to better represent the impacts related to these drivers it is necessary to map not only the capacity for ecosystem service provision (e.g. according to land cover type) but also the actual ecosystem service provision and the remaining soil erosion (Nelson et al. 2009).

This Chapter presents a spatially and temporally explicit assessment of the provision of SEP by vegetation in Mediterranean Europe between 2001 and 2013. It provides insights on past and current trends of ecosystem service provision and enables the mapping of vulnerable areas. Finally, it demonstrates the strength of having a coherent and complementary set of ecosystem service indicators to inform policy and land management decisions.

5.2 Methods

5.2.1 Study area

The Mediterranean Environmental Zones (Metzger et al., 2005) were used to define the geographic extent of the study, which was constrained to continental Europe and a few larger islands due to data availability. The study area corresponds to 1.06 Million km^2 and covers all European Mediterranean countries (Fig. 5.1). It encompasses three major environmental zones, i.e. Mediterranean Mountains, which experience more precipitation than elsewhere in the Mediterranean, Mediterranean North and Mediterranean South, both characterized by warm and dry summers and precipitation concentrated in the winter months (Metzger et al., 2008a; Metzger et al., 2008b). Within the region agriculture is generally constrained by water availability and poor soils, and grasslands, vineyards and orchards are important land cover/use features (Panagos et al. 2013; Almeida et al. 2013).

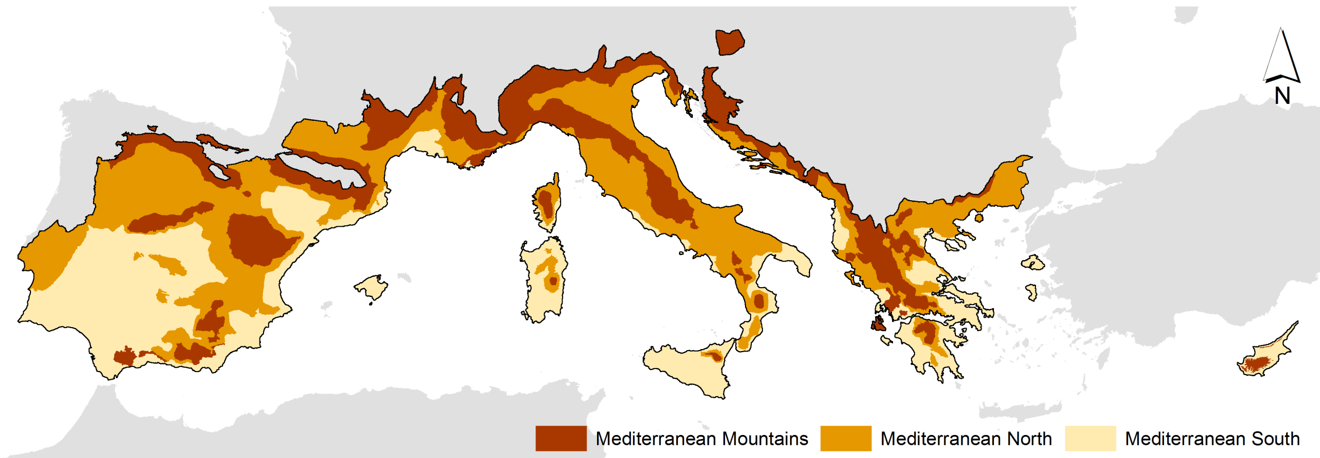


Figure 5.1 Geographic scope of the study area according to the European Environmental Stratification (Metzger et al. 2005).

5.2.2 Conceptual background

The conceptual approach for mapping and assessment of regulating services used in this Chapter has recently been described by Guerra et al. (2014), and is summarised in Fig. 5.2. SEP is provided at the interface between the structural components of the agro-forestry system and its land use/cover dynamics, which help mitigate the potential impacts from soil erosion (Guerra et al., 2014; Guerra et al., 2015). This approach combines a strong conceptual framework with the "avoided change" principle, characterizing regulating ecosystem service provision as the degradation that does not happen due to the contribution of the regulating ecosystem service provider (i.e. the vegetation cover) (Layke et al. 2012).

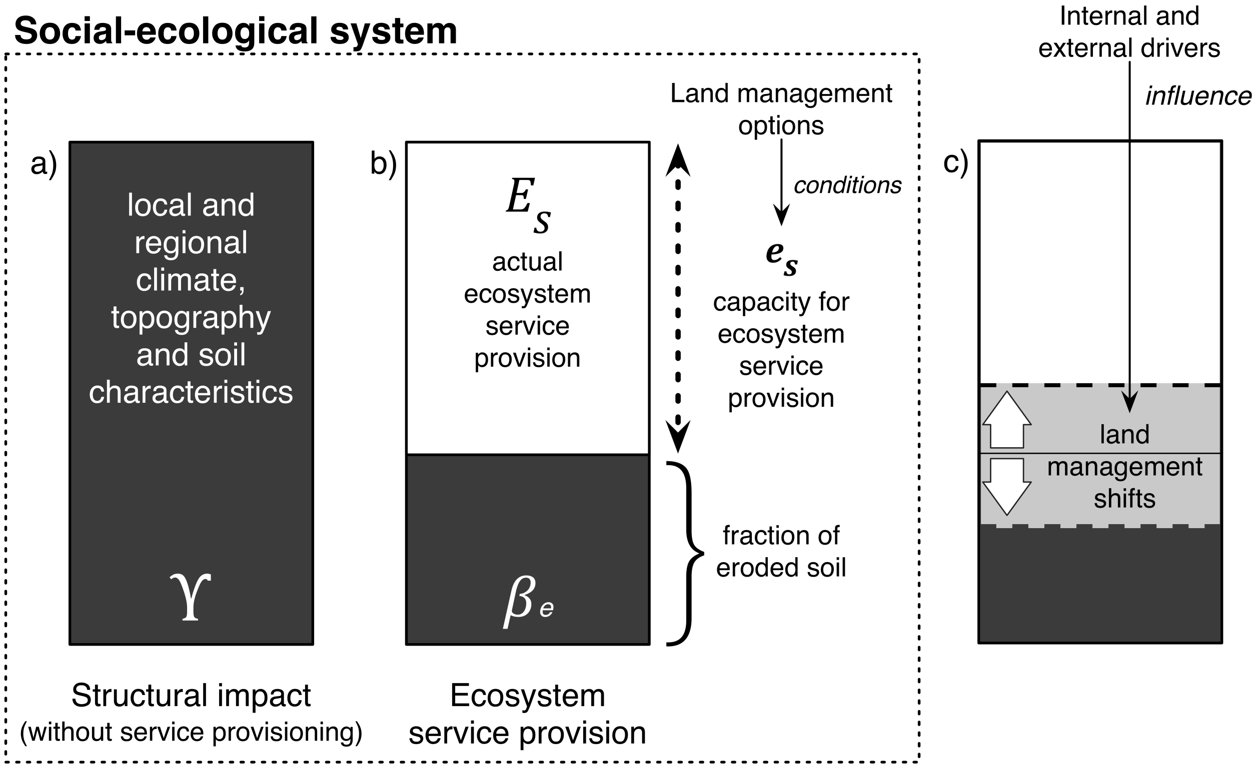


Figure 5.2 Conceptual framework for assessing the provision of regulating services (adapted from Guerra et al., 2015), where (a) presents the *structural impact* (ϒ) (i.e. the total soil erosion impact in the absence of soil erosion prevention); (b) distinguishes the *actual ecosystem service provision* (*Es*) as an avoided portion of the *structural impact* (measured in tons of soil not eroded) and determined by the *capacity for ecosystem service provision* (e_s) (i.e. the fraction of the *structural impact* that is mitigated by the ecosystem service, corresponding to an adimensional gradient ranging from 0 to 1), and the remaining *ecosystem service mitigated impact* (β_e) (i.e. the remaining soil erosion that is not regulated by soil erosion prevention); and (c) considers the variations in the *actual ecosystem service provision* resulting from changes in land management that occur at the local level although influenced by internal and external drivers.

To assess SEP following this framework it is necessary to first identify the *structural impact* (ϒ) related to soil erosion, i.e. the erosion that would occur when vegetation is absent and therefore no ecosystem service is provided (Fig. 5.2a). It determines the potential soil erosion in a given place and time and is related to rainfall erosivity (i.e. the erosive potential of rainfall), soil erodibility (as a characteristic of the soil type) and local topography (Ribeiro et al. 2004; Panagos et al. 2011b). Although external drivers can have an effect on these variables, they are less prone to be changed directly by human action.

The *actual ecosystem service provision* (*Es*) is a fraction of the total potential soil erosion (i.e. *structural impact*: ϒ), and it is determined by the *capacity for ecosystem service provision* (e_s) in a given place and time. We can then define the latter as a key component to quantify the fraction of the *structural impact* that is mitigated (Fig. 5.2b) and to determine the remaining soil erosion (i.e. the ecosystem service *mitigated impact* (β_e)). This *capacity for* ecosystem service *provision* is influenced by both internal drivers (including land management options, forest fires, and urban sprawl) and external drivers (including agricultural policy measures, spatial planning, and climate change). A detailed description of the methodological and conceptual frameworks is given in Guerra et al. (2014).

5.2.3 Indicators of ecosystem service provision

To understand the relation between drivers and the provision of ES, it is essential to translate the dynamics of the agro-forestry systems into a set of process related indicators that express system responses (Müller and Burkhard, 2012; Guerra et al., 2015). We propose a set of eight indicators that describe the different processes that contribute to SEP (Table 1), including indicators describing the state and dynamics of the *structural impact* (Υ), the ecosystem service *mitigated impact* (β_e), the *actual ecosystem service provision* (*Es*) and the *capacity for ecosystem service provision* (e_s).

Table 5.1 List of calculated indicators to describe the state and dynamics of ecosystem service provision (all indicators are computed at a 5 km grid resolution).

Indicator	**Description**	**Units**
Structural impact	Total soil erosion impact when no ecosystem service is provided	tons of soil per pixel area
Ecosystem service mitigated impact	Total of the remaining soil erosion after the ecosystem service provision	tons of soil per pixel area
Actual ecosystem service provision	Total of the actual ecosystem service provision corresponds to the total amount of ecosystem service provided, measured in ecosystem service providing units (tons of soil not eroded). It varies from season to season and year-to-year depending on the variation of the structural impact	tons of soil per pixel area
Ecosystem service provision capacity	Average fraction of the structural impact that is mitigated by the ecosystem service, it corresponds to an adimensional gradient from 0 to 1	-
Variation in structural impact	% variation in the total amount of structural impact considering the previous reference date	%
Rate of effective ecosystem service provision	% variation in the total amount of actual ecosystem service provision corrected by the structural impact fluctuations for a given time slice using the following expression: $100\times\left(\left[\frac{Es_{t+1}}{Es_t}-1\right]-\left[\frac{\Upsilon_{t+1}}{\Upsilon_t}-1\right]\right)$, where *Es* is the total actual ecosystem service provision, Υ is the total structural impact, and *t* corresponds to the temporal frame	%
Variation in ecosystem service provision capacity	% variation in the total amount of ecosystem service provision capacity considering the previous reference date	%
Variation in ecosystem mitigated impact	% variation in the total amount of ecosystem service mitigated impact considering the previous reference date	%

Together, these eight indicators are sensitive to changes in the climatic profile of each region, soil types, topography, management options and environmental drivers. Although all indicators have been produced at a 250 meters resolution, these were finally aggregated by summation to a 5 km grid (25km^2) resolution to better communicate changes and trends in ecosystem service provision and to avoid false precision related with the different data quality of the input datasets. In the case of the *capacity for ecosystem service provision* the average was used as, considering the adimentional character of this indicator, the sum does not provide any relevant interpretation value.

5.2.4. Datasets and methodological application

The Universal Soil Loss Equation (USLE; (Wischmeier and Smith 1978), a commonly used empirical model for the determination of potential soil losses (Fistikoglu and Harmancioglu 2002; Amore et al. 2004), was used to calculate soil erosion prevention between 2001 and 2013. Soil erosion is represented by a set of critical factors given by (Panagos et al. 2011b):

$$A = R \times LS \times K \times C \times P \tag{Eq.5.1}$$

where, *A* represents the amount of soil loss, *R* the rainfall erosivity, *LS* the topographic factor, *K* the soil erodibility, *C* the vegetation cover factor and *P* the conservation practices factor.

For the ecosystem service assessment, the *structural impact* (Υ) was calculated using the expression $\Upsilon = R \times LS \times K$ (Prasuhn et al. 2013), and the gradient of ecosystem service *mitigated impact* was determined by $\beta_e = \Upsilon \times \alpha$ (where $\alpha = C$ and $e_s = 1 - \alpha$). Technical infrastructure that could reduce impacts locally was not considered given the spatial scale of the study. Following these two expressions the *actual ecosystem service provision* (E_s) can be calculated by $E_s = \Upsilon - \beta_e$. Although no absolute measure of soil erosion is obtained, this mathematical formulation will generate a spatially explicit gradient of the potential soil loss and the related gradient of ecosystem service provided by vegetation cover (Guerra et al., 2014). Artificial surfaces were excluded from the evaluation and all parameters (after estimation) were directly resampled to a 250 meters resolution using an average filter.

5.2.4.1. Rainfall erosivity

The rainfall erosivity was estimated using the MedREM model proposed by Diodato and Bellocchi (2010) for Mediterranean conditions for the years of 2001, 2005, 2009 and 2013. This model was originally calibrated and validated using 66 meteorological stations distributed throughout the Mediterranean basin with multi-year data of rainfall erosivity (Diodato and Bellocchi 2010). It considers the variability in rainfall distribution and intensity and also the longitudinal differences within the Mediterranean basin. Rainfall erosivity was calculated between the months of August and November, corresponding to the most critical period for soil erosion in Mediterranean conditions (Luis et al. 2010). Daily rainfall observations, available through the European Climate Assessment and Dataset (ECA&D; http://eca.knmi.nl/; Haylock et al., 2008), were divided into four partially overlapping temporal time slices ([1991-2001]; [1995-2005]; [1999-2009]; [2003-2013]). For each time slice, the rainfall erosivity factor was calculated using the following expression (adapted from Diodato and Bellocchi 2010):

$$R_m = b_0 \times P_m \times \sqrt{d_m} \times \left(a + b_1 \times L\right) \tag{Eq.5.2}$$

where, R_m (MJ.mm.ha^{-1}.h^{-1}.month^{-1}) corresponds to the monthly erosivity factor for the month *m*, b_0 (MJ.ha^{-1}.h^{-1}) is a constant equal to 0.117, b_1 (d$^{0.5}$.mm$^{-0.50-1}$) is a constant equal to 2, *a* (d$^{0.5}$.mm$^{-0.50}$) is a constant equal to -0.015, *L* (^{o}) corresponds to the site longitude, P_m (mm) to the total amount of precipitation in a given month *m*, and d_m (mm.d-1) to the monthly maximum daily precipitation for month *m* averaged over a multi-year period (in this case a 10 years period was selected).

5.2.4.2. Soil erodibility

For the soil erodibility parameter we used the high resolution (500 meters resolution) European soil erodibility map (Panagos et al. 2014b) calculated from data collected in the Land Use/Cover Area frame Survey (LUCAS) soil survey for 2009. This was calculated based on the equation proposed by Wischmeier and Smith (1978) and Renard et al. (1997) (Panagos et al. 2014b):

$$K = \frac{2.1\times10^{-4} M^{1.14}\left(12-a\right)+3.25\left(s-2\right)+2.5\left(p-3\right)}{100}\times 0.1317 \qquad \text{(Eq.5.3)}$$

where, K corresponds to the soil erodibility factor, a is the percentage of organic matter, b the soil structure parameter, c the profile permeability class, and $M=\left(\%_{\text{silt}}+\%_{\text{very fine sand}}\right)\times\left(100-\%_{\text{sand}}\right)$.

5.2.4.3. Topography

For the topographic factor the SRTM shuttle digital elevation model was used following the expression proposed by Moore and Burch (1986):

$$LS=\left(\frac{a\times p}{22.13}\right)^{0.4}\times\left(\frac{\sin\left(d\right)}{0.0896}\right)^{1.3} \qquad \text{(Eq.5.4)}$$

where, LS represents the topographic factor, a refers to the flow accumulation model obtained from the topographic dataset, p to the pixel size, and d to the slope model in degrees.

5.2.4.4. Vegetation cover

The vegetation cover was estimated for each time slice using the relation between the Normalized Difference Vegetation Index (NDVI; calculated from MODIS 16 days NDVI composites with a 250 meters pixel resolution) and the USLE C Factor proposed by Van der Knijff et al. (1999, 2000) (Prasannakumar et al., 2012):

$$C=\exp\left[-a\times\frac{NDVI}{\left(b-NDVI\right)}\right] \qquad \text{(Eq.5.5)}$$

where, a=2 and b=*1*.

5.2.4.5. Integrated analysis and vulnerability assessment

The spatial distribution and temporal trends of the indicators (see Table 5.1) were analysed and mapped, and an overall ecosystem service provision profile was calculated for the entire study area. This was done using spatial statistics to obtain a total sum value (or an average value in the case of the *capacity for ecosystem service provision*) for the entire study area, and made it possible to isolate vulnerability areas and to pinpoint the periods with higher impact on SEP.

The vulnerability areas were identified by superimposing the variation of the *capacity for ecosystem service provision* (positive or negative), with the variation of the ecosystem service *mitigated impact* (positive or negative), both calculated between 2001 and 2013. A breakdown of the total land surface area covered by different combinations of these two variables reveals four groups related to each of the four quadrants (Fig. 5.5). The first group (1Q) represent areas that, despite their increase in the *capacity for ecosystem service provision*, reveal an increase of ecosystem service *mitigated impact*, i.e. despite the increase of vegetation capacity to halt soil erosion, there was an increase in the remaining soil erosion after the ecosystem service provision. The second (2Q) consisted of areas with a decrease of the *capacity for ecosystem service provision* and an increase of the ecosystem service *mitigated impact*, i.e. this group reflects the expected trend that a decrease in vegetation capacity to halt soil erosion resulted in more soil erosion. In the third group (3Q) are combined areas with a decrease of both the *capacity for ecosystem service provision* and the ecosystem service *mitigated impact*, i.e. reflecting a reduction in the efficiency of the ecosystem service to halt soil erosion, and finally the fourth group (4Q) included areas with an increase of capacity related to a decrease of the ecosystem service *mitigated impact*. This assessment thus identifies three types of vulnerable areas that require policy action (i.e. 1Q, 2Q, and 3Q).

Following this analysis, two smaller case-studies with contrasting regional ecosystem service provision profiles are described. Their specific ecosystem service provision profiles were constructed based on the description of the main indicators following the same methodological approach as for the overall ecosystem service provision profile described for the entire study area.

5.3. Results

5.3.1. States and trends of the Structural Impact

The *structural impact* (Υ) followed the rainfall dynamics during the same period: decreasing between 2001 and 2009 but increasing towards 2013. Overall, a decrease of 7.86% was observed between 2001 and 2013. Using 2013 as a reference year, the distribution of the *structural impact* (Fig. 5.3) showed relatively high values in the North of Italy, South of France, the East coast of the Adriatic Sea and the Western and Southern areas of the Iberian Peninsula. This spatial distribution remained throughout the period of the analysis with the exception of 2009, when the distribution was less pronounced. Between 2001 and 2013 the areas that experienced an increasing *structural impact* over the four months in analysis were located in the south of Italy and in the south of the Iberian Peninsula (Fig. 5.3). The results also showed that this increase is mainly related to an increase and higher variability of the *structural impact* (related to an increase in precipitation) in October following a dip in September.

5.3.2. States and trends of the ecosystem service mitigated impact

The ecosystem service *mitigated impact* (β_e) presented a different trend from the *structural impact* with an increase between 2001 and 2005 followed by a relatively constant decrease in its values until 2013. For 2013 (Fig. 5.3) it showed a concentration of high values mainly in the Southeast of the Iberian Peninsula, and in particular areas of the North of Italy and South of France. Together with some areas in the East of the Iberian Peninsula, South of Italy, and East of Greece, these areas also corresponded to the regions where this indicator has increased between 2001 and 2013. This trend implies a degradation of the conditions present in a given place as the total amount of soil loss (after the provision of SEP) increased. Despite of these degradation areas, the overall result for the entire region showed a decrease of 15.09% of ecosystem service *mitigated impact* between 2001 and 2013. This decrease was mainly located in Greece and in large portions of Italy, Spain and Portugal.

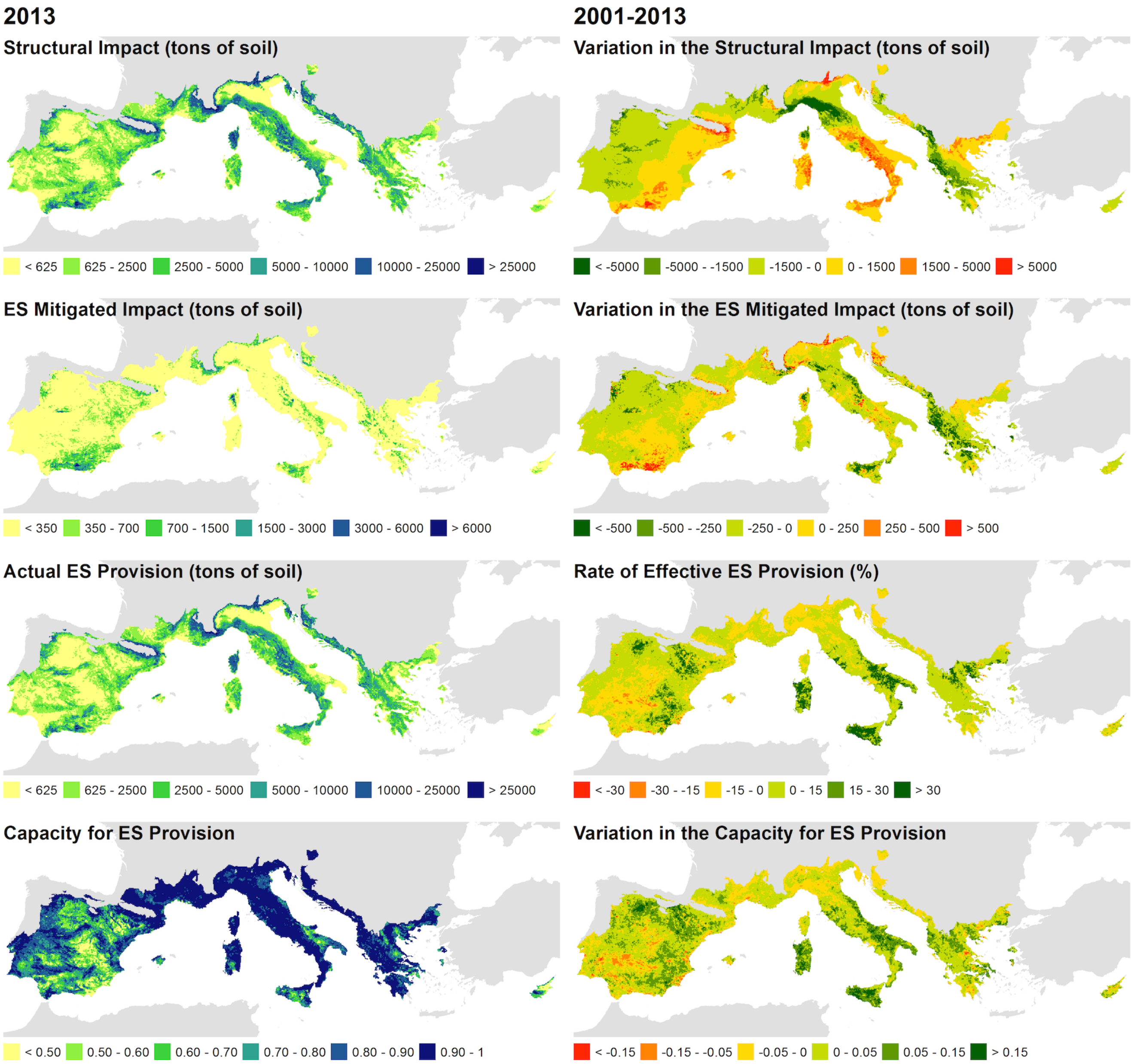


Figure 5.3 Spatial distribution of the different indicators calculated to illustrate the spatial and temporal distribution of soil erosion prevention in Mediterranean Europe (all indicators were computed based in a 5km grid).

5.3.3. States and trends of ecosystem service provision

As expected, the *actual ecosystem service provision* (*Es*) showed the same spatial and temporal pattern as the *structural impact* (Fig. 5.3). By contrast, the *capacity for ecosystem service provision* (e_s) revealed two very different patterns. The first pattern included the Iberian Peninsula and some areas in Southern Italy and in Eastern Greece, which were characterised by lower values and a more differentiated distribution of this indicator. The spatial location of these low values was similar to the spatial distribution of high values of *structural impact*, particularly in the South of the Iberian Peninsula and in the South coast of Italy (see Appendix 2). The second pattern concerned areas that showed more homogeneous distribution of higher values of the *capacity for ecosystem service provision*. Examples of these areas are the South of France, the East coast of the Adriatic Sea and the North of Italy. Despite this variable distribution, considering the entire region the overall values of the *capacity for ecosystem service provision* increased slightly between 2001 and 2013, from 0.815 to 0.844 (Fig. 5.4).

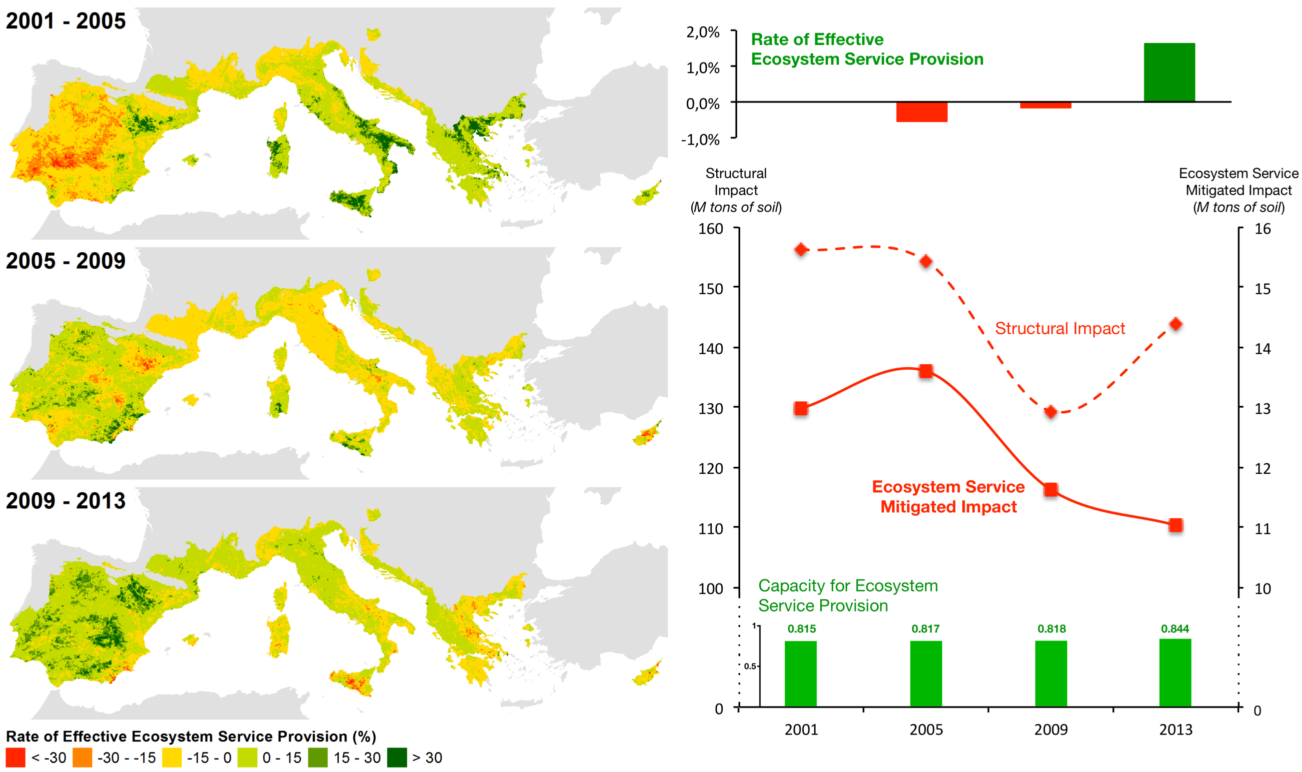


Figure 5.4 Spatial distribution of the *rate of effective ecosystem service provision* for the different periods considered (on the left) and the overall ecosystem service provision profile representing the different soil erosion prevention indicators aggregated for the entire study area (on the right).

This increase originated mainly from the South and East coast of Italy and from large areas in the North of Iberian Peninsula, while in the South of the Iberian Peninsula the *capacity for ecosystem service provision* decreased between 2001 and 2013. This overall increase is the result of a constant positive trend between 2001 and 2013 that is more substantial between 2009 and 2013 (Fig. 5.4). Regarding these areas in the South of the Iberian Peninsula, and using the monthly variation of the *capacity for ecosystem service provision*, we infer that this decreasing trend was related mainly to a decrease of provisioning capacity in October, particularly between 2001 and 2005. These spatial and temporal decrease patterns of the *capacity for ecosystem service provision* were in line with the increase of *structural impact* in the region.

A more detailed analysis of the *rate of effective ecosystem service provision* (Fig. 5.4) showed substantial dissimilarities between the different regions that were even more pronounced over the entire period (2001-2013) (see Fig. 5.5 for an example). While in the first period (2001-2005) the Iberian Peninsula showed substantial losses (*rate of effective ecosystem service provision* equal to -5.21%), in the following periods these losses were located more towards the North of Italy and the South of France (2005-2009) and to the South of Italy and Greece (2009-2013). Overall, although not statistically significant (p= 0.05), the entire study region presented a positive trend in terms of the effectiveness of service provision (0.66%), particularly in the period between 2009 and 2013 where the *rate of effective ecosystem service provision* increased by 1.62% (Fig. 4).

The vulnerability analysis revealed that 43.5% of the total area is related to one of the three groups of vulnerable areas (i.e. 1Q, 2Q and 3Q) (Fig. 5.5a). The second (2Q corresponding to 16.5% of the total area) and the fourth group (4Q corresponding to 56.5% of the total area) demonstrated the expected inverse relation between the *capacity for ecosystem service provision* and the ecosystem service *mitigated impact*. Put differently, the increased capacity to prevent soil erosion is generally positively correlated to a decrease in soil erosion. In contrast, the other two groups included areas where despite an increase of capacity there is still an increase of impact (1Q corresponding to 18.8% of the total area), as well as areas where a decrease of capacity is followed by a decrease of the ecosystem service *mitigated impact* (3Q corresponding to 8.2% of the total area).

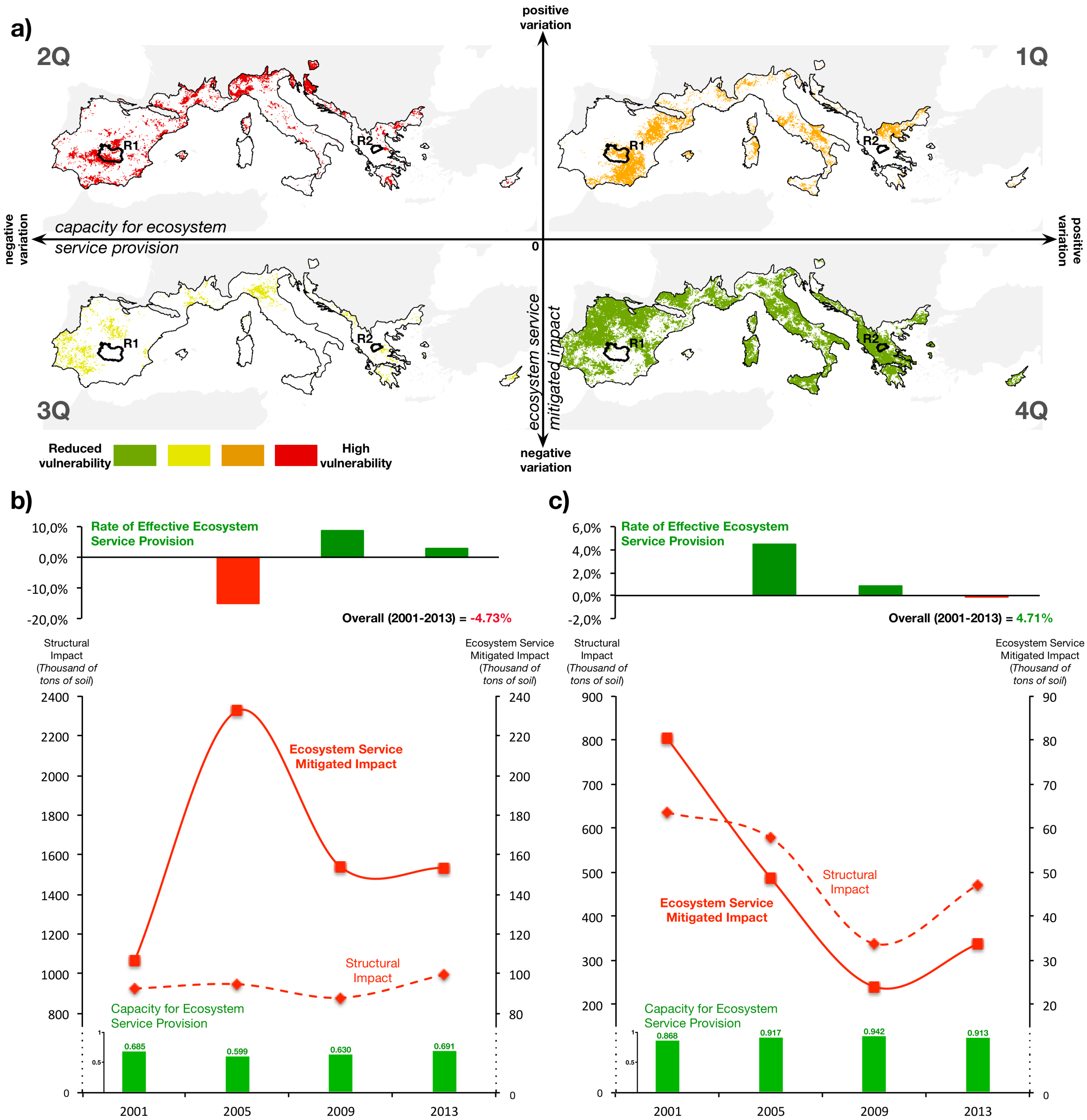


Figure 5.5 Representation of: a) the spatial distribution of the grid cells discriminated by the variation (2001-2013) of the *capacity for ecosystem service provision* (horizontal axis) and the variation (2001-2013) to the *ecosystem service mitigated impact* (vertical axis); b) the regional ecosystem service provision profile for R1, corresponding to the NUTS 3 Cuidad Real (Spain); and c) the regional ecosystem service provision profile for R2, corresponding to the NUTS 3 Trikala (Greece).

Therefore, these two different indicators (i.e. the *capacity for ecosystem service provision* and the *ecosystem service mitigated impact*) must be considered when interpreting trends of SEP provision in order to formulate effective mitigation measures. Combined, these two indicators give a clear picture of the underlying questions that rise in each area. Figure 5.5 suggests that in 64.7% of the total area the ecosystem service *mitigated impact* decreased, mainly due to an increase of the *capacity for ecosystem service provision*. In contrast, from the 35.3% of areas with an increase of the ecosystem service *mitigated impact*, 53.3% also showed an increase of the *capacity for ecosystem service provision*.

The two selected case-study regions (Fig. 5.5b and 5.5c) illustrate two very different trends. R1, the NUTS 3 Cuidad Real in Spain (Fig. 5.5b), presents an overall (2001-2013) negative trend of the *rate of effective ecosystem service provision* (-4.73%). This happens despite the slight increase in the *capacity for ecosystem service provision* (0.84%) in the same period and is related to the substantial increase (118.14%) in the ecosystem service *mitigated impact* in the first period (2001-2005), which resulted from a decrease of 15.08% in the *rate of effective ecosystem service provision* for the same period.

Despite the recent (2005-2013) improvements in the *rate of effective ecosystem service provision*, the regional SEP dynamics resulted in an increase of 43.98% of the ecosystem service *mitigated impact* between 2001 and 2013. In contrast, R2, the NUTS 3 Trikala in Greece (Fig. 5.5c), presents an overall (2001-2013) negative trend of the *rate of effective ecosystem service provision* (4.71%) accompanied by a decrease of 58.04% of the ecosystem service *mitigated impact* in the same period. Although this region presents a positive development in terms of SEP provision, the general trend of the *rate of effective ecosystem service provision* (2001-2013) shows a systematic decrease in the period of analysis, despite the increase of 5.19% in the *capacity for ecosystem service provision*.

5.4. Discussion

5.4.1. Methodological potential and limitations

The analysis of the spatial and temporal distribution of SEP used a diverse set of process indicators that encompass the impacts related to the dynamics of soil erosion and to the service provision generated by vegetation. Compared to other methodological approaches that usually base their assessments on a single indicator (e.g. Koschke et al., 2012; Helfenstein and Kienast, 2014; Frélichová et al., 2014), our approach provides more insight and it more easily identifies the relations between the underlying landscape processes and their consequences in terms of service provision and of the remaining impacts. Also, although the *actual ecosystem service provision* can be used as an indicator for valuation purposes, it is not a good "stand alone" indicator for trend analysis as it is dependent on the spatial distribution, magnitude and temporal trend of the *structural impact* (Guerra el al., 2015).

Our results show that the *rate of effective ecosystem service provision* can be a more insightful indicator as it provides a better grasp of the local/regional ecosystem service provision performance. This indicator corresponds to the percentual variation of the early time slice (e.g. 2001) in comparison to the following (e.g. 2005). This means that if a particular area has lost a considerable amount of ecosystem service provision in a given period, it is probable that in the next period it registers a gain (e.g. the recovery from a previous forest fire). Although this does not mean that the net provision of ecosystem service was positive considering the entire period (2001-2013). This was illustrated in the South of the Iberian Peninsula where in the first period (2001-2005) there was a substantial loss of the *rate of effective ecosystem service provision* accompanied by relative gains in the following periods, although, in the same area, there was a cumulative increase of the ecosystem service *mitigated impact*. In this case this dynamic can also be explained by the high variation in the *capacity for ecosystem service provision* registered in the region (Appendix 3).

SEP alone cannot be used to determine the effectiveness of ecosystem service provision in a given region (Fitter et al. 2010; Dunbar et al. 2013). It is also important to consider the interactions and eventual trade-offs between services in more strategic assessment of the net ecosystem service provision in a given region to better define local environmental targets.

5.4.2. SEP provision and vulnerability assessment

Our results illustrate the value of having a comprehensive and complementary group of process-based ecosystem service indicators. They show an overall, non-significant, increase in SEP in the region. A worrying trend becomes apparent when assessing areas that showed a decrease of the *capacity for ecosystem service provision* and an increase of the ecosystem service *mitigated impact* (Fig. 5.5 2Q). These areas (corresponding to 16.5% of the total study area) point to the eventual insufficiency, ineffectiveness or non-existence of soil protection measures and reflect very important regional differences. While in Italy, the Northeast coast of the Baltic Sea and the South of France this dynamic is related to a predominance of forest areas, in the Iberian Peninsula and in Greece it is related to a predominance of agricultural areas. This vulnerability analysis also shows that, between 2001 and 2013, 25% of areas with an increase of the *capacity for ecosystem service provision* were subject to a further increase of soil loss. These results are related to the 18.8% of areas with an increase of both the ecosystem service *mitigated impact* and the *capacity for ecosystem service provision* (Fig. 5.5 1Q), revealing a situation where the presence of protective vegetation cover did not result in an enhanced soil protection.

The two smaller case-studies (Fig. 5.5b and 5.5c) illustrate the power of creating a regional ecosystem service provision profile for assessing the efficiency of SEP. In R1 we observe that even with an overall increase of 0.88% in the *capacity for ecosystem service provision*, the region had an increase of 43.98% of the ecosystem service *mitigated impact* following a decrease of 4.73% in the *rate of effective ecosystem service provision*. Although there is an improvement in SEP provision in recent years (2005-2013), this exposes the insufficiency of current regional initiatives to halt soil erosion by promoting SEP. By contrast, R2 shows a completely different pattern with constant gains of efficiency, even when (between 2009 and 2013) there is a decrease of 3.07% in the *capacity for ecosystem service provision* that is reflected in a slight decrease of 0.08% on the *rate of effective ecosystem service provision*. Both examples demonstrate the possibility to define regional targets that can steer regional conservation and economic development policies that aim at minimising these impacts and their effects on human wellbeing.

5.4.3. Policy and research implications

Declines in regulating services provision like SEP can result in declines in ecosystem resilience (Bennett et al. 2009), and may affect the provision of other ES. Our results show that, in total, 43.5% of the entire study area presented some type of vulnerability regarding the mitigation of soil erosion. If this information would be available in national and international monitoring systems, policy and management decisions could be better informed and action could be taken timely.

The insight provided by the combination of indicators suggests that current policies and land management fail to safeguard SEP to halt soil erosion. One possible explanation could be that most policies that land managers follow correspond to generic top-down sectorial approaches. The spatial patterns and indicator values found here indicate that further disaggregation, consideration of context, and place-based or regional targets could improve SEP in Mediterranean Europe and prevent undesired ecosystem service provision trajectories.

Finally, in future research, the relative positive trends found in this Chapter should be contextualized and regionally assessed in relation to regional social, ecological and economic. This means that further research should identify whether the observed positive trends correspond to an increase of management efficiency and/or policy implementation or if they are related to land abandonment processes that eventually resulted in an increase in the capacity for SEP.

6

GENERAL DISCUSSION AND CONCLUSIONS

The primary aim of this Thesis was to develop a spatially and temporally explicit social-ecological approach to quantify, map and describe the provision of soil erosion prevention in the Mediterranean. A novel conceptual and methodological framework was developed (Chapter 2) and applied in three case studies (Chapters 3, 4 and 5) to evaluate potential applications and limitations. The case studies focus on using ecosystem service indicators for local scale policy (Chapter 3) and land management (Chapter 4) assessment and support, and for broad scale implementation for policy support and regional comparison (Chapter 5). The following sections discuss the main findings, strengths and limitations of the framework, future research and applications and ends with a set of final considerations.

6.1 Main findings

Overview

Our conceptual and methodological framework was developed based on the assumption that for regulating services the ecosystem service provision emerges from the mitigation of a potential impact. This means that in the absence of the impact no ecosystem service is provided, despite the ecosystem service provider is present. This was illustrated by calculating the within year variation of ecosystem service provision where it is clear that the absence of impact (in this case associated to the absence of rain) results in the non-existence of ecosystem service provision. This relation results in a heterogeneous spatial and temporal representation of the actual ecosystem service provision and explicitly quantification and maps are provided for studies at different scales.

At the same time, results show that the capacity for ecosystem service provision also varies in time and space and it represents a central element in the relation between land management and ecosystem service provision. The intensity of land management determines the variations in the capacity for ecosystem service provision and, consequently, the mitigation of an eventual impact event, i.e. the actual ecosystem service provision. This relation is particularly relevant in complex managed systems, like the Mediterranean silvo-pastoral systems, where important challenges to ecosystem service assessments, mapping and monitoring are raised. These challenges are described in Chapter 1. In the case of Mediterranean silvo-pastoral systems, they are mainly related to: a) the fuzziness of the landscape and also of the land management activities; and b) the multiple agents that commonly manage the land, making it difficult to identify and determine specific management practices and their impacts. These challenges represent a drawback to the use of ecosystem services as policy or land management indicators but if internalised can enforce ecosystem services as key indicators to support policy evaluation and design as well as providing land management support. Understanding the process of ecosystem service provision as well as the difference between the capacity for ecosystem service provision and the actual ecosystem service provision is essential to maximise the integration of these indicators in policy and management support.

While the capacity for ecosystem service provision represents the link between the land manager and the ecosystem service provision, it does not provide a good representation of the actual ecosystem service provision. As described before, the temporal and spatial mismatch is well demonstrated by the findings and results from the fact that the actual ecosystem service provision is highly dependent on the presence of an impact to be mitigated. Chapter 2 provides a good example of this mismatch by superimposing the results from the capacity for ecosystem service provision with the values of actual ecosystem service provision. In the example it is clear the difference in the spatial and temporal distribution of the capacity for ecosystem service provision and of the actual ecosystem service provision. At the same time we demonstrate the influence of land management and the importance of matching, through management, the capacity for ecosystem service provision and the existence of an impact to be mitigated.

Overall, the proposed conceptual and methodological framework provides a complete indicator set that allows to **identify**, **describe** and **monitor ecosystem service provision** across scales, identifying management impacts and policy outcomes. This was demonstrated for three domains, which are discussed in more detail below: policy evaluation; land management evaluation; and a Mediterranean assessment. Overall, the results suggest that current policy and land management practices do not prioritise nor safeguard the provision of soil erosion prevention in European Mediterranean social-ecological systems. In this context the proposed ecosystem service indicator set can be used to redesign current and future policies/land management practices as well as, if included in broad scale monitoring systems, allowing the monitoring and evaluation of the impact of future developments and policy implementation.

Policy evaluation (Research Question 1)

In the Mediterranean silvo-pastoral systems of South Portugal agricultural policy has severely affected soil erosion prevention in the last 60 years, particularly since the productivist push of the early CAP. The long-term assessment of policy measures helped to understand the historical effects of public agricultural policies in the region, illustrating the negative consequences of recent measures in soil erosion prevention. In Chapter 3 we test the applicability of this framework to evaluate and monitor the impacts that result from the implementation of relevant policy measures. Although in the beginning of the second half of the 20th century there was an improvement in the provision of soil erosion prevention, after the start of the Portuguese European Community integration process in the late 1970s, our case study shows that soil erosion prevention has been systematically decreasing. This decrease trend has been intensified in the 1980s and in the 1990s with the productivist push of the early Common Agricultural Policy and it was only slightly halted with the revision of the policy in the 2000s and the introduction of the 2nd Pillar environmental support payments.

A new indicator was introduced and the results underline the potential of using the rate of effective ecosystem service provision rather than the actual ecosystem service provision to detect these dynamics. These results are in line with other studies for the same region (e.g. (Jones et al. 2011b; Pinto-Correia et al. 2013; Almeida et al. 2015) and they strongly emphasize the need for spatially informed agricultural policies adapted to the social-ecological context of each region.

It is highlighted that the evaluation of ecosystem service indicators can positively contribute to identify, assess and foresee unintended consequences of policy implementation. Both in Chapter 3 and in Chapter 5 our results give strong evidence that current sectorial policies do not contribute to improve soil erosion prevention. Our framework has shown to be sensitive to changes in policy and it can be used for strategic assessments, both in long-term monitoring schemes and/or ex-ante evaluations. This was reflected in the definition of an ecosystem service profile that combines all indicators to produce a clear picture of the spatial and temporal distribution of ecosystem service provision as well as allows to identify specific vulnerability types (an example is given in Chapter 5) that have to be tackled with different policy/management strategies. Using the same principle used to backtrack ecosystem service provision in the case study presented in Chapter 3, foresight analysis can be implemented by combining ecosystem service indicators with land cover change scenarios and climate change foresight products. With this combination, changes in ecosystem service provision can be detected and anticipated, and policy measures can be designed to minimise impacts and to optimise service provision where it counts the most.

Land management evaluation (Research Question 2)

Land management affects ecosystem services provision (positively and negatively) by changing the spatial and temporal distribution of the capacity for ecosystem service provision. This is a result of different land management intensities and more specifically of different land management practices. In order to be considered a management support indicator, an ecosystem service indicator has to be able to detect differences between land management types and/or land management practices.

Our results show that at the local/farm level, ecosystem service indicators can be used to monitor the impacts resulting from different land management practices. In the second case study presented (Chapter 4), we evaluated the possibility to detect the effect of management practices using the proposed ecosystem service framework. Land management is not homogeneous in farms with the same general traits (Almeida et al. 2015) and the expectation was that the indicators used would be able to clearly discriminate between broad land management types. Instead, the ecosystem service indicators revealed sensitivity to specific management practices or bundles. A specific example is given by comparing soil mobilization technics with fertilization practices. As expected, a combination of more intensive soil mobilization with lower fertilization inputs resulted in a lower effective ecosystem service provision. In such complex silvo-pastoral systems, being able to identify and even to quantify these relations is a major step forward in the understanding of these systems, as well as in establishing ecosystem services as indicators for management support. The results also give an indication that in general land managers do not prioritise the maximisation of soil erosion prevention when planning their land management activities.

Our results highlight the importance of more in-depth land management characterizations as ecosystem service indicators can provide valuable information for land management support. As an example, although in this region livestock is one of the main factors to reduce soil erosion prevention (Pinto-Correia et al. 2011; Almeida et al. 2013), the results stress the importance of attaining data related to other relevant farm management practices (e.g. type of soil fertilization). Together this information allows a better understanding of particular aspects of the land management that can potentially hamper the provision of soil erosion prevention and, consequently, fully grasp the meaning of the indicator outputs. In this context, the case study developed in Chapter 4 showed that the proposed indicators are able to discriminate between management practices and land management types. Particularly it was possible to observe that the rate of effective ecosystem provision can encompass the individual and the combined variation of specific land management practices.

This provides useful indicators that allow land managers to correct their management strategies and thus minimize unintended impacts. It also provides complementary information to the one produced in land management surveys. If these and other management-ecosystem services relations were attained, with the use of remote sensing and other of-site methods the potential to produce management relevant information for large areas is very high.

Mediterranean Europe assessment (Research Question 3)

Although not statistically significative, soil erosion prevention is slightly increasing in Mediterranean Europe particularly between 2009 and 2013. In Chapter 5 we implemented a spatially and temporally explicit assessment of the provision of soil erosion prevention in Mediterranean Europe in the last decade (2001-2013). We found that in general ecosystem service provision is increasing in Mediterranean Europe (0.66%), particularly between 2009 and 2013 (1.62%). These overall variations are not statistically significant but a closer analysis of the study area reveals areas with high increases in service provision (e.g. South of Italy between 2001-2005) and areas with high decreases (e.g. large extents of the Iberian Peninsula between 2001-2005). The outputs from this analysis were also used to illustrate how this ecosystem service framework and particularly the ecosystem service profile could be used to compare between regions. The two examples used show how, within the same environmental region, changes in management strategies and/or the impacts from other drivers have a profound effect on soil erosion prevention with severe consequences for the resulting impacts.

Despite the positive results found, 43.5% of Mediterranean Europe is vulnerable and in need of focused attention to identify causes and implement effective mitigation measures. These vulnerable areas are divided between areas with a decrease in the capacity for ecosystem service provision (with positive or negative consequences in terms of the variation of the ecosystem service mitigated impact) and areas where despite an increase in the capacity for ecosystem service provision there was also an increase of the ecosystem service mitigated impact. The later (corresponding to 43.2% of the vulnerable areas) shows that although the capacity for ecosystem service provision can be increased, if this is not done taking in to account the optimization of ecosystem service provision, the impacts can still increase.

Particularly for soil erosion prevention this provided a clear representation of the different dynamics associated to the provision of the service. Consequently, the results suggest that current policy and land management actions are not safeguarding the provision of soil erosion prevention. It also emphasises the need to evaluate and assess regulating ecosystem services considering a bundle of process based ecosystem service indicators rather than a single proxy indicator.

6.2 Strengths and limitations of the framework

In the first Chapter we laid out as a primary objective of this Thesis the development and test of an ecosystem service assessment framework that provides a complementary approach to address modern conservation challenges in land management and policy design. As a result of the different case studies we were able to implement a novel conceptual and methodological framework that can tackle this main objective. By doing so, we were able to: a) encompass the evaluation of the impacts of historic and current policy implementation in the provision of soil erosion prevention; b) provide an indicator system that can be used to monitor land management practices and assess their positive or negative influence in ecosystem service provision; and c) produce relevant results at macro and sub-regional scales.

The fact that this framework corresponds to a process-based approach allows to determine past and future trends of soil erosion prevention and relating them to land management strategies and policy implementation. Simultaneously, it allows to identify and typify vulnerable areas of ecosystem service provision, anticipating their vulnerability factors.

The **strengths** of the proposed framework can be summarized as:

- being a **process based approach** that considers the social-ecological system dynamics as the basis for ecosystem service identification, assessment, and monitoring;
- a **contribution to strategic policy evaluations**, allowing to assess the influence of past and current policies on ecosystem service provision;
- the possibility to **identify and inform land management thresholds**, allowing to support land management decisions and giving a direct contribution to the system sustainability;
- the **possibility to determine past trends** of service provision considering climatic as well as political and management dynamics;
- the **potential for foresight analysis** by combining this methodological framework with other modelling/scenario platforms (e.g. land cover, climate); and
- the capacity to **identify and typify vulnerable areas** of ecosystem service provision

At the same time, the proposed framework raises a set of relevant issues related to its implementation (e.g. not including peak precipitation can result in an underestimation of both the impacts and the ecosystem service provision) and future use. Regarding the later, a major issue can rise from the utilization of the outputs from the framework to justify the reduction of the capacity for ecosystem service provision in areas/moments where it is not relevant. When assessing regulating services, although this optimization can be made, it is imperative to understand the current and future implications of this decision before it is implemented. This also means that our approach needs data and modelling technics that can capture and quantify the impacts resulting from such decisions.

The **limitations** of the proposed framework can be summarized as:

- in the analysis **peak precipitation events were not considered** in the assessment of soil erosion prevention;
- the indicator set **should not be used as "stand alone indicators"** being necessary to use them in a comparative approach (between time slices or between regions);
- **potentially reducing the necessary spatial/temporal system equilibrium** by "optimizing" the capacity for ecosystem service provision in areas/moments with reduced impacts; and

- **being data and model dependent**, particularly relevant in trend analysis, upscale exercises and to expand the general model to other ecosystem regulating services.

6.3 Future research and applications

This Thesis discussed the implementation of a novel conceptual and methodological framework to assess and map the provision of soil erosion prevention. The **implementation** of this framework **to other regulating services** will increase its applicability and relevance, particularly in multi-service assessments. This would **allow to tackle** the **challenges of double counting and competing ecosystem services** by discriminating the process by which each service is provided and by singling out the different ecosystem service providers. Also, considering this type of assessment, it should be possible to **describe the temporal and spatial complementarity between ecosystem services**, especially regarding their temporal and spatial distribution. At the same time, it may be possible to improve both the conceptual and methodological framework by **testing its implementation for different environmental zones** and assessing the consequences and particular challenges of each region (e.g. regions where the snow cap play a significant role or areas with high vegetative growth).

For specific regions, **improving the understanding of the links between ecosystem service provision and land management** can be achieved by combining the ecosystem service evaluations with the outputs from in-depth land management assessments. As discussed in Chapter 3, this can be achieved by implementing targeted surveys or as a result of other participatory approaches. Resulting for this a key question emerges: to **understand why some land managers give priority to (or at least improve) the provision of ecosystem services and others don't**. Understanding this relation could have important implications for policy design and management support as both policies and management support guidelines could be designed specifically to increase ecosystem service provision.

Finally, **current environmental monitoring systems should be evaluated** to identify the connections between current indicators being monitored and the ones proposed, particularly in less studied regions. Alternatively these environmental monitoring systems could be **tested to assess their permeability to these types of ecosystem service indicators**. This assessment will allow to understand the level of applicability of the proposed framework to other regions with different data collection procedures and targets.

6.4 Conclusions

This study demonstrates that current methodologies for the quantification and mapping of the capacity to provide ecosystem services fail to grasp the actual ecosystem service provision. Ultimately these methodological approaches can provide erroneous conclusions about the state, distribution and trends of regulating ecosystem service provision at local and broad scales.

For specific social-ecological systems, the use of a coherent set of ecosystem service indicators provides a better description of the systems' dynamics and it allows the underlying causes of change in ecosystem service provision to be identified. Our research identifies the need for more adaptive policy design for European Mediterranean systems. Such policy must be able to cope with local trends of ecosystem service provision and with the definition of regional ecosystem service provision targets that allow to effectively mitigate relevant impacts (e.g. soil erosion).

At the local scale, the research also shows how ecosystem service indicators could be used to monitor farm systems and their effective provision of soil erosion prevention as well as supporting managers' decision-making. Altogether, the combination of the different aspects treated in this dissertation provides a comprehensive overview of the potential applications of the methodological framework that was developed to quantify and map soil erosion prevention.

REFERENCES

Abbott I, Le Maitre D (2009) Monitoring the impact of climate change on biodiversity: The challenge of megadiverse Mediterranean climate ecosystems. Austral Ecol 35:406–422. doi: 10.1111/j.1442-9993.2009.02053.x

Abson DJ, Termansen M (2011) Valuing ecosystem services in terms of ecological risks and returns. Conserv Biol 25:250–8. doi: 10.1111/j.1523-1739.2010.01623.x

Acácio V, Holmgren M (2012) Pathways for resilience in Mediterranean cork oak land use systems. Ann For Sci 71:5–13. doi: 10.1007/s13595-012-0197-0

Almagro M, Vente J, Boix-Fayos C, et al. (2013) Sustainable land management practices as providers of several ecosystem services under rainfed Mediterranean agroecosystems. Mitig Adapt Strateg Glob Chang. doi: 10.1007/s11027-013-9535-2

Almeida M, Azeda C, Guiomar N, Pinto-Correia T (2015) The effects of grazing management in montado fragmentation and heterogeneity. Agrofor Syst. doi: 10.1007/s10457-014-9778-2

Almeida M, Guerra C, Pinto-Correia T (2013) Unfolding relations between land cover and farm management: high nature value assessment in complex silvo-pastoral systems. Geogr Tidsskr J Geogr 113:1–12. doi: 10.1080/00167223.2013.848611

Alpert P, Ben-Gai T, Baharad A, et al. (2002) The paradoxical increase of Mediterranean extreme daily rainfall in spite of decrease in total values. Geophys Res Lett 29:29–32.

Amore E, Modica C, Nearing MA, Santoro VC (2004) Scale effect in USLE and WEPP application for soil erosion computation from three Sicilian basins. J Hydrol 293:100–114. doi: 10.1016/j.jhydrol.2004.01.018

Antrop M, Brandt J, Loupa-Ramos I, et al. (2013) How landscape ecology can promote the development of sustainable landscapes in Europe: the role of the European Association for Landscape Ecology (IALE-Europe) in the twenty-first century. Landsc Ecol 28:1641–1647. doi: 10.1007/s10980-013-9914-9

Arnaez J, Lasanta T, Errea MP, Ortigosa L (2011) Land abandonment, landscape evolution, and soil erosion in a Spanish Mediterranean mountain region: The case of Camero Viejo. L Degrad Dev, 537–550.

Aronson J, Santos Pereira J, Pausas J (2009) Cork Oak Woodlands on the Edge: Ecology, Adaptive management, and restoration. Island Press

Avillez F, Finan T, Josling T (1988) Trade, exchange rate and agricultural pricing policies in Portugal. The World Bank, Washington

Bagstad KJ, Johnson GW, Voigt B, Villa F (2013) Spatial dynamics of ecosystem service flows: A comprehensive approach to quantifying actual services. Ecosyst Serv 4:117–125. doi: 10.1016/j.ecoser.2012.07.012

Bangash RF, Passuello A, Sanchez-Canales M, et al. (2013) Ecosystem services in Mediterranean river basin: Climate change impact on water provisioning and erosion control. Sci Total Environ 458-460C:246–255. doi: 10.1016/j.scitotenv.2013.04.025

Bennett EME, Peterson GD, Gordon LJ (2009) Understanding relationships among multiple ecosystem services. Ecol Lett 12:1394–404. doi: 10.1111/j.1461-0248.2009.01387.x

Benoît M, Rizzo D, Marraccini E, et al. (2012) Landscape agronomy: a new field for addressing agricultural landscape dynamics. Landsc Ecol 27:1385–1394. doi: 10.1007/s10980-012-9802-8

Bergengren JC, Waliser DE, Yung YL (2011) Ecological sensitivity: a biospheric view of climate change. Clim Change 107:433–457. doi: 10.1007/s10584-011-0065-1

Le Blanc D, Liu W, O'Connor D, Zubcevic I (2012) Issue 1: Development cooperation in the light of sustainable development and the SDGs: Preliminary exploration of the issues. UNDESA

Boardman J (2006) Soil erosion science: Reflections on the limitations of current approaches. Catena 68:73–86. doi: 10.1016/j.catena .2006.03.007

Boardman J, Poesen J (2006) Soil Erosion in Europe. John Wiley & Sons

Bosco C, de Rigo D, Dewitte O, et al. (2014) Modelling soil erosion at European scale: towards harmonization and reproducibility. Nat Hazards Earth Syst Sci Discuss 2:2639–2680. doi: 10.5194/nhessd-2-2639-2014

Bou Kheir R, Cerdan O, Abdallah C (2006) Regional soil erosion risk mapping in Lebanon. Geomorphology 82:347–359. doi: 10.1016/j.geomorph.2 006.05.012

Braat LC, de Groot R (2012) The ecosystem services agenda:bridging the worlds of natural science and economics, conservation and development, and public and private policy. Ecosyst Serv 1:4–15. doi: 10.1016/j.ecoser.2012.07.011

Brink P ten, Mazza L, Badura T, et al. (2012) Nature and its Role in the Transition to a Green Economy.

Brouwer F, van der Heide M (2009) Multifunctional Rural land Management: Economics and Policies. Earthscan, London

Brouwer F, van Ittersum M (2010) Environmental and Agricultural Modelling. Integrated Approaches for Policy Impact Assessment. Springer

Buckwell AE, Haynes J, Davidova S, et al. (1994) Feasibility of an agricultural strategy to prepare the countries of central and eastern Europe for EU accession.

Bugalho MN, Caldeira MC, Pereira JS, et al. (2011) Mediterranean cork oak savannas require human use to sustain biodiversity and ecosystem services. Front Ecol Environ 9:278–286. doi: 10.1890/100084

Buller H, Wilson G, Höll A (2000) Agri-environmental policy in the European Union.

Burger J a, Kelting DL (1999) Using soil quality indicators to assess forest stand management. For Ecol Manage 122:155–166. doi: 10.1016/S0378-1127(99)00039-0

Burkhard B, Kroll F, Müller F (2009) Landscapes' Capacities to Provide Ecosystem Services – a Concept for Land-Cover Based Assessments. Landsc Online 15:1–22. doi: 10.3097/LO.200915

Burkhard B, Kroll F, Nedkov S, Müller F (2012) Mapping ecosystem service supply, demand and budgets. Ecol Indic 21:17–29. doi: 10.1016/j.ecolind.2011.06.019

Capolongo D, Diodato N, Mannaerts CM, et al. (2008) Analyzing temporal changes in climate erosivity using a simplified rainfall erosivity model in Basilicata (southern Italy). J Hydrol 356:119–130. doi: 10.1016/j.jhydro l.2008.04.002

Caravello G, Giacomin F (1993) Landscape ecology aspects in a territory centuriated in Roman times. Landsc Urban Plan 24:77–85.

Cardoso JC (1965) Os solos de Portugal sua classificação, caracterização e génese: 1-A sul do rio tejo. Lisboa

Carpenter SR, Mooney H a, Agard J, et al. (2009) Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. Proc Natl Acad Sci U S A 106:1305–12. doi: 10.1073/pnas.0808772106

Carpenter SSR, Bennett EME, Peterson GGD (2006) Scenarios for ecosystem services: an overview. Ecol Soc. doi: 10.1007/978-1-4614-0186-5_10

Carter MR (2002) Soil Quality for Sustainable Land Management: Organic Matter and Aggregation Interactions that Maintain Soil Functions. Agron J 94:38–47.

Castro H, Lehsten V, Lavorel S, Freitas H (2010) Functional response traits in relation to land use change in the Montado. Agric Ecosyst Environ 137:183–191. doi: 10.1016/j.agee.2010.02.002

CBD (2012) Decision X/2 Strategic Plan for Biodiversity 2011–2020 and the Aichi Biodiversity Targets. Nagoya

CEC (2006) Thematic Strategy for Soil Protection. 12.

Chapin FS, Carpenter SR, Kofinas GP, et al. (2010) Ecosystem stewardship: sustainability strategies for a rapidly changing planet. Trends Ecol Evol 25:241–9. doi: 10.1016/j.tree.2009.10.008

Charles H, Dukes JS (2007) Impacts of Invasive Species on Ecosystem Services. Biol. Invasions. pp 217–237

Chiesura A, de Groot R (2003) Critical natural capital: a socio-cultural perspective. Ecol Econ 44:219–231. doi: 10.1016/S0921-8009(02)00275-6

Chisci G, Morgan RPC (1986) Soil erosion in the european community - impact of changing agriculture. DGAgri

Coelho C, Shakesby R, Walsh R (1995) Effects of forest fires and post-fire land management practice on soil erosion and stream dynamics, Águeda basin, Portugal. Luxembourg

Cooper T, Arblaster K, Baldock D, et al. (2007) Final report for the study on HNV Indicators for evaluation.

Costa A, Madeira M, Santos Lima J (2014a) Is cork oak (Quercus suber L.) woodland loss driven by eucalyptus plantation? A case-study in southwestern Portugal. iForest - Biogeosciences For 7:193–203. doi: 10.3832/ifor1031-007

Costa A, Madeira M, Lima Santos J, et al. (2014b) Fragmentation patterns of evergreen oak woodlands in Southwestern Iberia: identifying key spatial indicators. J Environ Manage 133:18–26. doi: 10.1016/j.jenvman.201 3.11.026

Costa A, Madeira M, Lima Santos J, Oliveira Â (2011) Change and dynamics in Mediterranean evergreen oak woodlands landscapes of Southwestern Iberian Peninsula. Landsc Urban Plan 102:164–176. doi: 10.1016/j.landurbplan.2011.04.002

Costa A, Pereira H, Madeira M (2010) Analysis of spatial patterns of oak decline in cork oak woodlands in Mediterranean conditions. Ann For Sci 67:-. doi: Artn 204\rDoi 10.1051/Forest/2009097

Costa A, Pereira H, Madeira M (2009) Landscape dynamics in endangered cork oak woodlands in Southwestern Portugal (1958–2005). Agrofor Syst 77:83–96. doi: 10.1007/s10457-009-9212-3

Costanza R, Arge R, de Groot R, et al. (1997) The value of the world's ecosystem services and natural capital. Nature 387:253–260.

Costanza R, Daly H (1992) Natural capital and sustainable development. Conserv. Biol.

Costanza R, de Groot R, Sutton P, et al. (2014) Changes in the global value of ecosystem services. Glob Environ Chang 26:152–158. doi: 10.1016/j.gloenvcha.2014.04.002

Costanza R, Kubiszewski I (2012) The authorship structure of "ecosystem services" as a transdisciplinary field of scholarship. Ecosyst Serv 1:16–25. doi: 10.1016/j.ecoser.2012.06.002

Cowling RM, Egoh B, Knight AT, et al. (2008) An operational model for mainstreaming ecosystem services for implementation. Proc Natl Acad Sci U S A 105:9483–8. doi: 10.1073/pnas.0706559105

Crossman ND, Burkhard B, Nedkov S, et al. (2013) A blueprint for mapping and modelling ecosystem services. Ecosyst Serv 4:4–14. doi: 10.1016/j.ecoser.2013.02.001

Daily G, Söderqvist T, Aniyar S, et al. (2000) The value of nature and the nature of value. Science (80) 289:395–396.

Daily GC, Matson PA (2008) Ecosystem services: from theory to implementation. Proc Natl Acad Sci USA 105:9455–6. doi: 10.1073/pna s.0804960105

Daily GC, Polasky S, Goldstein J, et al. (2009) Ecosystem services in decision making: time to deliver. Front Ecol Environ 7:21–28. doi: 10.1890/080025

Darnhofer I (2014) Resilience and why it matters for farm management. Eur Rev Agric Econ 41:461–484. doi: 10.1093/erae/jbu012

Darnhofer I, Bellon S, Dedieu B, Milestad R (2010) Adaptiveness to enhance the sustainability of farming systems. A review. Agron Sustain Dev 30:545–555. doi: 10.1051/agro/2009053

Darnhofer I, Gibbon D, Didieu B (2012) Farming Systems Research into the 21st Century: The New Dynamic. Springer, London

Davison P, Hutchins MG, Anthony SG, et al. (2005) The relationship between potentially erosive storm energy and daily rainfall quantity in England and Wales. Sci Total Environ 344:15–25. doi: 10.1016/j.scitotenv.20 05.02.002

Dawson RJ, Dickson ME, Nicholls RJ, et al. (2009) Integrated analysis of risks of coastal flooding and cliff erosion under scenarios of long term change. Clim Change 95:249–288. doi: 10.1007/s10584-008-9532-8

Desmet PJJ, Govers G (1996) A GIS procedure for automatically calculating the USLE LS factor on topographically complex landscape units. J Soil Water Conserv 51:427–433.

Dibden J, Potter C, Cocklin C (2009) Contesting the neoliberal project for agriculture: Productivist and multifunctional trajectories in the European Union and Australia. J Rural Stud 25:299–308. doi: 10.1016/ j.jrurstud.2008.12.003

Diodato N, Bellocchi G (2010) MedREM, a rainfall erosivity model for the Mediterranean region. J Hydrol 387:119–127. doi: 10.1016/j.jhydrol.20 10.04.003

Diodato N, Bellocchi G (2007) Estimating monthly (R)USLE climate input in a Mediterranean region using limited data. J Hydrol 345:224–236. doi: 10.1016/j.jhydrol.2007.08.008

Drakou EG, Crossman ND, Willemen L, et al. (2015) A visualization and data-sharing tool for ecosystem service maps : Lessons learnt , challenges and the way forward. Ecosyst Serv 1–7. doi: 10.1016/j.ecoser.2014.12.002

Driesen PM (1986) Erosion hazards and conservation needs as a function of land characteristics and land qualities. In: Siderius T (Ed) L. Eval. land-use Plan. Conserv. sloping areas. ILRI, The Netherlands, pp 32–39

Drusch M, Del Bello U, Carlier S, et al. (2012) Sentinel-2: ESA's Optical High-Resolution Mission for GMES Operational Services. Remote Sens Environ 120:25–36. doi: 10.1016/j.rse.2011.11.026

Dunbar MB, Panagos P, Montanarella L (2013) European perspective of ecosystem services and related policies. Integr Environ Assess Manag 9:231–6. doi: 10.1002/ieam.1400

EC (2011) Our life insurance, our natural capital: an EU biodiversity strategy to 2020. 17.

EEA (2015) Global megatrends: Growing pressures on ecosystems (GMT 8). Eur. Environ. - state outlook 2015 A Compr. Assess. Eur. Environ. state, trends Prospect. a Glob. Context. EEA, Copenhagen, p 6 pp.

EEA (2007) European Forest Types. Categories and types for sustainable forest management reporting and policy. Copenhagen

Eigenbrod F, Armsworth PR, Anderson BJ, et al. (2010b) The impact of proxy-based methods on mapping the distribution of ecosystem services. J Appl Ecol 47:377–385. doi: 10.1111/j.1365-2664.2010.01777.x

Eigenbrod F, Armsworth PR, Anderson BJ, et al. (2010a) Error propagation associated with benefits transfer-based mapping of ecosystem services. Biol Conserv 143:2487–2493. doi: 10.1016/j.biocon.2010.06.015

Ellis EC (2011) Anthropogenic transformation of the terrestrial biosphere. Philos Trans A Math Phys Eng Sci 369:1010–1035. doi: 10.1098/rst a.2010.0331

Elmqvist T, Maltby E (2010) Biodiversity, ecosystems and ecosystem services. In: Kumar P (Ed) TEEB Ecol. Econ. Found. TEEB, p 442

Emanuelsson U, Arding M, Petersson M (2009) The Rural Landscapes of Europe: How Man has Shaped European Nature. Formas, Stockholm

Erskine WD, Mahmoudzadeh A, Myers C (2002) Land use effects on sediment yields and soil loss rates in small basins of Triassic sandstone near. Catena 49:271–287.

Faber JH, van Wensem J (2012) Elaborations on the use of the ecosystem services concept for application in ecological risk assessment for soils. Sci Total Environ 415:3–8. doi: 10.1016/j.scitotenv.2011.05.059

Fancy SG, Gross JE, Carter SL (2009) Monitoring the condition of natural resources in US national parks. Environ Monit Assess 151:161–74. doi: 10.1007/s10661-008-0257-y

Farber SC, Costanza R, Wilson MA (2002) Economic and ecological concepts for valuing ecosystem services. Ecol Econ 41:375–392. doi: 10.1016/S0921-8009(02)00088-5

Fensholt R, Proud SR (2012) Evaluation of Earth Observation based global long term vegetation trends - Comparing GIMMS and MODIS global NDVI time series. Remote Sens Environ 119:131–147. doi: 10.1016/ j.rse.2011.12.015

Fernandes RF, Vicente JR, Georges D, et al. (2014) A novel downscaling approach to predict plant invasions and improve local conservation actions. Biol Invasions. doi: 10.1007/s10530-014-0688-z

Ferreira DB (2001) Evolução da paisagem de montado no Alentejo interior al longo do Século XX: dinâmica e incidências ambientais. Finisterra XXXVI:179–193.

Figueiredo J, Pereira HM (2011) Regime shifts in a socio-ecological model of farmland abandonment. Landsc Ecol 26:737–749. doi: 10.1007/s10980-011-9605-3

Fisher B, Turner K, Zylstra M, et al. (2008) Ecosystem services and economic theory: integration for policy-relevant research. Ecol Appl 18:2050–67.

Fisher B, Turner RK, Morling P (2009) Defining and classifying ecosystem services for decision making. Ecol Econ 68:643–653. doi: 10.1016/j.ecolecon.2008.09.014

Fistikoglu O, Harmancioglu NB (2002) Integration of GIS with USLE in Assessment of Soil Erosion. Water Resour Manag 16:447–467.

Fitter A, Elmqvist T, Haines-Young R, et al. (2010) An assessment of ecosystem services and biodiversity in Europe. *in*: Hester R, Harrison R (Eds) Ecosyst. Serv. Royal Society of Chemistry, pp 1–29

Folke C, Carpenter SR, Walker B, et al. (2010) Resilience Thinking: Integrating Resilience , Adaptability and Transformability. Ecol Soc 15:[online].

Frélichová J, Vačkář D, Pártl A, et al. (2014) Integrated assessment of ecosystem services in the Czech Republic. Ecosyst Serv 8:110–117. doi: 10.1016/j.ecoser.2014.03.001

Fritsch S, Machwitz M, Ehammer A, et al. (2012) Validation of the collection 5 MODIS FPAR product in a heterogeneous agricultural landscape in arid Uzbekistan using multitemporal RapidEye imagery. Int J Remote Sens 33:37–41.

Fu B, Liu Y, Lü Y, et al. (2011) Assessing the soil erosion control service of ecosystems change in the Loess Plateau of China. Ecol Complex 8:284–293. doi: 10.1016/j.ecocom.2011.07.003

Fürst C, Helming K, Lorz C, et al. (2013) Integrated land use and regional resource management: a cross-disciplinary dialogue on future perspec-tives for a sustainable development of regional resources. J Environ Manage 1–5. doi: 10.1016/j.jenvman.2012.12.015

Galic N, Schmolke A, Forbes V, et al. (2012) The role of ecological models in linking ecological risk assessment to ecosystem services in agroecosystems. Sci Total Environ 415:93–100. doi: 10.1016/j.scitotenv.2011.05.065

García-Ruiz JM (2010) The effects of land uses on soil erosion in Spain: A review. Catena 81:1–11. doi: 10.1016/j.catena.2010.01.001

García-Ruiz JM, Lana-Renault N (2011) Hydrological and erosive consequences of farmland abandonment in Europe, with special reference to the Mediterranean region – A review. Agric Ecosyst Environ 140:317–338. doi: 10.1016/j.agee.2011.01.003

Geeson N, Brandt C, Thornes J (2002) Mediterranean Desertification: A Mosaic of Processes and Responses. John Wiley & Sons Ltd, London

Giráldez JV, Laguna A, González P (1988) Soil conservation under minimum tillage techniques in Mediterranean dry farming. In: Schwertmann U, Rickson RJ, Auerswald K (eds) Soil Eros. Prot. Meas. Eur. European Commission, Germany, pp 139–147

Gobin A, Jones R, Kirkby M, et al. (2004) Indicators for pan-European assessment and monitoring of soil erosion by water. Environ Sci Policy 7:25–38. doi: 10.1016/j.envsci.2003.09.004

Godinho S, Guiomar N, Machado R, et al. (2014) Assessment of environment, land management, and spatial variables on recent changes in montado land cover in southern Portugal. Agrofor Syst. doi: 10.1007/s10457-014-9757-7

Gómez-Baggethun E, de Groot R, Lomas PL, Montes C (2010) The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. Ecol Econ 69:1209–1218. doi: 10.1016/j.ecolecon.2009.11.007

De Groot RS, Wilson M, Boumans RM. (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol Econ 41:393–408. doi: 10.1016/S0921-8009(02) 00089-7

Grove AT, Rackam O (2001) The Nature of Mediterranean Europe. Yale University Press, London

Guerra C, Metzger MJ, Maes J, Pinto-Correia T (2015) Policy impacts on regulating ecosystem services: looking at the implications of 60 years of landscape change on soil erosion prevention in a Mediterranean silvo-pastoral system. Landsc Ecol [Accepted]

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

Haines-young R, Potschin M (2013) Common International Classification of Ecosystem Services (CICES): Consultation on Version 4 - August-December 2012. Nottingham

Haines-Young R, Potschin M, Kienast F (2012) Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs. Ecol Indic 21:39–53. doi: 10.1016/j.ecolind.2011.09.004

Hauck J, Görg C, Varjopuro R, et al. (2013) Benefits and limitations of the ecosystem services concept in environmental policy and decision making: Some stakeholder perspectives. Environ Sci Policy 25:13–21. doi: 10.1016/j.envsci.2012.08.001

Haylock MR, Hofstra N, Klein Tank a. MG, et al. (2008) A European daily high-resolution gridded data set of surface temperature and precipitation for 1950–2006. J Geophys Res 113:1–12. doi: 10.1029/2008JD010201

Helfenstein J, Kienast F (2014) Ecosystem service state and trends at the regional to national level: A rapid assessment. Ecol Indic 36:11–18. doi: 10.1016/j.ecolind.2013.06.031

Henle K, Alard D, Clitherow J, et al. (2008) Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe–A review. Agric Ecosyst Environ 124:60–71. doi: 10.1016/j.agee.2007.0 9.005

Herrick JE (2000) Soil quality: an indicator of sustainable land management? Appl Soil Ecol 15:75–83. doi: 10.1016/S0929-1393(00)00073-1

Hill J, Stellmes M, Udelhoven T, et al. (2008) Mediterranean desertification and land degradation: Mapping related land use change syndromes based on satellite observations. Glob Planet Change 64:146–157. doi: 10.1016/j.gloplacha.2008.10.005

Hoerling M, Eischeid J, Perlwitz J, et al. (2012) On the Increased Frequency of Mediterranean Drought. J Clim 25:2146–2161. doi: 10.1175/JCLI-D-11-00296.1

Holling CS (2001) Understanding the Complexity of Economic, Ecological, and Social Systems. Ecosystems 4:390–405. doi: 10.1007/s10021-001-0101-5

Hughes TP, Carpenter S, Rockström J, et al. (2013) Multiscale regime shifts and planetary boundaries. Trends Ecol Evol 28:389–395. doi: 10.1016/j.tree.2013.05.019

Iglesias A, Garrote L, Diz A, et al. (2011) Re-thinking water policy priorities in the Mediterranean region in view of climate change. Environ Sci Policy 14:744–757. doi: 10.1016/j.envsci.2011.02.007

INAG (2010) Sistema Nacional de Informação de Recursos Hídricos.

INE (2011) Recenseamento agrícola - análise dos principais resultados: 2009. Lisboa

IPCC (2007) Climate Change 2007: impacts, adaptation and vulnerability: contribution of Working Group II to the fourth assessment report of the Intergovernmental Panel.

Jiang Z, Huete AR, Chen J, et al. (2006) Analysis of NDVI and scaled difference vegetation index retrievals of vegetation fraction. Remote Sens Environ 101:366–378. doi: 10.1016/j.rse.2006.01.003

Johansen K, Roelfsema C, Phinn S (2008) High spatial resolution remote sensing for environmental monitoring and management preface: J Spat Sci 53:43–47. doi: 10.1080/14498596.2008.9635134

Jones JPG, Collen B, Atkinson G, et al. (2011a) The why, what, and how of global biodiversity indicators beyond the 2010 target. Conserv Biol J Soc Conserv Biol 25:450–7. doi: 10.1111/j.1523-1739.2010.01605.x

Jones N, de Graaff J, Rodrigo I, Duarte F (2011b) Historical review of land use changes in Portugal (before and after EU integration in 1986) and their implications for land degradation and conservation, with a focus on Centro and Alentejo regions. Appl Geogr 31:1036–1048. doi: 10.1016/j.apgeog.2011.01.024

Jose S (2009) Agroforestry for ecosystem services and environmental benefits: an overview. Agrofor Syst 76:1–10. doi: 10.1007/s10457-009-9229-7

Kirkby MJ, Jones R, Irvine B, et al. (2004) Pan-European Soil Erosion Risk Assessment: The PESERA Map. Luxembourg

Van der Knijff J, Jones R, Montanarella L (1999) Soil erosion risk assessment in Italy. Joint Research Center

Van der Knijff J, Jones R, Montanarella L (2000) Soil erosion risk assessment in Europe. Joint Research Center

Korhonen J, Seager T (2008) Beyond eco-efficiency: a resilience perspective. Bus Strateg Environ 17:411–419. doi: 10.1002/bse

Koschke L, Fürst C, Frank S, Makeschin F (2012) A multi-criteria approach for an integrated land-cover-based assessment of ecosystem services provision to support landscape planning. Ecol Indic 21:30–38. doi: 10.1016/j.ecolind.2011.12.010

Kosmas C, Danalatos N (1993) Climate change, desertification and the Mediterranean region. In: Rounsevel M, Loveland P (eds) Soil responses to Clim. Chang. Implic. Nat. Manag. Ecosyst. Silsoe, England, pp 25–38

Kosmas C, Danalatos N, Cammeraat LH, et al. (1997) The effect of land use on runoff and soil erosion rates under Mediterranean conditions. Catena 29:45–59. doi: 10.1016/S0341-8162(96)00062-8

Kosmas C, Danalatos NG, Gerontidis S (2000) The effect of land parameters on vegetation performance and degree of erosion under Mediterranean conditions. Catena 40:3–17. doi: 10.1016/S0341-8162(99)00061-2

Kouli M, Soupios P, Vallianatos F (2008) Soil erosion prediction using the Revised Universal Soil Loss Equation (RUSLE) in a GIS framework, Chania, Northwestern Crete, Greece. Environ Geol 57:483–497. doi: 10.1007/s00254-008-1318-9

Kumar P (2010) TEEB Ecological and Economic Foundations. TEEB

Langbein W, Schumm SA (1958) Yield of sediment in relation to mean annual precipitation. Am Geophys Union Trans 39:1076–1084. doi: 10.1029/TR039i006p01076

Layke C, Mapendembe A, Brown C, et al. (2012) Indicators from the global and sub-global Millennium Ecosystem Assessments: An analysis and next steps. Ecol Indic 17:77–87. doi: 10.1016/j.ecolind.2011.04.025

Leadley P, Pereira HM, Alkemade R, et al. (2010) Biodiversity Scenarios: Projections of 21st century change in biodiversity and associated ecosystem services, Technical . Secretariat of the Convention on Biological Diversity, Montreal

Lindner M, Maroschek M, Netherer S, et al. (2010) Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems. For Ecol Manage 259:698–709. doi: 10.1016/j.foreco.2009.09.023

Liu C, Frazier P, Kumar L (2007) Comparative assessment of the measures of thematic classification accuracy. Remote Sens Environ 107:606–616. doi: 10.1016/j.rse.2006.10.010

Llasat M, Llasat-Botija M, Prat M, et al. (2010) High-impact floods and flash floods in Mediterranean countries: the FLASH preliminary database. Adv Geosci 47–55.

Locatelli B, Imbach P, Vignola R, et al. (2010) Ecosystem services and hydroelectricity in Central America: modelling service flows with fuzzy logic and expert knowledge. Reg Environ Chang 11:393–404. doi: 10.1007/s10113-010-0149-x

Lomba A, Guerra C, Alonso J, et al. (2014) Mapping and monitoring High Nature Value farmlands: Challenges in European landscapes. J Environ Manage 143C:140–150. doi: 10.1016/j.jenvman.2014.04.029

López-Vicente M, Poesen J, Navas A, Gaspar L (2013) Predicting runoff and sediment connectivity and soil erosion by water for different land use scenarios in the Spanish Pre-Pyrenees. Catena 102:62–73. doi: 10.1016/j.catena.2011.01.001

Ludwig D (2000) Limitations of Economic Valuation of Ecosystems. Ecosystems 3:31–35.

Luis M, González-Hidalgo JC, Longares LA (2010) Is rainfall erosivity increasing in the Mediterranean Iberian Peninsula? L Degrad Dev 139–144.

MA (2005a) Ecosystems and Human Well-being: Current Status and Trends. Cambridge University Press, Cambridge

MA (2005b) Ecosystems and Human Well-being: Policy responses. Island Press, USA

MA (2003) Ecosystems and Human Well-Being: a Framework for Assessment. Island Press, Washington

Mace GM, Norris K, Fitter AH (2012) Biodiversity and ecosystem services: a multilayered relationship. Trends Ecol Evol 27:19–26. doi: 10.1016/j.tree.2011.08.006

Maes J, Egoh B, Willemen L, et al. (2012) Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst Serv 1:31–39. doi: 10.1016/j.ecoser.2012.06.004

Maes J, Fabrega N, Zulian G, et al. (2015) Mapping and Assessment of Ecosystems and their Services: Trends in ecosystems and ecosystem services in the European Union between 2000 and 2010. Ispra

Maes J, Hauck J, Paracchini ML, et al. (2013a) Mainstreaming ecosystem services into EU policy. Curr Opin Environ Sustain 5:128–134. doi: 10.1016/j.cosust.2013.01.002

Maes J, Paracchini M, Zulian G (2011) A European assessment of the provision of ecosystem services: towards an atlas of ecosystem services. JRC Scientific and Technical Reports

Maes J, Teller A, Erhard M, et al. (2013b) Mapping and Assessment of Ecosystems and their Services. An analytical framework for ecosystem assessments under action 5 of the EU biodiversity strategy to 2020. doi: 10.2779/12398

Maes J, Teller A, Erhard M, et al. (2014) Mapping and Assessment of Ecosystems and their Services. doi: 10.2779/75203

Mäler K-G, Aniyar S, Jansson A (2008) Accounting for ecosystem services as a way to understand the requirements for. Proc Natl Acad Sci U S A 105:9501–9506.

Martín-Fernández L, Martínez-Núñez M (2011) An empirical approach to estimate soil erosion risk in Spain. Sci Total Environ 409:3114–23. doi: 10.1016/j.scitotenv.2011.05.010

McGinnis MD, Ostrom E (2014) Social-ecological system framework: Initial changes and continuing challenges. Ecol Soc. doi: 10.5751/ES-06387-190230

Van Meijl H, van Rheenen T, Tabeau a., Eickhout B (2006) The impact of different policy environments on agricultural land use in Europe. Agric Ecosyst Environ 114:21–38. doi: 10.1016/j.agee.2005.11.006

Metzger MJ, Bunce RGH, Jongman RHG, et al. (2005) A climatic stratification of the environment of Europe. Glob Ecol Biogeogr 549–563. doi: 10.1111/j.1466-822x.2005.00190.x

Metzger MJ, Bunce RGH, Leemans R, Viner D (2008a) Projected environmental shifts under climate change: European trends and regional impacts. Environ Conserv 35:64–75. doi: 10.1017/S037689290 8004529

Metzger MJ, Rounsevell MD, Acosta-Michlik L, et al. (2006) The vulnerability of ecosystem services to land use change. Agric Ecosyst Environ 114:69–85. doi: 10.1016/j.agee.2005.11.025

Metzger MJ, Schröter D, Leemans R, Cramer W (2008b) A spatially explicit and quantitative vulnerability assessment of ecosystem service change in Europe. Reg Environ Chang 8:91–107. doi: 10.1007/s10113-008-0044-x

Moore ID, Burch GJ (1986) Physical Basis of the Length-slope Factor in the Universal Soil Loss Equation. Soil Sci Soc Am J 50:1294. doi: 10.2136/sssaj1986.03615995005000050042x

Moreno G, Franca A, Pinto-Correia T, Godinho S (2014) Multifunctionality and dynamics of silvopastoral systems. Options Méditerranéennes A:421–436.

Morgan RPC (1986) Soil degradation and soil erosion in the loamy belt of northern Europe. In: Chisci G, Morgan RPC (eds) Soil Eros. Eur. Community - Impact Chang. Agric. DGAgri, 165–172

Morgan RPC (2005) Soil Erosion & Conservation, Third edit. Blackwell Publishing, Cornwall, UK

Müller F, Burkhard B (2012) The indicator side of ecosystem services. Ecosyst Serv 1:26–30. doi: 10.1016/j.ecoser.2012.06.001

Naidoo R, Balmford a, Costanza R, et al. (2008) Global mapping of ecosystem services and conservation priorities. Proc Natl Acad Sci U S A 105:9495–500. doi: 10.1073/pnas.0707823105

Navarra A, Tubiana L (2013) Regional Assessment of Climate Change in the Mediterranean. doi: 10.1007/978-94-007-5772-1

Nedkov S, Burkhard B (2011) Flood regulating ecosystem services - Mapping supply and demand, in the Etropole municipality, Bulgaria. Ecol Indic. doi: 10.1016/j.ecolind.2011.06.022

Nelson E, Mendoza G, Regetz J, et al. (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front Ecol Environ 7:4–11. doi: 10.1890/080023

Nunes JT (2007) Livestock production in the “montado” - pigs. Rev Ciências Agrárias 30:251–259.

Ojima D, Galvin K, Turner B (1994) The Global Impact of Land-use Change. Bioscience

Olea L, Miguel-ayanz AS (2006) The Spanish dehesa, a traditional Mediterranean silvopastoral system. 21st Gen. Meet. Eur. Grassl. Fed. Badajoz, pp 1–15

Olesen JE, Trnka M, Kersebaum KC, et al. (2011) Impacts and adaptation of European crop production systems to climate change. Eur J Agron 34:96–112. doi: 10.1016/j.eja.2010.11.003

Oñate JJ, Atance I, Bardají I, Llusia D (2007) Modelling the effects of alternative CAP policies for the Spanish high-nature value cereal-steppe farming systems. Agric Syst 94:247–260. doi: 10.1016/j.agsy.2006.09.003

Ortega M, Guerra C, Honrado JP, et al. (2013) Surveillance of habitats and plant diversity indicators across a regional gradient. Ecol Indic 33:36–44. doi: 10.1016/j.ecolind.2012.12.004

Otero I, Boada M, Badia A, et al. (2011) Loss of water availability and stream biodiversity under land abandonment and climate change in a Mediterranean catchment (Olzinelles, NE Spain). Land use policy 28:207–218. doi: 10.1016/j.landusepol.2010.06.002

Van Oudenhoven APE, Petz K, Alkemade R, et al. (2012) Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecol Indic 21:110–122. doi: 10.1016/j.ecolind.20 12.01.012

Panagos P, Borrelli P, Meusburger K, et al. (2015) Modelling the effect of support practices (P-factor) on the reduction of soil erosion by water at European scale. Environ Sci Policy 51:23–34. doi: 10.1016/j.envsci.20 15.03.012

Panagos P, Hiederer R, Van Liedekerke M, Bampa F (2013) Estimating soil organic carbon in Europe based on data collected through an European network. Ecol Indic 24:439–450. doi: 10.1016/j.ecolind.2012.07.020

Panagos P, Jones A, Bosco C, Kumar PSS (2011a) European digital archive on soil maps (EuDASM): preserving important soil data for public free access. Int J Digit Earth 4:434–443. doi: 10.1080/17538947.2011.59 6580

Panagos P, Karydas C, Borrelli P, et al. (2014a) Advances in soil erosion modelling through remote sensing data availability at European scale. In: Hadjimitsis DG, Themistocleous K, Michaelides S, Papadavid G (eds) Second Int. Conf. Remote Sens. Geoinf. Environ. pp 92290I–1 – 92290I–10

Panagos P, Karydas CG, Gitas IZ (2011b) Monthly soil erosion monitoring based on remotely sensed biophysical parameters: a case study in Strymonas river basin towards a functional pan-European service. Int J Digit Earth 37–41.

Panagos P, Meusburger K, Alewell C, Montanarella L (2012) Soil erodibility estimation using LUCAS point survey data of Europe. Environ Model Softw 30:143–145. doi: 10.1016/j.envsoft.2011.11.002

Panagos P, Meusburger K, Ballabio C, et al. (2014b) Soil erodibility in Europe: A high-resolution dataset based on LUCAS. Sci Total Environ 479-480C:189–200. doi: 10.1016/j.scitotenv.2014.02.010

Paracchini ML, Petersen J-E, Hoogeveen Y, et al. (2008) High Nature Value Farmland in Europe: An estimate of the distribution patterns on the basis of land cover and biodiversity data, JRC, p. 102

Pe'er G, Dicks L, Visconti P, et al. (2014) EU agricultural reform fails on biodiversity. Science (80) 344:1090–1092.

Pereira H, Ferrier S, Walters M, et al. (2013) Essential Biodiversity Variables. Science (80) 339:277–278.

Pereira HM, Navarro LM, Martins IS (2012a) Global Biodiversity Change: The Bad, the Good, and the Unknown. Annu Rev Environ Resour 37:25–50. doi: 10.1146/annurev-environ-042911-093511

Pereira P, Godinho C, Gomes M, Rabaça JE (2012b) The importance of the surroundings: are bird communities of riparian galleries influenced by agroforestry matrices in SW Iberian Peninsula? Ann For Sci 71:33–41. doi: 10.1007/s13595-012-0228-x

Pereira PM, Fonseca MP (2003) Nature vs Nurture : the Making of the Montado Ecosystem. Conserv. Ecol. 7:

Perrings C, Folke C, Mäler K (1992) The Ecology and Economics of Biodiversity Loss: The Research Agenda. Ambio 21:201–211.

Pettorelli N, Laurance WF, O'Brien TG, et al. (2014) Satellite remote sensing for applied ecologists: Opportunities and challenges. J Appl Ecol 51:839–848. doi: 10.1111/1365-2664.12261

Pinto-Correia T (2000) Future development in Portuguese rural areas: how to manage agricultural support for landscape conservation? Landsc Urban Plan 50:95–106. doi: 10.1016/S0169-2046(00)00082-7

Pinto-Correia T (1993) Threatned landscape in Alentejo, Portugal: the "montado" and other 'agro-silvo-pastoral' systems. Landsc Urban Plan 24:43–48.

Pinto-Correia T, Godinho S (2013) Changing agriculture – changing landscape: what is going on in the high valued Montado landscapes of Southern Portugal? In: Ortiz-Miranda D, Moragues-Faus AM, Arnalte-Alegre E (eds) Agric. Mediterr. Eur. between old new Paradig., Research i. Emerald, pp 75–90

Pinto-Correia T, Kristensen L (2013) Linking research to practice: The landscape as the basis for integrating social and ecological perspectives of the rural. Landsc Urban Plan 120:248–256. doi: 10.1016/j.landurb plan.2013.07.005

Pinto-Correia T, Machado C, Barroso F, et al. (2013) How do policy options modify landscape amenities? An assessment approach based on public expressed preferences. Environ Sci Policy 1–11. doi: 10.1016/ j.envsci.2013.01.016

Pinto-Correia T, Mascarenhas J (1999) Contribution to the extensification/ intensification debate : new trends in the Portuguese montado. Landsc Urban Plan 46:125–131.

Pinto-Correia T, Menezes H, Barroso LF (2014) The Landscape as an Asset in Southern European Fragile Agricultural Systems: Contrasts and Contradictions in Land Managers Attitudes and Practices. Landsc Res 39:205–217. doi: 10.1080/01426397.2013.790948

Pinto-Correia T, Primdahl J (2009) When rural landscapes change functionality: constraints and development options for multifunctional landscapes: examples from contrasting case-studies in Portugal and Denmark. In: Brouwer F, van der Heide M (Eds) Multifunct. Rural L. Manag. Econ. Policies. Earthscan, pp 213–234

Pinto-Correia T, Ribeiro N, Sá-Sousa P (2011) Introducing the montado, the cork and holm oak agroforestry system of Southern Portugal. Agrofor Syst 82:99–104. doi: 10.1007/s10457-011-9388-1

Pinto-Correia T, Vos W (2004) Multifunctionality in Mediterranean landscapes-past and future. In: Jongman RHG (Ed) new Dimens. Eur. Landsc. Wageningen EU Frontis Series, Springer, pp 135–164

Plieninger T, Bieling C (2013) Resilience-based perspectives to guiding high-nature-value farmland through socioeconomic change. Ecol Soc 18:20. doi: 10.5751/ES-05877-180420

Plieninger T, Dijks S, Oteros-Rozas E, Bieling C (2013) Assessing, mapping, and quantifying cultural ecosystem services at community level. Land use policy 33:118–129. doi: 10.1016/j.landusepol.2012.12.013

Podmanicky L, Balázs K, Belényesi M, et al. (2011) Modelling soil quality changes in Europe. An impact assessment of land use change on soil quality in Europe. Ecol Indic 11:4–15. doi: 10.1016/j.ecolind.20 09.08.002

Power AG (2010) Ecosystem services and agriculture: tradeoffs and synergies. Philos Trans R Soc Lond B Biol Sci 365:2959–71. doi: 10.1098/ rstb.2010.0143

Prasannakumar V, Vijith H, Abinod S, Geetha N (2012) Estimation of soil erosion risk within a small mountainous sub-watershed in Kerala, India, using Revised Universal Soil Loss Equation (RUSLE) and geo-information technology. Geosci Front 3:209–215. doi: 10.1016/ j.gsf.2011.11.003

Prasuhn V, Liniger H, Gisler S, et al. (2013) A high-resolution soil erosion risk map of Switzerland as strategic policy support system. Land use policy 32:281–291. doi: 10.1016/j.landusepol.2012.11.006

Presbitero AL, Escalante MC, Rose CW, et al. (1995) Erodibility evaluation and the effect of land management practices on soil erosion from steep slopes in Leyte, the Philippines. Soil Technol 8:205–213.

Primdahl J, Andersen E, Swaffield S, Kristensen L (2013) Intersecting Dynamics of Agricultural Structural Change and Urbanisation within European Rural Landscapes: Change Patterns and Policy Implications. Landsc Res 38:799–817. doi: 10.1080/01426397.2013.772959

Purevdorj T, Tateishi R, Ishiyam T, Honda Y (1998) Relationships between percent vegetation cover and vegetation indices. Int Jour Rem Sens 19:3519–3535.

Rabbinge R, van Diepen C a. (2000) Changes in agriculture and land use in Europe. Eur J Agron 13:85–99. doi: 10.1016/S1161-0301(00)00067-8

Rapport DJ, Costanza R, McMichael a J (1998) Assessing ecosystem health. Trends Ecol Evol 13:397–402.

Van Remortel RD, Maichle RW, Hickey RJ (2004) Computing the LS factor for the Revised Universal Soil Loss Equation through array-based slope processing of digital elevation data using a C++ executable. Comput Geosci 30:1043–1053. doi: 10.1016/j.cageo.2004.08.001

Renard K, Foster G, Weessies G, McCool D (1997) Predicting soil erosion bywater: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). Agric. Handb. 703

Renwick A, Jansson T, Verburg PH, et al. (2013) Policy reform and agricultural land abandonment in the EU. Land use policy 30:446–457. doi: 10.1016/j.landusepol.2012.04.005

Reyers B, Biggs R, Cumming GS, et al. (2013) Getting the measure of ecosystem services: a social–ecological approach. Front Ecol Environ 11:268–273. doi: 10.1890/120144

Ribeiro NA, Dias S, Surovy P, et al. (2004) The importance of crown cover on the sustainability of cork oak stands. In: Schnabel S, Ferreira A (eds) Adv. GeoEcology 37 - Sustain. Agrosilvopastoral Syst. CATENA VERLAG, pp 275–286

Ribeiro PPF, Santos JLJ, Bugalho MNM, et al. (2014) Modelling farming system dynamics in High Nature Value Farmland under policy change. Agric Ecosyst Environ 183:138–144. doi: 10.1016/j.agee.2013.11.002

Rickson RJ (2014) Can control of soil erosion mitigate water pollution by sediments? Sci Total Environ 468-469:1187–97. doi: 10.1016/j.scitot env.2013.05.057

Rigueiro-Rodríguez A, McAdam J, Mosquera-Losada M (2009) Agroforestry in Europe: Current Status and Future Prospects. Springer

Van Rompaey A, Bazzoffi P, Jones RJA, et al. (2003) Validation of soil erosion risk assessment in Italy.

Rounsevell MD a., Pedroli B, Erb K-H, et al. (2012) Challenges for land system science. Land use policy 29:899–910. doi: 10.1016/j.landusepol.20 12.01.007

Rounsevell MD, Dawson TP, Harrison P a. (2010) A conceptual framework to assess the effects of environmental change on ecosystem services. Biodivers Conserv 19:2823–2842. doi: 10.1007/s10531-010-9838-5

Rundel P, Montenegro G, Jaksic F (1998) Landscape disturbance and biodiversity in Mediterranean-type ecosystems. Springer

Sales-Baptista E, Abreu MC, Ferraz-de-oliveira MI (2015) Overgrazing in the Montado? The need for monitoring grazing pressure at paddock scale. Agrofor Syst. doi: 10.1007/s10457-014-9785-3

Santana J, Reino L, Stoate C, et al. (2013) Mixed effects of long-term conservation investment in Natura 2000 farmland. Conserv Lett 00:n/a–n/a. doi: 10.1111/conl.12077

Santos-Martín F, Martín-López B, García-Llorente M, et al. (2013) Unraveling the Relationships between Ecosystems and Human Wellbeing in Spain. PLoS One. doi: 10.1371/journal.pone.0073249

SCBD-UNEP (2010) Nagoya Declaration on Biodiversity in Development Cooperation. Nagoya

Schäfer RB (2012) Biodiversity, ecosystem functions and services in environmental risk assessment: introduction to the special issue. Sci Total Environ 415:1–2. doi: 10.1016/j.scitotenv.2011.08.012

Schägner JP, Brander L, Maes J, Hartje V (2013) Mapping ecosystem services' values: Current practice and future prospects. Ecosyst Serv 4:33–46. doi: 10.1016/j.ecoser.2013.02.003

Scholes R, Reyers B, Biggs R, et al. (2013) Multi-scale and cross-scale assessments of social–ecological systems and their ecosystem services. Curr Opin Environ Sustain 5:16–25. doi: 10.1016/j.cosust.2013.01.004

Scholze M, Knorr W, Arnell NW, Prentice IC (2006) A climate-change risk analysis for world ecosystems. Proc Natl Acad Sci U S A 103:13116–20. doi: 10.1073/pnas.0601816103

Schröder C (2011) Land use dynamics in the dehesas in the Sierra Morena (Spain): the role of diverse management strategies to cope with the drivers of change. Eur Countrys 3:11–28. doi: 10.2478/v10091-011-0006-z

Schröter D, Cramer W, Leemans R, et al. (2005) Ecosystem service supply and vulnerability to global change in Europe. Science 310:1333–7. doi: 10.1126/science.1115233

Schuler J, Sattler C (2010) The estimation of agricultural policy effects on soil erosion—An application for the bio-economic model MODAM. Land use policy 27:61–69. doi: 10.1016/j.landusepol.2008.05.001

Schwertmann U, Rickson RJ, Auerswald K (1988) Soil erosion protection measures in Europe. European Commission

Seppelt R, Dormann CF, Eppink FV, et al. (2011) A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. J Appl Ecol 48:630–636. doi: 10.1111/j.1365-2664.2010.01952.x

Seto KC, Fleishman E, Fay JP, Betrus CJ (2004) Linking spatial patterns of bird and butterfly species richness with Landsat TM derived NDVI. Int J Remote Sens 25:4309–4324. doi: 10.1080/0143116042000192358

Shakesby R (2011) Post-wildfire soil erosion in the Mediterranean: Review and future research directions. Earth-Science Rev 105:71–100. doi: 10.1016/j.earscirev.2011.01.001

Shrestha S, Ciaian P, Himics M, Doorslaer B (2013) Impacts of climate change on EU agriculture. Rev Agric Appl Econ 24–39.

Spangenberg JH, Settele J (2010) Precisely incorrect? Monetising the value of ecosystem services. Ecol Complex 7:327–337. doi: 10.1016/j.ecocom.20 10.04.007

Surová D, Pinto-Correia T (2008) Landscape preferences in the cork oak Montado region of Alentejo, southern Portugal: Searching for valuable landscape characteristics for different user groups. Landsc Res 33:311–330. doi: 10.1080/01426390802045962

Surová D, Pinto-Correia T, Marušák R (2013) Visual complexity and the montado do matter: landscape pattern preferences of user groups in Alentejo, Portugal. Ann For Sci. doi: 10.1007/s13595-013-0330-8

Sutherland L, Darnhofer I, Wilson G, Zagata L (2014) Transition pathways towards sustainability in European agriculture. CAB International

Swaffield S, Primdahl J (2010) Globalisation and local agricultural landscapes: patterns of change, policy dilemmas and research questions. In: Primdahl J, Swaffield S (eds) Glob. Agric. Landscapes - Chang. Patterns Policy Trends Dev. Ctries. Cambridge University Press, Cambridge, pp 245–270

Syrbe R-U, Walz U (2012) Spatial indicators for the assessment of ecosystem services: Providing, benefiting and connecting areas and landscape metrics. Ecol Indic 21:80–88. doi: 10.1016/j.ecolind.2012.02.013

Tallis H, Kareiva P, Marvier M, Chang A (2008) An ecosystem services framework to support both practical conservation and economic development Heather. Proc Natl Acad Sci U S A 105:9457–9464.

Tallis H, Mooney H, Andelman S, et al. (2012) A Global System for Monitoring Ecosystem Service Change. Bioscience 62:977–986. doi: 10.1525/bio.2012.62.11.7

Thuiller W, Albert C, Araújo MB, et al. (2008) Predicting global change impacts on plant species' distributions: Future challenges. Perspect Plant Ecol Evol Syst 9:137–152. doi: 10.1016/j.ppees.2007.09.004

Tress B, Tress G (2001) Capitalising on multiplicity: a transdisciplinary systems approach to landscape research. Landsc Urban Plan 57:143–157. doi: 10.1016/S0169-2046(01)00200-6

Van-Camp L, Bujarrabal B, Gentile AR, et al. (2004) Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection. Luxembourg

Vanslembrouck I, Van Huyenbroeck G (2005) Landscape Amenities: Economic Assessment of Agricultural Landscapes. Springer, Netherlands

Vanwalleghem T, Laguna A, Giráldez JV, Jiménez-Hornero FJ (2010) Applying a simple methodology to assess historical soil erosion in olive orchards. Geomorphology 114:294–302. doi: 10.1016/j.geomorph.20 09.07.010

De Vente J, Poesen J, Verstraeten G, et al. (2008) Spatially distributed modelling of soil erosion and sediment yield at regional scales in Spain. Glob Planet Change 60:393–415. doi: 10.1016/j.gloplacha.2007.05.002

Verheijen FG, Jones RJ, Rickson RJ, Smith CJ (2009) Tolerable versus actual soil erosion rates in Europe. Earth-Science Rev 94:23–38. doi: 10.1016/j.earscirev.2009.02.003

Verheijen FGA, Jones RJA, Rickson RJ, et al. (2013) Concise overview of European soil erosion research and evaluation. Acta Agric Scand Sect B - Soil Plant Sci 62:185–190.

Vicente JR, Fernandes RF, Randin CF, et al. (2013a) Will climate change drive alien invasive plants into areas of high protection value? An improved model-based regional assessment to prioritise the management of invasions. J Environ Manage 131:185–195. doi: 10.1016/j.jenvman.20 13.09.032

Vicente JR, Pinto AT, Araújo MB, et al. (2013b) Using Life Strategies to Explore the Vulnerability of Ecosystem Services to Invasion by Alien Plants. Ecosystems 16:678–693. doi: 10.1007/s10021-013-9640-9

Viglizzo EF, Paruelo JM, Laterra P, Jobbágy EG (2012) Ecosystem service evaluation to support land-use policy. Agric Ecosyst Environ 154:78–84. doi: 10.1016/j.agee.2011.07.007

Vihervaara P, Kumpula T, Tanskanen A, Burkhard B (2010) Ecosystem services–A tool for sustainable management of human–environment systems. Case study Finnish Forest Lapland. Ecol Complex 7:410–420. doi: 10.1016/j.ecocom.2009.12.002

Van Vliet J, de Groot HLF, Rietveld P, Verburg PH (2015) Manifestations and underlying drivers of agricultural land use change in Europe. Landsc Urban Plan 133:24–36. doi: 10.1016/j.landurbplan.2014.09.001

Vrieling A (2006) Satellite remote sensing for water erosion assessment: A review. Catena 65:2–18. doi: 10.1016/j.catena.2005.10.005

Wallace KJ (2007) Classification of ecosystem services: Problems and solutions. Biol Conserv 139:235–246. doi: 10.1016/j.biocon.2007.07.015

Wang ZY, Wang G, Huang G (2008) Modeling of state of vegetation and soil erosion over large areas. Int J Sediment Res 23:181–196. doi: 10.1016/S1001-6279(08)60018-9

Wawer R, Nowocieñ E, Podolski B (2005) Real and Calculated K USLE Erodibility Factor for Selected Polish Soils. Polish J Environ Stud 14:655–658.

Van Wijnen HJ, Rutgers M, Schouten AJ, et al. (2012) How to calculate the spatial distribution of ecosystem services--natural attenuation as example from The Netherlands. Sci Total Environ 415:49–55. doi: 10.1016/ j.scitotenv.2011.05.058

Williams L, Kapustka L (2000) Ecosystem vulnerability: a complex interface with technical components. Environ Toxicol Chem 19:1055–1058.

Wischmeier W, Smith D (1978) Predicting Rainfall Erosion Losses. A Guide to Conservation Planning.

Woods M (2005) Rural geography. Sage Publications, London

Woods M (2011) Rural. Routledge, London

Yu B, Hashim GM, Eusof Z (2001) Estimating the r-factor with limited rainfall data: a case study from peninsular Malaysia. J Soil Water Conserv 56:101–105.

Zhang XC, Nearing MA (2005) Impact of climate change on soil erosion, runoff, and wheat productivity in central Oklahoma. Catena 61:185–195. doi: 10.1016/j.catena.2005.03.009

Zhongming W, Lees BG, Feng J, et al. (2010) Stratified vegetation cover index: A new way to assess vegetation impact on soil erosion. Catena 83:87–93. doi: 10.1016/j.catena.2010.07.006

APPENDIX

Appendix 1 Pearson correlation values between the calculated indicators and correspondent significance values (p).

Columns 1–13:

Correlations		Grazing intensity	Variation in grazing intensity	Total arable land and permanent crops	Total agro-forestry area	Total area of permanent pastures	Total forest area	% of changing area	% of maintenance area	% of artificial surfaces	% of irrigated land	% of non-irrigated land	% of olive groves	% of orchards
Grazing intensity	r	1	-0.771	-0.788	-0.72	0.774	0.764	-0.932	0.932	.973*	0.558	-0.861	.950*	0.819
	p		0.439	0.212	0.28	0.226	0.236	0.236	0.236	0.027	0.442	0.139	0.05	0.181
Variation in grazing intensity	r	-0.771	1	0.954	-0.347	-0.938	0.086	0.487	-0.487	-0.852	0.6	.997*	-0.462	-.998*
	p	0.439		0.194	0.775	0.226	0.945	0.676	0.676	0.351	0.59	0.05	0.694	0.039
Total arable land and permanent crops	r	-0.788	0.954	1	0.318	-.904*	-.838*	0.551	-0.551	-.890*	-0.531	.986**	-0.659	-0.809
	p	0.212	0.194		0.538	0.013	0.037	0.336	0.336	0.017	0.279	0	0.154	0.051
Total agro-forestry area	r	-0.72	-0.347	0.318	1	-0.672	-0.388	0.781	-0.781	-0.555	-0.22	0.35	-0.434	-0.257
	p	0.28	0.775	0.538		0.144	0.447	0.119	0.119	0.253	0.675	0.497	0.39	0.623
Total area of permanent pastures	r	0.774	-0.938	-.904*	-0.672	1	0.738	-0.712	0.712	.898*	0.388	-.887*	0.629	0.708
	p	0.226	0.226	0.013	0.144		0.094	0.177	0.177	0.015	0.447	0.019	0.181	0.115
Total forest area	r	0.764	0.086	-.838*	-0.388	0.738	1	-.931*	.931*	.850*	.896*	-.905*	0.788	0.774
	p	0.236	0.945	0.037	0.447	0.094		0.022	0.022	0.032	0.016	0.013	0.063	0.071
% of changing area	r	-0.932	0.487	0.551	0.781	-0.712	-.931*	1	-1.000**	-.941*	-0.392	0.764	-.884*	-0.136
	p	0.236	0.676	0.336	0.119	0.177	0.022		0	0.017	0.513	0.133	0.047	0.827
% of maintenance area	r	0.932	-0.487	-0.551	-0.781	0.712	.931*	-1.000**	1	.941*	0.392	-0.764	.884*	0.136
	p	0.236	0.676	0.336	0.119	0.177	0.022	0		0.017	0.513	0.133	0.047	0.827
% of artificial surfaces	r	.973*	-0.852	-.890*	-0.555	.898*	.850*	-.941*	.941*	1	0.608	-.932**	.898*	0.69
	p	0.027	0.351	0.017	0.253	0.015	0.032	0.017	0.017		0.201	0.007	0.015	0.13
% of irrigated land	r	0.558	0.6	-0.531	-0.22	0.388	.896*	-0.392	0.392	0.608	1	-0.642	0.699	0.516
	p	0.442	0.59	0.279	0.675	0.447	0.016	0.513	0.513	0.201		0.169	0.123	0.294
% of non-irrigated land	r	-0.861	.997*	.986**	0.35	-.887*	-.905*	0.764	-0.764	-.932**	-0.642	1	-0.763	-.826*
	p	0.139	0.05	0	0.497	0.019	0.013	0.133	0.133	0.007	0.169		0.078	0.043
% of olive groves	r	.950*	-0.462	-0.659	-0.434	0.629	0.788	-.884*	.884*	.898*	0.699	-0.763	1	0.566
	p	0.05	0.694	0.154	0.39	0.181	0.063	0.047	0.047	0.015	0.123	0.078		0.241
% of orchards	r	0.819	-.998*	-0.809	-0.257	0.708	0.774	-0.136	0.136	0.69	0.516	-.826*	0.566	1
	p	0.181	0.039	0.051	0.623	0.115	0.071	0.827	0.827	0.13	0.294	0.043	0.241	
% of agro-forestry areas with >50% of tree cover	r	-0.94	0.931	0.69	.885*	-.916*	-0.63	.880*	-.880*	-.859*	-0.348	0.71	-0.687	-0.526
	p	0.06	0.238	0.129	0.019	0.01	0.18	0.049	0.049	0.028	0.499	0.114	0.131	0.283
% of agro-forestry areas with 30-50% of tree cover	r	.984*	-0.697	-0.653	-0.476	0.675	0.658	-0.824	0.824	.877*	0.465	-0.731	.940**	0.642
	p	0.016	0.509	0.16	0.34	0.141	0.156	0.086	0.086	0.022	0.353	0.099	0.005	0.169
% of agro-forestry areas with 30-50% of tree cover	r	0.539	-0.52	-0.705	-0.769	.906*	0.543	-0.462	0.462	0.66	0.183	-0.665	0.342	0.658
	p	0.461	0.652	0.117	0.074	0.013	0.266	0.434	0.434	0.154	0.728	0.15	0.507	0.156
% of agro-forestry areas with <30% of tree cover and	r	-0.469	-0.621	0.084	-0.234	0.055	-0.135	-0.605	0.605	0.142	-0.08	0.084	0.137	-0.61
	p	0.531	0.574	0.874	0.656	0.918	0.798	0.28	0.28	0.789	0.88	0.875	0.795	0.198
% of agro-forestry areas with <30% of tree cover and	r	0.932	-0.983	-.912*	-0.573	.951**	0.765	-0.837	0.837	.977**	0.446	-.923**	0.806	0.705
	p	0.068	0.117	0.011	0.234	0.004	0.076	0.077	0.077	0.001	0.376	0.009	0.053	0.118
% of permanent pastures	r	0.774	-0.938	-.904*	-0.672	1.000**	0.738	-0.712	0.712	.898*	0.388	-.887*	0.629	0.708
	p	0.226	0.226	0.013	0.144	0	0.094	0.177	0.177	0.015	0.447	0.019	0.181	0.115
% of mixed forest	r	0.895	-0.971	-.956**	-0.514	.936**	.889*	-.884*	.884*	.979**	0.624	-.977**	.811*	0.746
	p	0.105	0.153	0.003	0.297	0.006	0.018	0.046	0.046	0.001	0.186	0.001	0.05	0.089
% of production forest	r	0.718	0.353	-.818*	-0.352	0.703	.997**	-0.845	0.845	0.808	.910*	-.884*	0.745	0.772
	p	0.282	0.77	0.047	0.494	0.119	0	0.071	0.071	0.052	0.012	0.019	0.089	0.072
% of shrubs and/or herbaceous vegetation	r	0.874	-0.502	-0.462	-0.478	0.513	0.561	-.892*	.892*	0.796	0.486	-0.567	.948**	0.355
	p	0.126	0.665	0.356	0.338	0.298	0.247	0.042	0.042	0.058	0.328	0.241	0.004	0.49
% of water surfaces	r	0.869	-0.919	-.957**	-0.361	.852*	.950**	-.887*	.887*	.926**	0.726	-.990**	0.798	.852*
	p	0.131	0.258	0.003	0.482	0.031	0.004	0.045	0.045	0.008	0.102	0	0.057	0.031
% of changing area	r	-0.957	0.552	0.584	0.836	-0.771	-.906*	.992**	-.992**	-.942*	-0.315	0.78	-0.846	-0.235
	p	0.188	0.628	0.301	0.078	0.127	0.034	0.001	0.001	0.017	0.606	0.119	0.071	0.703
% of maintenance area	r	0.957	-0.552	-0.584	-0.836	0.771	.906*	-.992**	.992**	.942*	0.315	-0.78	0.846	0.235
	p	0.188	0.628	0.301	0.078	0.127	0.034	0.001	0.001	0.017	0.606	0.119	0.071	0.703
Number of patches	r	0.924	-0.915	-.930**	-0.56	.932**	.886*	-.927*	.927*	.991**	0.629	-.962**	.847*	0.742
	p	0.076	0.264	0.007	0.248	0.007	0.019	0.023	0.023	0	0.181	0.002	0.033	0.092
Total edge	r	0.924	-0.895	-.926**	-0.568	.930**	.894*	-.932*	.932*	.988**	0.637	-.961**	.852*	0.769
	p	0.076	0.294	0.008	0.24	0.007	0.016	0.021	0.021	0	0.174	0.002	0.031	0.074
Mean patch size	r	-0.87	0.936	.953**	0.5	-.917*	-.933**	.917*	-.917*	-.962**	-0.688	.983**	-0.808	-0.804
	p	0.13	0.229	0.003	0.313	0.01	0.007	0.028	0.028	0.002	0.131	0	0.052	0.054
Shannon's diversity index	r	0.862	-0.238	-0.799	-0.638	.829*	.940**	-.981**	.981**	.935**	0.796	-.874*	.881*	0.7
	p	0.138	0.847	0.056	0.172	0.042	0.005	0.003	0.003	0.006	0.058	0.023	0.02	0.122
Structural impact	r	-.982*	0.634	0.393	.940**	-0.709	-0.398	.903*	-.903*	-0.682	-0.186	0.436	-0.616	-0.328
	p	0.018	0.563	0.441	0.005	0.115	0.434	0.036	0.036	0.136	0.724	0.387	0.193	0.525
Ecosystem service mitigated impact	r	.990**	-0.742	-0.596	0.284	0.287	0.594	-0.2	0.2	0.603	0.575	-0.649	0.695	0.384
	p	0.01	0.468	0.212	0.585	0.581	0.214	0.747	0.747	0.205	0.232	0.163	0.126	0.452
Actual ecosystem service provision	r	-.983*	0.643	0.432	.925**	-0.729	-0.437	.920*	-.920*	-0.722	-0.223	0.478	-0.662	-0.354
	p	0.017	0.555	0.392	0.008	0.1	0.386	0.027	0.027	0.105	0.671	0.337	0.152	0.492
Ecosystem service provision capacity	r	0.027	0.498	0.238	0.504	-0.503	0.136	0.137	-0.137	-0.126	0.512	0.126	0.215	-0.241
	p	0.973	0.668	0.65	0.309	0.309	0.797	0.826	0.826	0.812	0.299	0.812	0.683	0.645
Variation in structural impact	r	0.424	-0.904	0.036	.917*	-0.5	-0.665	0.561	-0.561	-0.353	-0.34	0.15	-0.26	-0.273
	p	0.721	0.282	0.954	0.028	0.391	0.221	0.325	0.325	0.56	0.576	0.81	0.672	0.657
Variation in ecosystem service provision	r	0.381	-0.882	0.068	.931*	-0.527	-0.689	0.592	-0.592	-0.389	-0.343	0.186	-0.289	-0.268
	p	0.751	0.312	0.913	0.022	0.362	0.199	0.293	0.293	0.518	0.572	0.765	0.637	0.663
Rate of effective ecosystem service provision	r	-0.985	0.651	0.646	0.846	-0.825	-0.876	.963**	-.963**	-.923*	-0.279	0.796	-0.734	-0.063
	p	0.109	0.549	0.239	0.071	0.085	0.051	0.009	0.009	0.025	0.65	0.107	0.158	0.919
Variation in ecosystem service provision capacity	r	0.631	0.008	0.093	0.414	-0.344	0.182	-0.104	0.104	0.084	0.501	-0.008	0.301	-0.785
	p	0.566	0.995	0.881	0.488	0.571	0.769	0.868	0.868	0.893	0.39	0.99	0.622	0.116
Variation in ecosystem mitigated impact	r	0.58	-0.966	-0.25	0.723	-0.225	-0.431	0.269	-0.269	-0.035	-0.304	-0.164	-0.034	-0.335
	p	0.606	0.167	0.685	0.167	0.716	0.469	0.662	0.662	0.955	0.619	0.792	0.956	0.582

Columns 14–25:

Correlations		% of agro-forestry areas with >50% of tree cover	% of agro-forestry areas with 30-50% of tree cover and >50% of shrubs	% of agro-forestry areas with 30-50% of tree cover and <50% of shrubs	% of agro-forestry areas with <30% of tree cover and >50% of shrubs	% of agro-forestry areas with <30% of tree cover and <50% of shrubs	% of permanent pastures	% of mixed forest	% of production forest	% of shrubs and/or herbaceous vegetation associations	% of water surfaces	% of changing area	% of maintenance area
Grazing intensity	r	-0.94	.984*	0.539	-0.469	0.932	0.774	0.895	0.718	0.874	0.869	-0.957	0.957
	p	0.06	0.016	0.461	0.531	0.068	0.226	0.105	0.282	0.126	0.131	0.188	0.188
Variation in grazing intensity	r	0.931	-0.697	-0.52	-0.621	-0.983	-0.938	-0.971	0.353	-0.502	-0.919	0.552	-0.552
	p	0.238	0.509	0.652	0.574	0.117	0.226	0.153	0.77	0.665	0.258	0.628	0.628
Total arable land and permanent crops	r	0.69	-0.653	-0.705	0.084	-.912*	-.904*	-.956**	-.818*	-0.462	-.957**	0.584	-0.584
	p	0.129	0.16	0.117	0.874	0.011	0.013	0.003	0.047	0.356	0.003	0.301	0.301
Total agro-forestry area	r	.885*	-0.476	-0.769	-0.234	-0.573	-0.672	-0.514	-0.352	-0.478	-0.361	0.836	-0.836
	p	0.019	0.34	0.074	0.656	0.234	0.144	0.297	0.494	0.338	0.482	0.078	0.078
Total area of permanent pastures	r	-.916*	0.675	.906*	0.055	.951**	1.000**	.936**	0.703	0.513	.852*	-0.771	0.771
	p	0.01	0.141	0.013	0.918	0.004	0	0.006	0.119	0.298	0.031	0.127	0.127
Total forest area	r	-0.63	0.658	0.543	-0.135	0.765	0.738	.889*	.997**	0.561	.950**	-.906*	.906*
	p	0.18	0.156	0.266	0.798	0.076	0.094	0.018	0	0.247	0.004	0.034	0.034
% of changing area	r	.880*	-0.824	-0.462	-0.605	-0.837	-0.712	-.884*	-0.845	-.892*	-.887*	.992**	-.992**
	p	0.049	0.086	0.434	0.28	0.077	0.177	0.046	0.071	0.042	0.045	0.001	0.001
% of maintenance area	r	-.880*	0.824	0.462	0.605	0.837	0.712	.884*	0.845	.892*	.887*	-.992**	.992**
	p	0.049	0.086	0.434	0.28	0.077	0.177	0.046	0.071	0.042	0.045	0.001	0.001
% of artificial surfaces	r	-.859*	.877*	0.66	0.142	.977**	.898*	.979**	0.808	0.796	.926**	-.942*	.942*
	p	0.028	0.022	0.154	0.789	0.001	0.015	0.001	0.052	0.058	0.008	0.017	0.017
% of irrigated land	r	-0.348	0.465	0.183	-0.08	0.446	0.388	0.624	.910*	0.486	0.726	-0.315	0.315
	p	0.499	0.353	0.728	0.88	0.376	0.447	0.186	0.012	0.328	0.102	0.606	0.606
% of non-irrigated land	r	0.71	-0.731	-0.665	0.084	-.923**	-.887*	-.977**	-.884*	-0.567	-.990**	0.78	-0.78
	p	0.114	0.099	0.15	0.875	0.009	0.019	0.001	0.019	0.241	0	0.119	0.119
% of olive groves	r	-0.687	.940**	0.342	0.137	0.806	0.629	.811*	0.745	.948**	0.798	-0.846	0.846
	p	0.131	0.005	0.507	0.795	0.053	0.181	0.05	0.089	0.004	0.057	0.071	0.071
% of orchards	r	-0.526	0.642	0.658	-0.61	0.705	0.708	0.746	0.772	0.355	.852*	-0.235	0.235
	p	0.283	0.169	0.156	0.198	0.118	0.115	0.089	0.072	0.49	0.031	0.703	0.703
% of agro-forestry areas with >50% of tree cover	r	1	-0.738	-.859*	-0.216	-.888*	-.916*	-.831*	-0.582	-0.67	-0.695	.927*	-.927*
	p		0.094	0.028	0.681	0.018	0.01	0.041	0.226	0.146	0.125	0.023	0.023
% of agro-forestry areas with 30-50% of tree cover	r	-0.738	1	0.455	0.01	.844*	0.675	0.779	0.604	.932**	0.748	-0.827	0.827
	p	0.094		0.365	0.984	0.035	0.141	0.068	0.205	0.007	0.087	0.084	0.084
% of agro-forestry areas with 30-50% of tree cover	r	-.859*	0.455	1	-0.129	0.754	.906*	0.721	0.52	0.246	0.637	-0.56	0.56
	p	0.028	0.365		0.807	0.083	0.013	0.106	0.29	0.639	0.174	0.326	0.326
% of agro-forestry areas with <30% of tree cover and	r	-0.216	0.01	-0.129	1	0.116	0.055	0.069	-0.175	0.295	-0.141	-0.533	0.533
	p	0.681	0.984	0.807		0.827	0.918	0.897	0.74	0.57	0.79	0.355	0.355
% of agro-forestry areas with <30% of tree cover and	r	-.888*	.844*	0.754	0.116	1	.951**	.969**	0.718	0.719	.894*	-0.864	0.864
	p	0.018	0.035	0.083	0.827		0.004	0.001	0.108	0.108	0.016	0.059	0.059
% of permanent pastures	r	-.916*	0.675	.906*	0.055	.951**	1	.936**	0.703	0.513	.852*	-0.771	0.771
	p	0.01	0.141	0.013	0.918	0.004		0.006	0.119	0.298	0.031	0.127	0.127
% of mixed forest	r	-.831*	0.779	0.721	0.069	.969**	.936**	1	.857*	0.659	.965**	-.898*	.898*
	p	0.041	0.068	0.106	0.897	0.001	0.006		0.029	0.154	0.002	0.039	0.039
% of production forest	r	-0.582	0.604	0.52	-0.175	0.718	0.703	.857*	1	0.503	.933**	-0.817	0.817
	p	0.226	0.205	0.29	0.74	0.108	0.119	0.029		0.31	0.007	0.092	0.092
% of shrubs and/or herbaceous vegetation	r	-0.67	.932**	0.246	0.295	0.719	0.513	0.659	0.503	1	0.591	-0.859	0.859
	p	0.146	0.007	0.639	0.57	0.108	0.298	0.154	0.31		0.216	0.062	0.062
% of water surfaces	r	-0.695	0.748	0.637	-0.141	.894*	.852*	.965**	.933**	0.591	1	-.900*	.900*
	p	0.125	0.087	0.174	0.79	0.016	0.031	0.002	0.007	0.216		0.037	0.037
% of changing area	r	.927*	-0.827	-0.56	-0.533	-0.864	-0.771	-.898*	-0.817	-0.859	-.900*	1	-1.000**
	p	0.023	0.084	0.326	0.355	0.059	0.127	0.039	0.092	0.062	0.037		0
% of maintenance area	r	-.927*	0.827	0.56	0.533	0.864	0.771	.898*	0.817	0.859	.900*	-1.000**	1
	p	0.023	0.084	0.326	0.355	0.059	0.127	0.039	0.092	0.062	0.037	0	
Number of patches	r	-.859*	.821*	0.722	0.075	.975**	.932**	.996**	.851*	0.712	.956**	-.941*	.941*
	p	0.028	0.045	0.105	0.888	0.001	0.007	0	0.032	0.113	0.003	0.017	0.017
Total edge	r	-.860*	.833*	0.731	0.031	.971**	.930**	.992**	.860*	0.714	.960**	-.952*	.952*
	p	0.028	0.04	0.099	0.954	0.001	0.007	0	0.028	0.111	0.002	0.012	0.012
Mean patch size	r	0.802	-0.767	-0.72	0.034	-.941**	-.917*	-.992**	-.908*	-0.63	-.985**	.939*	-.939*
	p	0.055	0.075	0.107	0.949	0.005	0.01	0	0.012	0.18	0	0.018	0.018
Shannon's diversity index	r	-.833*	0.797	0.651	0.04	.866*	.829*	.925**	.915*	0.741	.910*	-.982**	.982**
	p	0.04	0.058	0.162	0.94	0.026	0.042	0.008	0.011	0.092	0.012	0.003	0.003
Structural impact	r	.928**	-0.707	-0.718	-0.252	-0.704	-0.709	-0.596	-0.342	-0.7	-0.438	.949*	-.949*
	p	0.008	0.117	0.108	0.63	0.119	0.115	0.212	0.506	0.122	0.386	0.014	0.014
Ecosystem service mitigated impact	r	-0.122	0.568	-0.123	0.118	0.512	0.287	0.58	0.577	0.574	0.649	-0.102	0.102
	p	0.817	0.24	0.817	0.824	0.3	0.581	0.227	0.231	0.234	0.163	0.87	0.87
Actual ecosystem service provision	r	.938**	-0.745	-0.712	-0.26	-0.738	-0.729	-0.634	-0.38	-0.738	-0.48	.962**	-.962**
	p	0.006	0.089	0.112	0.619	0.094	0.1	0.176	0.457	0.094	0.335	0.009	0.009
Ecosystem service provision capacity	r	0.48	-0.042	-0.747	0.121	-0.318	-0.503	-0.178	0.16	0.168	-0.05	0.248	-0.248
	p	0.335	0.937	0.088	0.819	0.54	0.309	0.736	0.762	0.75	0.925	0.687	0.687
Variation in structural impact	r	0.702	-0.268	-0.679	-0.033	-0.316	-0.5	-0.357	-0.709	-0.265	-0.302	0.621	-0.621
	p	0.186	0.663	0.207	0.958	0.604	0.391	0.556	0.18	0.667	0.622	0.263	0.263
Variation in ecosystem service provision	r	0.728	-0.294	-0.689	-0.067	-0.35	-0.527	-0.392	-0.727	-0.294	-0.336	0.651	-0.651
	p	0.164	0.631	0.198	0.915	0.563	0.362	0.513	0.164	0.631	0.58	0.234	0.234
Rate of effective ecosystem service provision	r	.936*	-0.681	-0.603	-0.687	-0.861	-0.825	-.926*	-0.794	-0.747	-0.868	.966**	-.966**
	p	0.019	0.206	0.282	0.2	0.061	0.085	0.024	0.108	0.147	0.056	0.008	0.008
Variation in ecosystem service provision capacity	r	0.295	-0.001	-0.7	0.67	-0.091	-0.344	0.014	0.185	0.272	-0.004	0.02	-0.02
	p	0.63	0.999	0.188	0.216	0.884	0.571	0.982	0.766	0.659	0.995	0.975	0.975
Variation in ecosystem mitigated impact	r	0.431	-0.071	-0.53	0.286	-0.01	-0.225	-0.024	-0.515	-0.034	0	0.339	-0.339
	p	0.468	0.91	0.358	0.641	0.987	0.716	0.97	0.374	0.957	1	0.577	0.577

Columns 26–38:

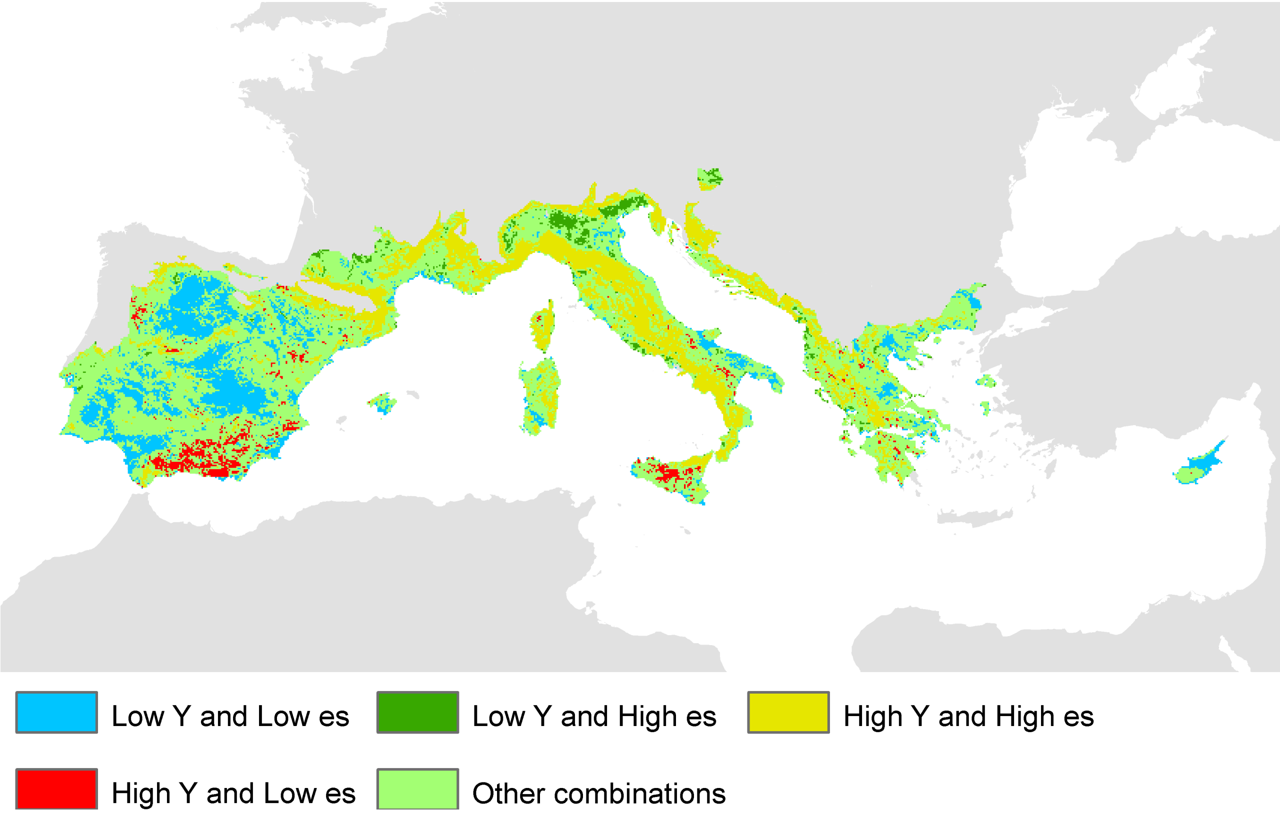
Correlations		Number of patches	Total edge	Mean patch size	Shannon's diversity index	Structural impact	Ecosystem service mitigated impact	Actual ecosystem service provision	Ecosystem service provision capacity	Variation in structural impact	Variation in ecosystem service provision	Rate of effective ecosystem service provision	Variation in ecosystem service provision capacity	Variation in ecosystem mitigated impact
Grazing intensity	r	0.924	0.924	-0.87	0.862	-.982*	.990**	-.983*	0.027	0.424	0.381	-0.985	0.631	0.58
	p	0.076	0.076	0.13	0.138	0.018	0.01	0.017	0.973	0.721	0.751	0.109	0.566	0.606
Variation in grazing intensity	r	-0.915	-0.895	0.936	-0.238	0.634	-0.742	0.643	0.498	-0.904	-0.882	0.651	0.008	-0.966
	p	0.264	0.294	0.229	0.847	0.563	0.468	0.555	0.668	0.282	0.312	0.549	0.995	0.167
Total arable land and permanent crops	r	-.930**	-.926**	.953**	-0.799	0.393	-0.596	0.432	0.238	0.036	0.068	0.646	0.093	-0.25
	p	0.007	0.008	0.003	0.056	0.441	0.212	0.392	0.65	0.954	0.913	0.239	0.881	0.685
Total agro-forestry area	r	-0.56	-0.568	0.5	-0.638	.940**	0.284	.925**	0.504	.917*	.931*	0.846	0.414	0.723
	p	0.248	0.24	0.313	0.172	0.005	0.585	0.008	0.309	0.028	0.022	0.071	0.488	0.167
Total area of permanent pastures	r	.932**	.930**	-.917*	.829*	-0.709	0.287	-0.729	-0.503	-0.5	-0.527	-0.825	-0.344	-0.225
	p	0.007	0.007	0.01	0.042	0.115	0.581	0.1	0.309	0.391	0.362	0.085	0.571	0.716
Total forest area	r	.886*	.894*	-.933**	.940**	-0.398	0.594	-0.437	0.136	-0.665	-0.689	-0.876	0.182	-0.431
	p	0.019	0.016	0.007	0.005	0.434	0.214	0.386	0.797	0.221	0.199	0.051	0.769	0.469
% of changing area	r	-.927*	-.932*	.917*	-.981**	.903*	-0.2	.920*	0.137	0.561	0.592	.963**	-0.104	0.269
	p	0.023	0.021	0.028	0.003	0.036	0.747	0.027	0.826	0.325	0.293	0.009	0.868	0.662
% of maintenance area	r	.927*	.932*	-.917*	.981**	-.903*	0.2	-.920*	-0.137	-0.561	-0.592	-.963**	0.104	-0.269
	p	0.023	0.021	0.028	0.003	0.036	0.747	0.027	0.826	0.325	0.293	0.009	0.868	0.662
% of artificial surfaces	r	.991**	.988**	-.962**	.935**	-0.682	0.603	-0.722	-0.126	-0.353	-0.389	-.923*	0.084	-0.035
	p	0	0	0.002	0.006	0.136	0.205	0.105	0.812	0.56	0.518	0.025	0.893	0.955
% of irrigated land	r	0.629	0.637	-0.688	0.796	-0.186	0.575	-0.223	0.512	-0.34	-0.343	-0.279	0.501	-0.304
	p	0.181	0.174	0.131	0.058	0.724	0.232	0.671	0.299	0.576	0.572	0.65	0.39	0.619
% of non-irrigated land	r	-.962**	-.961**	.983**	-.874*	0.436	-0.649	0.478	0.126	0.15	0.186	0.796	-0.008	-0.164
	p	0.002	0.002	0	0.023	0.387	0.163	0.337	0.812	0.81	0.765	0.107	0.99	0.792
% of olive groves	r	.847*	.852*	-0.808	.881*	-0.616	0.695	-0.662	0.215	-0.26	-0.289	-0.734	0.301	-0.034
	p	0.033	0.031	0.052	0.02	0.193	0.126	0.152	0.683	0.672	0.637	0.158	0.622	0.956
% of orchards	r	0.742	0.769	-0.804	0.7	-0.328	0.384	-0.354	-0.241	-0.273	-0.268	-0.063	-0.785	-0.335
	p	0.092	0.074	0.054	0.122	0.525	0.452	0.492	0.645	0.657	0.663	0.919	0.116	0.582
% of agro-forestry areas with >50% of tree cover	r	-.859*	-.860*	0.802	-.833*	.928**	-0.122	.938**	0.48	0.702	0.728	.936*	0.295	0.431
	p	0.028	0.028	0.055	0.04	0.008	0.817	0.006	0.335	0.186	0.164	0.019	0.63	0.468
% of agro-forestry areas with 30-50% of tree cover	r	.821*	.833*	-0.767	0.797	-0.707	0.568	-0.745	-0.042	-0.268	-0.294	-0.681	-0.001	-0.071
	p	0.045	0.04	0.075	0.058	0.117	0.24	0.089	0.937	0.663	0.631	0.206	0.999	0.91
% of agro-forestry areas with 30-50% of tree cover	r	0.722	0.731	-0.72	0.651	-0.718	-0.123	-0.712	-0.747	-0.679	-0.689	-0.603	-0.7	-0.53
	p	0.105	0.099	0.107	0.162	0.108	0.817	0.112	0.088	0.207	0.198	0.282	0.188	0.358
% of agro-forestry areas with <30% of tree cover and	r	0.075	0.031	0.034	0.04	-0.252	0.118	-0.26	0.121	-0.033	-0.067	-0.687	0.67	0.286
	p	0.888	0.954	0.949	0.94	0.63	0.824	0.619	0.819	0.958	0.915	0.2	0.216	0.641
% of agro-forestry areas with <30% of tree cover and	r	.975**	.971**	-.941**	.866*	-0.704	0.512	-0.738	-0.318	-0.316	-0.35	-0.861	-0.091	-0.01
	p	0.001	0.001	0.005	0.026	0.119	0.3	0.094	0.54	0.604	0.563	0.061	0.884	0.987
% of permanent pastures	r	.932**	.930**	-.917*	.829*	-0.709	0.287	-0.729	-0.503	-0.5	-0.527	-0.825	-0.344	-0.225
	p	0.007	0.007	0.01	0.042	0.115	0.581	0.1	0.309	0.391	0.362	0.085	0.571	0.716
% of mixed forest	r	.996**	.992**	-.992**	.925**	-0.596	0.58	-0.634	-0.178	-0.357	-0.392	-.926*	0.014	-0.024
	p	0	0	0	0.008	0.212	0.227	0.176	0.736	0.556	0.513	0.024	0.982	0.97
% of production forest	r	.851*	.860*	-.908*	.915*	-0.342	0.577	-0.38	0.16	-0.709	-0.727	-0.794	0.185	-0.515
	p	0.032	0.028	0.012	0.011	0.506	0.231	0.457	0.762	0.18	0.164	0.108	0.766	0.374
% of shrubs and/or herbaceous vegetation	r	0.712	0.714	-0.63	0.741	-0.7	0.574	-0.738	0.168	-0.265	-0.294	-0.747	0.272	-0.034
	p	0.113	0.111	0.18	0.092	0.122	0.234	0.094	0.75	0.667	0.631	0.147	0.659	0.957
% of water surfaces	r	.956**	.960**	-.985**	.910*	-0.438	0.649	-0.48	-0.05	-0.302	-0.336	-0.868	-0.004	0
	p	0.003	0.002	0	0.012	0.386	0.163	0.335	0.925	0.622	0.58	0.056	0.995	1
% of changing area	r	-.941*	-.952*	.939*	-.982**	.949*	-0.102	.962**	0.248	0.621	0.651	.966**	0.02	0.339
	p	0.017	0.012	0.018	0.003	0.014	0.87	0.009	0.687	0.263	0.234	0.008	0.975	0.577
% of maintenance area	r	.941*	.952*	-.939*	.982**	-.949*	0.102	-.962**	-0.248	-0.621	-0.651	-.966**	-0.02	-0.339
	p	0.017	0.012	0.018	0.003	0.014	0.87	0.009	0.687	0.263	0.234	0.008	0.975	0.577
Number of patches	r	1	.999**	-.988**	.944**	-0.651	0.563	-0.689	-0.173	-0.421	-0.456	-.943*	-0.01	-0.1
	p		0	0	0.005	0.161	0.245	0.13	0.743	0.48	0.44	0.016	0.987	0.872
Total edge	r	.999**	1	-.990**	.951**	-0.659	0.549	-0.696	-0.178	-0.461	-0.494	-.937*	-0.067	-0.154
	p	0		0	0.004	0.155	0.259	0.125	0.735	0.434	0.397	0.019	0.915	0.804
Mean patch size	r	-.988**	-.990**	1	-.943**	0.564	-0.569	0.602	0.143	0.462	0.496	.946*	0.068	0.146
	p	0	0		0.005	0.244	0.238	0.206	0.787	0.433	0.395	0.015	0.914	0.815
Shannon's diversity index	r	.944**	.951**	-.943**	1	-0.68	0.486	-0.713	-0.009	-0.703	-0.729	-.946*	0.004	-0.446
	p	0.005	0.004	0.005		0.137	0.329	0.112	0.987	0.185	0.162	0.015	0.995	0.452
Structural impact	r	-0.651	-0.659	0.564	-0.68	1	0.08	.998**	0.465	0.777	0.799	.906*	0.33	0.547
	p	0.161	0.155	0.244	0.137		0.88	0	0.353	0.122	0.105	0.034	0.588	0.34
Ecosystem service mitigated impact	r	0.563	0.549	-0.569	0.486	0.08	1	0.017	0.532	0.662	0.638	-0.067	0.815	0.794
	p	0.245	0.259	0.238	0.329	0.88		0.974	0.278	0.223	0.247	0.915	0.093	0.108
Actual ecosystem service provision	r	-0.689	-0.696	0.602	-0.713	.998**	0.017	1	0.433	0.749	0.773	.917*	0.291	0.511
	p	0.13	0.125	0.206	0.112	0	0.974		0.391	0.145	0.126	0.028	0.635	0.379
Ecosystem service provision capacity	r	-0.173	-0.178	0.143	-0.009	0.465	0.532	0.433	1	0.393	0.396	0.293	0.784	0.319
	p	0.743	0.735	0.787	0.987	0.353	0.278	0.391		0.513	0.51	0.632	0.117	0.601
Variation in structural impact	r	-0.421	-0.461	0.462	-0.703	0.777	0.662	0.749	0.393	1	.999**	0.599	0.528	.934*
	p	0.48	0.434	0.433	0.185	0.122	0.223	0.145	0.513		0	0.285	0.36	0.02
Variation in ecosystem service provision	r	-0.456	-0.494	0.496	-0.729	0.799	0.638	0.773	0.396	.999**	1	0.631	0.509	.919*
	p	0.44	0.397	0.395	0.162	0.105	0.247	0.126	0.51	0		0.253	0.381	0.027
Rate of effective ecosystem service provision	r	-.943*	-.937*	.946*	-.946*	.906*	-0.067	.917*	0.293	0.599	0.631	1	-0.049	0.28
	p	0.016	0.019	0.015	0.015	0.034	0.915	0.028	0.632	0.285	0.253		0.937	0.648
Variation in ecosystem service provision capacity	r	-0.01	-0.067	0.068	0.004	0.33	0.815	0.291	0.784	0.528	0.509	-0.049	1	0.654
	p	0.987	0.915	0.914	0.995	0.588	0.093	0.635	0.117	0.36	0.381	0.937		0.231
Variation in ecosystem mitigated impact	r	-0.1	-0.154	0.146	-0.446	0.547	0.794	0.511	0.319	.934*	.919*	0.28	0.654	1
	p	0.872	0.804	0.815	0.452	0.34	0.108	0.379	0.601	0.02	0.027	0.648	0.231	

Correlations		Variation in grazing intensity	% of changing area	% of maintenance area	% of changing area	% of maintenance area	Variation in structural impact	Variation in ecosystem service provision	Rate of effective ecosystem service provision	Variation in ecosystem service provision capacity	Variation in ecosystem mitigated impact
Variation in grazing intensity	r	1	0.487	-0.487	0.552	-0.552	-0.904	-0.882	0.651	0.008	-0.966
	p		0.676	0.676	0.628	0.628	0.282	0.312	0.549	0.995	0.167
% of changing area	r	0.487	1	-1.000**	.992**	-.992**	0.561	0.592	.963**	-0.104	0.269
	p	0.676		0	0.001	0.001	0.325	0.293	0.009	0.868	0.662
% of maintenance area	r	-0.487	-1.000**	1	-.992**	.992**	-0.561	-0.592	-.963**	0.104	-0.269
	p	0.676	0		0.001	0.001	0.325	0.293	0.009	0.868	0.662
% of changing area	r	0.552	.992**	-.992**	1	-1.000**	0.621	0.651	.966**	0.02	0.339
	p	0.628	0.001	0.001		0	0.263	0.234	0.008	0.975	0.577
% of maintenance area	r	-0.552	-.992**	.992**	-1.000**	1	-0.621	-0.651	-.966**	-0.02	-0.339
	p	0.628	0.001	0.001	0		0.263	0.234	0.008	0.975	0.577
Variation in structural impact	r	-0.904	0.561	-0.561	0.621	-0.621	1	.999**	0.599	0.528	.934*
	p	0.282	0.325	0.325	0.263	0.263		0	0.285	0.36	0.02
Variation in ecosystem service provision	r	-0.882	0.592	-0.592	0.651	-0.651	.999**	1	0.631	0.509	.919*
	p	0.312	0.293	0.293	0.234	0.234	0		0.253	0.381	0.027
Rate of effective ecosystem service provision	r	0.651	.963**	-.963**	.966**	-.966**	0.599	0.631	1	-0.049	0.28
	p	0.549	0.009	0.009	0.008	0.008	0.285	0.253		0.937	0.648
Variation in ecosystem service provision capacity	r	0.008	-0.104	0.104	0.02	-0.02	0.528	0.509	-0.049	1	0.654
	p	0.995	0.868	0.868	0.975	0.975	0.36	0.381	0.937		0.231
Variation in ecosystem mitigated impact	r	-0.966	0.269	-0.269	0.339	-0.339	.934*	.919*	0.28	0.654	1
	p	0.167	0.662	0.662	0.577	0.577	0.02	0.027	0.648	0.231	

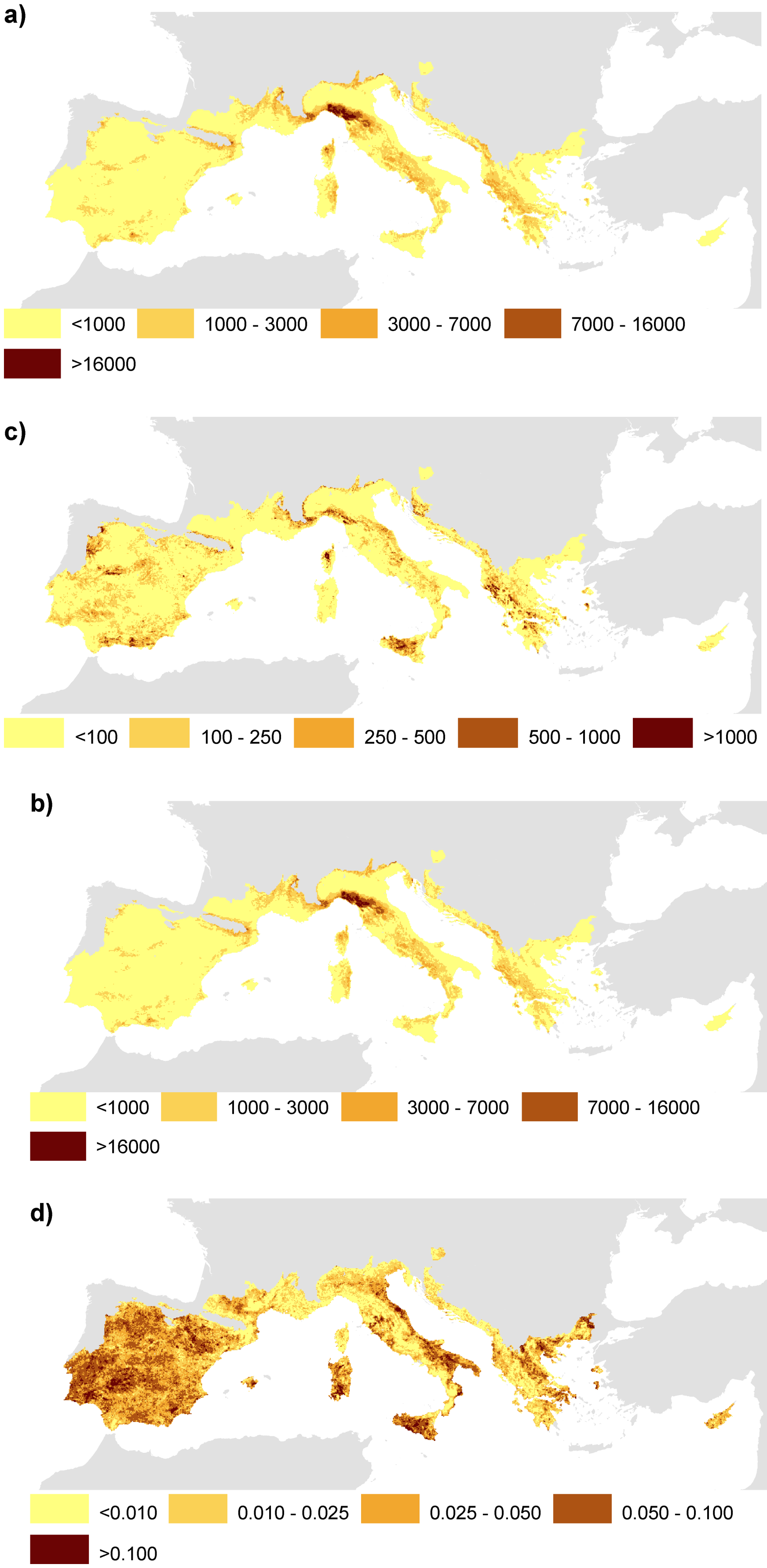
* Correlation is significant at p<0.05 (2-tailed).

** Correlation is significant at p<0.01 (2-tailed).

Appendix 2 – Spatial distribution of the co-occurrence of High/Low values of both the structural impact and the capacity of ES provision (values represent the 2013 time slice).



Appendix 2 – Spatial distribution of the IQ range for: a) the structural impact (in tons of soil loss per pixel); b) the actual ES provision (in tons of soil loss per pixel); c) the ES mitigated impact (in tons of soil loss per pixel); and d) the capacity for ES provision (adimentional).



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