Ecosystem Services 1 (2012) 31–39

Ecosystem Services

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

Mapping ecosystem services for policy support and decision making in the European Union

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ARTICLE INFO

Article history:
Received 29 March 2012
Received in revised form
27 June 2012
Accepted 29 June 2012
Available online 31 July 2012

Keywords:
Biodiversity policy
Mapping ecosystem services
Water purification
European Union

ABSTRACT

Mainstreaming ecosystem services into policy and decision making is dependent on the availability of spatially explicit information on the state and trends of ecosystems and their services. In particular, the EU Biodiversity Strategy to 2020 addresses the need to account for ecosystem services through biophysical mapping and valuation. This paper reviews current mapping methods, identifies current knowledge gaps and provides the elements for a methodological framework for mapping and assessing ecosystems and their services at European scale. Current mapping methodologies go beyond purely land cover based assessments and include the use of primary data of ecosystem services, the use of functional traits to map ecosystem services and the development of models and ecological production functions. Additional research is needed to cover marine ecosystems and to include the resilience of ecosystems to environmental change in spatially explicit assessments. The ecosystem services cascade which connects ecosystems to human wellbeing is argued to provide a suitable, stepwise framework for mapping ecosystem services in order to support EU policies in a more effective way. We demonstrate the use of this framework for mapping using the water purification service as case.

1. Introduction

In 2010, the tenth meeting of the Conference of Parties (COP 10) to the Convention on Biological Diversity (CBD) led to the adoption of a global Strategic Plan for biodiversity for the period 2011–2020. The Strategic Plan includes, besides strategic goals, also 20 targets, known as the Aichi Targets. The Aichi biodiversity targets complement the previous, conservation-based biodiversity targets with the addition of ecosystem services (ES) as an element to be considered in the global expansion of protected areas (Target 11), as well as a component of priority for protection and restoration (Target 14). ES, the benefits that people obtain from ecosystems (Millennium Ecosystem Assessment, 2005), could be instrumental in making a case for biodiversity if such benefits are made explicit. Countries that are signatory to the CBD are bound by the commitment to change their biodiversity strategy in order to accomplish these targets. The European Union (EU), a party to the Convention, has laid down this global commitment in a Biodiversity Strategy to 2020 which integrates ES as underpinning elements of Member States' economy to complement the conservation approach to biodiversity (European Commission, 2011a).

It becomes more and more evident to 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 technical infrastructures (Daily and Matson, 2008; Ervin et al., 2012). Therefore, other EU policies are now integrating the ES approach into their planning. For instance, the ecosystem service concept has been identified as one of the pillars of the assessment of impacts in the preparation of the 2012 Commission's Blueprint to safeguard the future of European Waters by 2015. Furthermore, restoring and preserving ES is one of the six priorities identified by the rural development pillar in the new proposal for the EU's Common Agriculture Policy (European Commission 2011b). Importantly, the EU's regional and cohesion policy now recognizes the importance of investing in nature as a source of economic development (European Commission 2011c).

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2212-0416
http://dx.doi.org/10.1016/j.ecoser.2012.06.004

The design, implementation and management of policies to deliver plans incorporating biodiversity conservation and the multiple services provided by ecosystems is dependent on the availability of spatially explicit information describing ES (Cowling et al., 2008). These policy decisions need to be based on reliable estimates of current and expected trends in ecosystem service supply and their economic values, taking into account the spatial distribution of resources providing the ES. The EU Biodiversity Strategy makes this policy request for accurate spatial assessments of ecosystem services explicit. Target 2 aims to maintain and enhance ecosystems and their services by 2020, by establishing a green infrastructure and restoring at least 15% of degraded ecosystems (European Commission, 2011a). In its supporting Action 5, which aims to improve knowledge on ecosystems and their services, the European Commission is committed to assist EU Member States to map and assess the state of ecosystems and their services in their national territory by 2014, to assess the economic value of such services, and to promote the integration of these values into accounting and reporting systems at EU and national level by 2020 (European Commission, 2011a). Action 5 is driven by policy questions and needs, which are spelled out in Table 1. This list of questions was formulated by EU biodiversity policy makers (European Commission—Directorate-General Environment, personal communication) as a first basis for discussions with EU Member States' experts on how to implement Action 5. To the list, we added concrete policy and research actions that are needed to address these questions. Essential actions are to increase awareness of ES among key stakeholder groups, to develop a clear typology of ES so that they can be used for setting management objectives, and to develop an analytical framework for mapping and assessing ES that serve the multiple objectives addressed by policies. Above all, Table 1 clearly demonstrates that place based information on ecosystem services plays a crucial role to address many of the outstanding policy questions, for example when deciding where to restore ecosystems and where and how much to invest in green infrastructure so that multiple services are delivered. These decisions depend on the availability of spatially explicit information describing ecosystems and the flow of their services. Furthermore, quantitative spatial information on the delivery of and the demand for ecosystem services provides baseline data to measure net future gains or losses for policy impact assessment and can support the development of financial instruments to finance investments in ecosystems.

The overall objective of this paper is to summarize current practices and methods for mapping ES which can be applied to deliver on the request of biodiversity and sectorial EU policies. Secondly we identify current gaps that prevent us to move from mapping ecosystem services to providing the tools for planning and decision making. Finally, we adopt a commonly used conceptual model for framing ecosystem services (the ecosystem service cascade model, Haines-Young and Potschin, 2010a) to show how it can be used for mapping and assessing ES at European scale. The use of this framework is then demonstrated making the case of water purification services based on a pan-European study.

2. Current practices in mapping ecosystem services

Research on mapping ES has grown substantially in the past decade (Nelson and Daily, 2010; Seppelt et al., 2011). In particular, new initiatives focusing on modeling ES are being established, such as the Natural Capital Project and the Ecosystem Services Partnership. Recently, the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) was launched to

Table 1
Linking EU policy questions to policy and research actions (European Commission—DG Environment, personal communication).

Policy questions	Policy and research actions
What is the current public understanding of ecosystem services and the benefits they provide? Why should we incorporate the economic values of ecosystem services into decision making?	Raising awareness
How have we advanced our understanding links between ecosystems, ecosystem functions and ecosystem services? More broadly, what is the influence of ecosystem services on long-term human well-being and what are the knowledge constraints on more informed decision making? Which vital provisioning services are produced outside the EU?	Setting and using an analytical framework for ecosystem assessment; Promoting consistency in the typology of ecosystems and ecosystem services
What are the status and trends of the EU's ecosystems and the services they provide to society? What are the drivers causing changes in the EU's ecosystems and their services?	Biophysical mapping of ecosystem services using data and models
What are the economic implications of different plausible futures? How do ecosystem services affect human well-being, who and where are the beneficiaries, and how does this affect how they are valued and managed?	Monetary and non-monetary valuation of ecosystem services
How might ecosystems and their services change in the EU under plausible future scenarios—including the development of scenarios and options for implementing the 15% restoration target? What would be needed in terms of review of financing instruments? How can we secure and improve the continued delivery of ecosystem services? Can we set priorities for ecosystem restoration within a strategic framework at sub-national, national and EU level? Can we design prioritization criteria for restoration and at which scale to get significant benefits in a cost-effective way? Can we define where to strategically deploy green infrastructure in the EU in urban and rural areas to improve ecosystem resilience and habitat connectivity and to enhance the delivery of ecosystem services at Member State and sub-national level? How to foster synergies between existing and planned initiatives at local, regional or national levels in Member States, as well as how to promote further investments, thereby providing added value to Member States action?	Mapping and valuation of ecosystem services as part of an integrated and stakeholder based approach to sustainable land management and use of natural resources

guide the flow of scientific information related to biodiversity and ES to governments and practitioners. The increased research interest is partly due to the inclusion of ES in conservation policies as well as policies that address the use of natural resources as mentioned above, but also because of interest from the business sector. Companies asses their opportunities and risks related to ecosystem functioning and need information on how to comply with current policies or to understand how they could be affected by potential new policies (Hanson et al., 2012). To support this increased interests in the use and regulation of ES mapping and modeling techniques play an ever important role.

2.1. Reasons for mapping ecosystem services

Information based on mapping and modeling exercises has been used to analyze the spatial distribution of multiple ES at local (Naidoo and Ricketts, 2006; Nelson et al., 2008; Lautenbach et al., 2011; Lavorel et al., 2011), regional (Chan et al., 2006; Metzger et al., 2006) and global (Naidoo et al., 2008; Luck et al., 2009) spatial scales. The rationale for mapping ES varies strongly among studies and includes: evaluation of spatial congruence with biodiversity (Chan et al., 2006; Egoh et al., 2009; Bai et al., 2011), analyzing synergies and trade-offs between different ES (Raudsepp-Hearne et al., 2010; Chisholm, 2010), analyzing trends in ES (Li and Ren, 2008; Harrison et al., 2010), estimating costs and benefits (Coiner et al., 2001; Naidoo and Adamowicz, 2006; Termansen et al., 2008; Nelson et al., 2009), comparing ES supply with demand (Burkhard et al., 2012a; Nedkov and Burkhard, 2011, Willemen et al., 2012), monetary valuation on biophysical quantities (Deng et al., 2011; O'Farrell et al., 2011; Gascoigne et al., 2011; La Notte et al., 2012) or the prioritization of areas in spatial planning and management (Chan et al., 2006, Egoh et al.,2011) . Typically, planning studies, like cost-benefit, prioritization and trade-off analyses are carried out on sub-national levels, while studies focusing on general trends like ES spatial distribution and congruence are carried out on continental or global level.

2.2. Mapping approaches

Several approaches to map ES exist and reviews of methodologies are available (Burkhard et al., 2009; Eigenbrod et al., 2010). A simple approach is to derive information on ES directly from land-use/cover or habitat maps (Burkhard et al., 2009; Kienast et al., 2009; Vihervaara et al., 2010; Haines-Young et al., 2012). Such approaches may be appropriate at large scales, for areas where the dominant service relates directly to land use (e.g., crop and timber production) or where data availability or expertise is limited, and where the focus is on the assumed presence of ES rather than on quantification of the supply.

Primary data to map ES are used for provisioning services where statistics are available (e.g., FAOSTAT or national statistics). Examples include timber, food, or water supply (Van Jaarsveld et al., 2005). Although primary data offer clearly the most accurate information, such information is not readily available for other ecosystem services and the collection of such information is often resource intensive. Thus, while provisioning ES can often be directly quantified, most regulating, supporting, and cultural services are less straightforward to be put on maps and researchers must rely on proxies for their quantification. Such proxies can be derived from model outputs, for instance the use of modeled runoff as indicator for water provision or the quantity of air pollutants captured by leaves as indicator for air purification services (Feld et al., 2009; Layke et al., 2011).

Another considerable body of literature assesses ES by spatially explicit maps of ecosystem service values (Costanza et al., 1997; Troy and Wilson, 2006; Bateman, 2009; Brainard et al., 2009). Typically, ecosystem service values are transferred from existing primary valuation studies to the study site area (Brander and Koetse, 2011). Value transfer methodologies differ in their way of adjusting transferred values to site specific circumstances. Whereas some studies attribute uniform values to ES supply indicators (Kreuter et al., 2001; Isely et al., 2010; Troy and Wilson, 2006), others use value functions including a number of spatial variables (Costanza et al., 2008; Termansen et al., 2008; Brander and Koetse, 2011). The emergence of advanced GIS technology and the availability of more qualitative spatial socioeconomic and biophysical datasets have enhanced the potential and application of spatially explicit valuation exercises (Bateman et al., 2002). The quality of such value maps depends not only on how the values of ES incorporate spatial variations but also on the underlying biophysical maps of ecosystem service supply (Plummer, 2009; Eigenbrod et al., 2010).

Because ecosystem service supply is based on functions and characteristics of the biodiversity (including genes, species, and habitats), recent mapping techniques are based on biological data such as functional traits of plants (Lavorel et al., 2011; Lavorel and Grigulis, 2012) or ecosystem structure and habitat data (Raffaelli, 2006). Functional traits, such as vegetative height, leaf dry matter content, leaf nitrogen and phosphorous concentration, flowering onset, can be used to model trade-offs between ES (Lavorel et al., 2011). Habitat classification, such as the European Nature Information System (EUNIS) classification (Davies et al., 2004), include detailed data on the associated biodiversity, which makes their use reasonable in mapping relationships between biodiversity and ES. Remote sensing applications of developing automated mapping techniques have a great potential as have been shown already in some case studies (e.g., Dubois et al., 2011).

Finally, more integrated mapping approaches are based on the application of dynamic process-based ecosystem models (Morales et al., 2005; Schröter et al., 2005) or models which estimate ecological production functions such as the ones developed by the Natural Capital Project available in the InVEST tool (Nelson et al., 2009; Kareiva et al., 2011). Such mapping approaches take account of the underlying mechanisms which drive ecosystem service delivery and are therefore more likely to produce realistic changes in ecosystem service supply at the local and landscape scales, but they require significant investment in terms of data acquisition and expert knowledge.

3. Challenges in mapping ecosystem services

To answer the EU policy questions raised in Table 1 and to achieve target 2 (Action 5) of the EU biodiversity strategy which deals with mapping and valuing of ecosystem services, several challenges need to be addressed. These explicitly include setting a consistent typology of ecosystem services for which ES maps can be developed and understand how pressures on ecosystems affect the flow of services. In the process to tackle these challenges the development of standardised models and indicators for ES mapping is of primary importance. Here we outline three main research needs to support the process of ES mapping.

3.1. Filling the data gaps

More primary data are needed to map directly the stocks and actual flows of ES as well as to validate the current ES models. At present, the use of proxy data is common in the mapping of ES (Chan et al., 2006, Egoh et al., 2008; Naidoo et al., 2008; Eigenbrod et al., 2010). Quite often such proxy data were not generated in the context of ES but are now being cleverly re-used to map ES due to the lack of baseline information. For instance, species–area relationships (SAR) are a biogeographical proxy used in scientific literature to estimate a mechanism for species addition (Eigenbrod et al., 2010), while Nelson et al. (2009) used SAR scores to quantify a supporting ES of a freshwater ecosystems' habitat suitability for vertebrate species. In fact, several ES are hardly addressed in the scientific literature due to the lack of sufficient baseline information (Feld et al., 2009; Harrison et al., 2010; Layke et al., 2011). Key gaps in knowledge are especially evident for the provision of genetic and medicinal resources, the supporting services of life cycle and gene pool maintenance for the regulation services of disease and pest control and seed dispersal and for all cultural services but recreation (Harrison et al., 2010). Ecological knowledge of species interactions and ecosystem structure, such as trophic levels or key stone species, are seldom translated to ES assessments. To avoid risks of creating a policy bias by focusing on a subset of indicators which are high on the political agenda such as food, water provision and climate regulation services, indicators of these not-yet-quantified ES must be developed and their benefits need to be assessed.

Regarding ecosystem types for which ES are assessed and mapped, terrestrial ecosystems are those mainly capturing researchers' attention. In contrast, marine ecosystem services are largely overlooked and increasing efforts are needed to map the contributions of marine systems to the provision of ES. In particular, in the marine realm, several conceptual frameworks have been proposed (Costanza, 1999; Beaumont et al., 2007; Foley et al., 2010) but there are too few available examples of ES mapping and they are mostly focused on protected areas at local scales (Mumby et al., 2008, Roncin et al., 2008, Mangi et al., 2011, Stoeckl et al., 2011). Due to the absence of spatially explicit information on ecosystem service supply in most cases only coarse estimates or statistics at the state level are developed (Lange and Jiddawi, 2009; Brenner et al., 2010; Austen et al., 2011) sometimes with uncertainty in the location of sources and benefits. The main challenges with mapping marine ES are (i) the lack of coverage and resolution in the available data (e.g., habitat mapping) as well as scarce georeferenced data and ambiguous maritime boundaries, (ii) the 4D structure formed by benthic and pelagic habitats which is highly dynamic across time, (iii) a poorer knowledge on ecosystem functions and processes quantification relative to the better explored terrestrial ecosystems. In particular, there is no comprehensive habitat layer in the marine ecosystems equivalent to the land cover information in the terrestrial environment, while the connectivity amongst habitats is more difficult to assess (Somerfield et al., 2008). Further challenges for valuing marine ES are summarized by Barbier (2012).

3.2. Consistency in mapping approaches

As discussed above, different approaches are used to map ES. In many cases a different set of indicators is used to map a single service, resulting in different units in which ES are expressed. For example, different proxies are often used to map air quality regulation including fluxes in atmospheric gasses, atmospheric cleansing capacity or levels of pollutants in the air (Layke et al., 2011). Often based on the definition of the service and the objective of the study, some consider as ES only those derived from natural systems (Jansson et al., 1998) while others include natural or human-transformed systems (Metzger et al., 2006, Reyers et al., 2009). Discrepancies are sometimes compounded by the fact that some studies map stocks (Kalacska et al., 2008) while others map flows of ES (van Jaarsveld et al., 2005; Naidoo et al., 2011). Depending on the objective and set of indicators used, the same ES mapped could produce different results in the same study area (Lamarque et al., 2011). These discrepancies evidently have implications for estimating monetary values. Thus, the need to adopt a more rigid, methodological framework as well as the need to standardize definitions for each service and methods for mapping them are both essential for comparing results among different EU Members States and measuring effectiveness of different policy measures. Consistency in mapping approaches is therefore a major challenge. This requires a more detailed classification and definition of ES such as the CICES proposal for ecosystem accounting (Common International Classification of Ecosystem Goods and Services, Haines-Young and Potschin, 2010b).

3.3. Incorporating ecosystem status in ecosystem service maps

In the EU, legislation to protect the environment focuses on improving the status of ecosystems. In particular, the EU aims to bring habitats and threatened species into favorable conservation status, freshwater and coastal ecosystems into good ecological status and marine ecosystems into good environmental status. The concept of ecosystem services is appealing to help the implementation of environmental legislation. Mainstreaming ecosystem services in EU policies that focus on the protection of terrestrial, freshwater or marine ecosystems assumes that there is a connection between ecosystem status and the services they deliver. However, this connection is until now poorly explored across Europe and needs to be demonstrated yet, also considering that the relationships between ecosystem functioning, ecosystem status, biodiversity and ecosystem services are issue of scientific debate. Changes in ecosystems and their services are often non-linear and can often be accelerating, abrupt and potentially irreversible (MA, 2005). The loss of biodiversity and increasing pressures from drivers causing ecosystem change increase the likelihood of these non-linear changes. There are a few studies that have dealt with scenarios of change in the ecosystem service provision by constantly changing ecosystems (Posthumus et al., 2010; Lautenbach et al., 2011). Although science is increasingly able to predict some of these risks and non-linearities, predicting the thresholds at which these changes will happen is generally not possible. The GBO3 (Secretariat of the Convention on Biological Diversity, 2010) documents clearly a great number of such cases.

At present, some mapping approaches select indicators that do not take into consideration the multi-dimensional spatial and temporal aspects and the sustainable thresholds to keep the ecosystem functioning or the negative effects on biodiversity. As mentioned above, an example is the re-use of data (e.g., agricultural statistics/census; Raudsepp-Hearne et al., 2010) which were not collected for the purpose of ecosystem services and has no sustainability criteria in them as ecosystem service proxies. Other examples include the use of unsustainable grazing, timber and water extraction data as proxies for fodder provision, timber production and water provision respectively; or the use of nitrogen concentration as a proxy for water purification that does not take into account the sustainable potential of the ecosystem.

4. A frame for mapping and modeling ecosystem services

Despite several important research gaps outlined in this paper, the multitude of mapping methods which are reported here as well as the availability of numerous indicators for ES (Layke et al., 2011) illustrates the rapid build of an ES knowledge base. The next step is to operationalize this knowledge and make it applicable to address the EU policy questions (Table 1) and assess scenarios that involve terrestrial or maritime spatial planning and the use of natural resources. This requires a modeling approach to synthesize and quantify our understanding of ES and to understand dynamic, spatially explicit trade-offs as part of the larger socio-ecological systems (Salzau message, Burkhard et al., 2012b). Integrated natural and economic research is needed to assess the spatial and temporal flow of ES relevant to human well-being; to demonstrate the role of biodiversity and ecosystem health in underpinning ES; to add sustainability criteria avoiding overexploitation of ecosystems; and to operationalize these concepts within key regulatory frameworks.

4.1. The ecosystem services cascade framework

Here we argue that the ES cascade (De Groot et al., 2010; Haines-Young and Potschin, 2010a; TEEB, 2010) is a useful concept to frame spatially explicit, quantitative assessments of ecosystems, ES and benefits. This framework links biodiversity and ecosystems stepwise to human wellbeing through the flow of ES. Ecosystems provide the necessary structure and processes that underpin ecosystem functions which are defined as the capacity or potential to deliver services. ES are derived from ecosystem functions and represent the realized flow of services in relation to the benefits and values of people. As mentioned above, EU member states are to map and value ecosystem services in their territories by 2014. This framework is important in understanding and standardising the outputs needed to meet the requirement of this specific policy and to tackle the challenge of different member states interpreting and producing different outputs in different units which are not comparable. The cascade model also helps in emphasizing the importance of resilience as ecosystem functions are a crucial step in providing ecosystem services and overcomes the challenge of not accounting for sustainable use of ecosystems.

4.2. Application of the framework using water purification services as example

The conceptual cascade framework was applied in a pan-European exercise to map several ES at multiple scales in support of European biodiversity policy (Maes et al., 2011). For the purpose of this paper, we used a single case study on water purification services delivered by freshwater ecosystems to demonstrate how the cascade model can be used to move from a conceptual to a methodological approach for mapping ES. More specifically, we use the frame to link several spatial indicators for water purification services and we show how the cascade frame provides entry points to including assessment information on the status of ecosystems to incorporate sustainability. Throughout the case study, nitrogen is used as a common water quality indicator.

The application of the cascade framework for mapping water purification services is explained in Fig. 1. Water which is polluted by excess nitrogen is filtered as it moves through rivers and streams, lakes, estuaries and coastal marshes. These ecosystems provide the biophysical (infra)structure to deliver services. The ecological processes at the basis of nitrogen services are denitrification (bacteria in oxygen poor sediments that convert nitrogen compounds into atmospheric nitrogen gasses), uptake of nitrogen by vegetation, and sedimentation and burial of nitrogen. Both biophysical structure and the processes taking place therein define the capacity of ecosystems to remove nitrogen, referred to as ecosystem function in the cascade model. The actual or realized ecosystem service takes place if nitrogen enters the river network and is subsequently removed from the water phase. Almost all nitrogen that enters the environment has an anthropogenic source, coming from households, industrial discharges, exhaust gases from traffic and, particularly, agricultural application of manure and fertilizers (Sutton et al., 2011). The removal of nitrogen results in improved water quality in downstream areas as represented by a reduction of nitrogen concentration. This social benefit can be valued using currency by estimating the averted costs of water treatment or by measuring the consumer's willingness to pay for cleaner water. The present state of research allows for a detailed, spatially-explicit assessment of where this service is delivered and where the benefits are enjoyed. There is indeed sound knowledge of the spatial distribution of freshwater

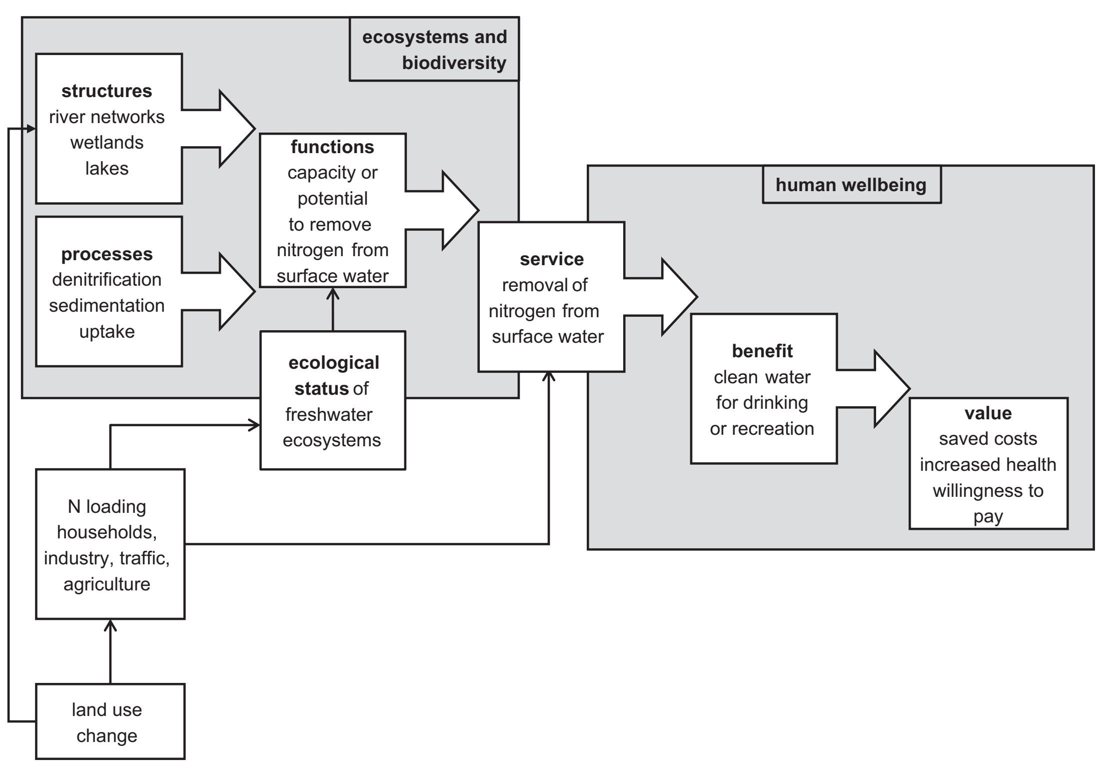

Fig. 1. Application of the ecosystem services cascade framework to water purification.

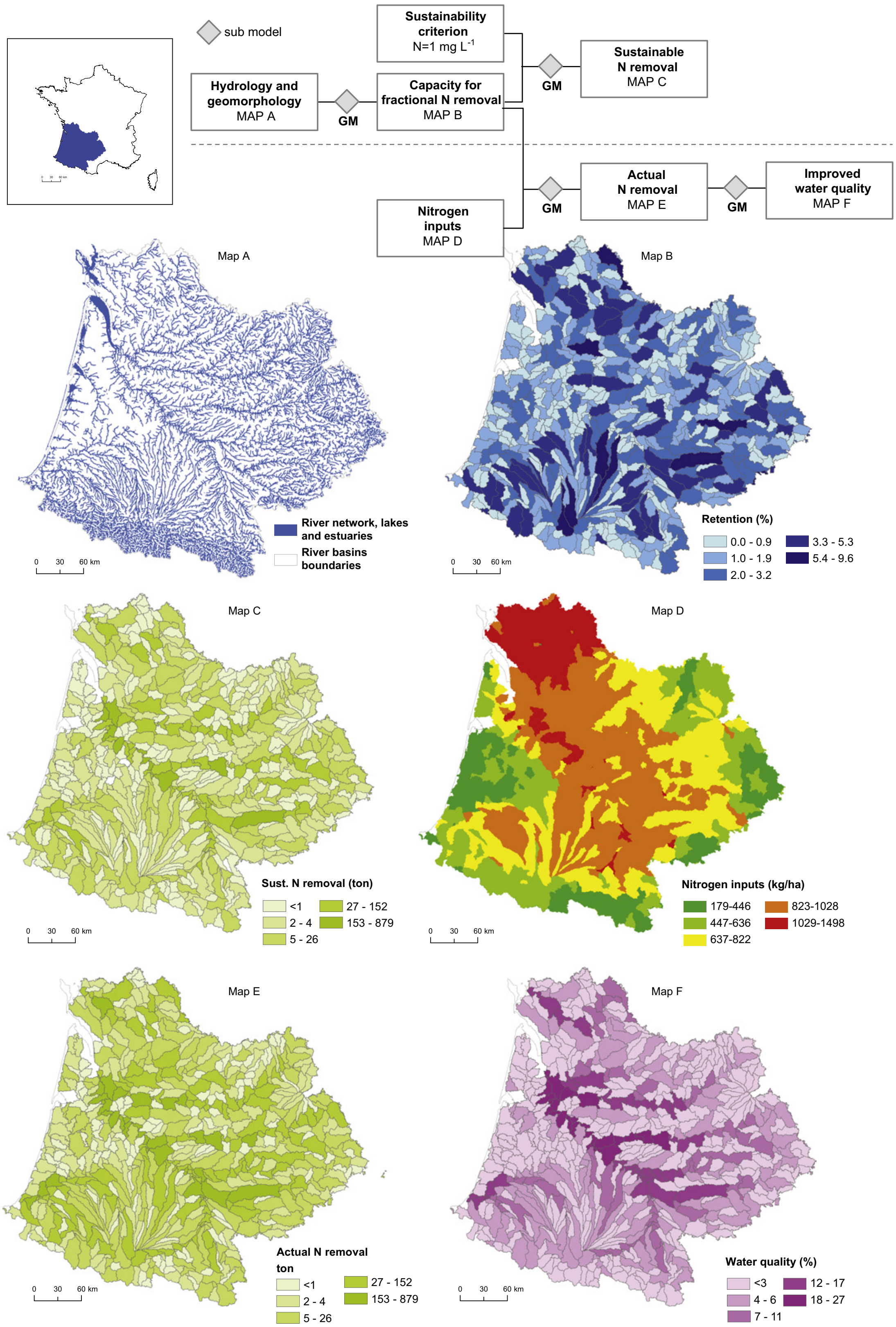

Fig. 2. Mapping water purification services in the river basin district of the Adour-Garonne (France) based on a European scaled model for nitrogen assessments (GREEN, Grizzetti et al. 2008). Maps represent values for the river network the district which is subdivided into 721 catchments. Values on maps B, S, E, and F refer to services delivered by the river network but were plotted per catchment to enhance visual interpretation. Values for nitrogen inputs (Map D) are relevant for the surface area of the catchment. Improved water quality as a result of nitrogen retention is calculated using nitrogen concentration based on model runs with and without retention.
GM: Sub routines of the Green Model

ecosystems and of the understanding of biological nitrogen retention processes. Nitrogen stocks and flows are quantifiable, measurable and scalable across space.

The cascade framework is further operationalized by linking it to two pressures on ecosystems that affect the capacity to remove nitrogen from surface waters. Firstly, land conversion of wetlands to developed land has a direct effect along the different steps of the water purification cascade and result in a loss of benefits, given nitrogen loading (Fig. 1). Land use change may also cause altered spatial patterns of nitrogen loading, for instance through changed farming practices (Fig. 1). Nitrogen loading is a second key pressure that directly impairs the capacity to remove nitrogen by wetlands. High nitrogen concentrations in fresh water result in poor ecological status inhibiting the denitrification process itself (Fig. 1, Mulholland et al., 2008).

In addition, the cascade framework offers the entry points for quantifying how biodiversity underpins water purification. Cardinale (2011) showed that a higher diversity of the community of algal species increased the nitrogen uptake capacity justifying efforts to protect and conserve aquatic biodiversity.

Finally, we can make inferences about the sustainable use of this ecosystem service by considering the effects of nitrogen on the environment. By defining critical nitrogen loadings which correspond to a nitrogen concentration below which no harm to the environment is expected, it is relatively straightforward to calculate the capacity of wetlands to remove nitrogen in a sustainable way and to assess whether or not the wetland is overexploited with respect to water purification services.

4.3. Mapping water purification services

Fig. 2 contains a series of maps that illustrate how the ecosystem services cascade was used as a frame to map water purification services at a regional scale, making the case of the French river basin district of the Adour and Garonne. River basin districts are large river basin management units established under the EU Water Framework Directive which aims to achieve good ecological status of European surface waters by 2015. This example considered nitrogen as an indicator substance.

We used a pan-European statistical model developed to estimate total nitrogen fluxes to surface water in large river basins (Grizzetti et al., 2008; Grizzetti et al., 2012) to map nitrogen services. We mapped the natural capital that delivers the service, corresponding to the network of streams, rivers, lakes, and estuaries (Fig. 2A). Hydrological and geomorphological conditions control the residence time of water in river networks and thus the processing time of nitrogen within an aquatic system. This, in turn, affects the proportion of nitrogen inputs that are removed (Fig. 2B). At the same time, nitrogen inputs (Fig. 2D) limit the amount of nitrogen available for removal (Fig. 2E). With increasing residence time of water in a system, a higher proportion of the available nitrogen can be removed. But also the higher the nitrogen loading, the more nitrogen is removed through denitrification and this is observed across whole range of lakes, rivers, estuaries (Seitzinger et al., 2002). The removal of nitrogen from rivers and lakes results in increased water quality in downstream reaches, which is presented in Fig. 1F. So far, we thus mapped the blue infrastructure which delivers the service, the capacity of the natural capital to provide the service, the actual service which is the removal of nitrogen as a result of nitrogen loading, and the benefits that result from an improvement of the water quality as expressed by a percentage reduction of nitrogen concentration.

The mapping methodology can be used to test scenarios that include nitrogen mitigation measures (e.g., fertilizer reduction and manure management in agriculture) or that comprise ecosystem restoration and investment in natural capital. Wetland restoration increases the retention capacity which results in increased removal of nitrogen, given nitrogen loading.

A map of monetary value was not presented, but the maps of nitrogen removal (Fig. 2E) and improved water quality (Fig. 2F) can be overlaid with monetary values using avoided treatment costs or willingness to pay for clean water, respectively, as was shown by La Notte et al. (2012).

One map remains to be discussed. Fig. 2C estimates the sustainable nitrogen removal by the river network. It is expressing how much of a service can be delivered in a sustainable way. This requires setting a certain criterion for sustainability of water resources with respect to nitrogen. As an example, we used a total nitrogen concentration of 1 mg L^{-1} as maximum threshold concentration below which we do not expect harm to the environment. Clearly, this threshold concentration serves as an example for the purpose of this study only and will change depending on the vulnerability of different aquatic ecosystems to nitrogen loading. Sustainable targets for total nitrogen concentration in freshwater systems can for instance be inspired on the requirements for good or high ecological status required by the Water Framework Directive.

The difference between sustainable nitrogen removal and the actual nitrogen removal shows to what extent aquatic ecosystems are overexploited for their water purification capacity and represents a useful indicator to measure distance to a sustainable target for a regulating ecosystem service.

5. Conclusion

Mainstreaming natural capital and ES into policy and decision making requires a better understanding of the complex decision making processes of the private and public sector across different policy levels. A better understanding of ecosystem service production functions, underpinned by biodiversity, is also essential to link natural capital with human well-being and society. The EU Biodiversity Strategy sets an ambitious research agenda recognizing the high potential of mapping ES for policy support and decision making.

To support EU policies in a more effective way, clear and specific definitions of the different ES including their appropriate units are needed so that they can be used for setting policy and management objectives as well as for natural capital accounting. In addition, several knowledge gap needs to be addressed making reference to ecosystems and their services for which additional data are required in order to map the complete spectrum of ES.

Finally, we argue that the ES cascade model provides a conceptual approach to developing a methodological framework for mapping biophysical flows and social values coming from ecosystems. The cascade model helps also in identifying and distinguishing among ES indicators, and avoiding misunderstandings in decision-making that might arise based on varying results between the studies. The water purification case illustrated how substantial efforts which have gone into environmental modeling and monitoring can be used to derive sets of scalable and harmonized maps that depict the sustainable flow of ecosystem services from ecosystems to society. Such model based approaches result in a better exploration of scenarios and policy alternatives and can reveal potential future synergies and conflicts among ES and between ES and other policy targets.

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