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A spatial assessment of ecosystem services in Europe: Methods, case studies and policy analysis - phase 1

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Summary

In order to make a comprehensive and compelling economic case for the conservation of ecosystems and biodiversity it is essential that we are able to understand, quantify and map the benefits received from ecosystems and biodiversity, and assign values to those benefits.

The PRESS (PEER Research on Ecosystem Services) initiative is a collaboration between PEER (Partnership for European Environmental Research) research institutes addressing some of the knowledge gaps which stand in the way of performing a spatially-explicit, biophysical, monetary and policy assessment of ecosystem services in Europe. The starting point for this report is the need to upgrade the knowledge basis of land-use information and mapping to reflect the existing knowledge about ecosystem services and their social and economic values to better inform policy design and decision making processes.

Ecosystem service maps have been developed for water purification and recreation as examples of regulating and cultural services, respectively. In the water purification case, the contribution of rivers, streams and lakes to purifying water through the removal of nutrient pollutants from runoff water was estimated in ton per km river network. The analysis showed that at a European scale, 1.5 milion ton of nitrogen is removed from surface waters, an amount equalling the combined input of point sources. The recreation study case has developed an approach for mapping recreation services offered by agricultural, semi-natural and natural areas considering also the accessibility of nature to citizens. Results show that the majority of the European population has access to areas where accessibility is high and where nature is of medium quality. For forest services, the assessment of methodological needs for mapping was completed.

The analysis at regional and EU level revealed that there is high potential for integrating ecosystem services into policies and for supporting this with mapping exercises. The appearance of synergies and trade-offs and their relevance for decision-making is strongly dependent on the scale of the discussion and on the specific ways in which ecosystems are managed. This means that policies have a great potential to harmonise trade-offs or conflicts between ecosystem services e.g. by supporting specific management practices.

There is a need for the development of hierarchical sets of ecosystem service indicators, following the European SEBI (streamlining European biodiversity indicators) example, but geographically explicit and linked to the EU-2020 Biodiversity Strategy.

PART 1. SYNTHESIS REPORT

1. Introduction

Until now, global, European or national biodiversity policies which aimed to reduce or stop the loss of biodiversity, essentially focussed on the protection and conservation of endangered habitats and species. At a European scale, a well known exponent of a conservation approach to biodiversity protection is the development of the Natura 2000 network established under the Habitats Directive.

In spite of substantial efforts in order to better protect nature, there is compelling evidence that the globally agreed 2010 target of stopping the loss of biodiversity has not been met. In contrast, biodiversity, ecosystems and the services they provide continue to deteriorate. Many of the pressures that affect habitats and species, including the conversion of ecosystems for other purposes of land use, climate change, invasive species, fragmentation of the land, pollution and overexploitation of biological resources, continue to impact biodiversity.

The Millennium Ecosystem Assessment has increased the awareness of the negative consequences of biodiversity loss to human welfare by addressing the value of ecosystems and biodiversity for sustaining livelihoods, economies and human well-being. As a follow-up to the MA, the economic value of biodiversity and ecosystem services (ESS) has been demonstrated in The Economics of Ecosystems and Biodiversity study (TEEB 2009; 2010), which also highlighted the costs of biodiversity loss and ecosystem degradation. Failing to incorporate the values of ecosystem services and biodiversity into economic decision making has resulted in investments and activities that degrade the natural capital.

As a consequence of these studies, new policies at global and European levels have complemented conservation based biodiversity targets with the argument of ecosystem services. The assumption is that by protecting ecosystems and the services they provide as benefits to humankind, the world's biodiversity resources can be better safeguarded for future generations. In particular, the international Convention on Biological Diversity (CBD) has reached consensus on a new strategic plan envisioning that by 2020 ecosystems are resilient and continue to provide essential services, thereby securing the planet's variety of life, and contributing to human well-being and poverty eradication.

Following the agreement reached at a global level, the European Commission is developing a post 2010 strategy which responds to the challenging targets advanced by the CBD and which aims to mainstream the value of nature in other policies. The inclusion of ecosystem services into biodiversity policies increased the demand for demonstrating the value of natural capital in order to justify investments in biodiversity protection. Hence, in order to make a comprehensive and compelling economic case for the conservation of ecosystems and biodiversity it is essential that we are able to understand, quantify and map the benefits received from ecosystems and biodiversity, and assign values to those benefits.

Such an assessment necessitates the development of ecosystem services maps and models in order to estimate where ecosystem services are produced, to quantify the changes in service provision over time, to describe the production of ecosystem services as a function of patterns of land use, climate and environmental variation. Importantly, a spatially explicit assessment of ecosystem services can couple biophysical estimates of service provision to an economic and monetary valuation. Assessing and valuing ecosystem services in this manner, is, however, not a purely scientific activity, but to a strong degree also depends on societal choices. It must thus be embedded into a policy context and be connected to decision making processes.

1.1. Objectives of this study

The PRESS initiative is a collaboration between PEER research institutes addressing some of the knowledge gaps which stand in the way of performing a spatially-explicit, biophysical, monetary and policy assessment of ecosystem services in Europe. Here we report on the first results focused on a selection of cases at different spatial scales to test and further develop methodologies for mapping indicators and policy analysis. In 2011, it is proposed to complete a set of examples on how the spatial explicit distribution of ecosystem services is influenced by changing scenarios (land use, climate) and what is the associated monetary valuation.
The objectives of this report are threefold:

- To demonstrate the present research capacity for developing maps at different spatial scales that quantify the flow of ecosystem services.
- To identify methods for assessing and reporting on ecosystem service targets and trade-offs and synergies between them.
- To assess policies affecting the current and future management of ecosystem services, including policies in the environmental, agricultural, fisheries, transportation, regional development and other domains.

1.2 Structure of the report

The report is structured in two main parts. The first part contains a synthesis of the main results and achievements of this study. It is built around the conclusions of the policy and methodological analyses and it includes the essential maps of ecosystem services developed so far. A second part of the report is more technical and develops in more depth the different approaches and methodologies that have been used and reports extensively on the results.

Box 1. Ecosystem services in brief

Definition. *Ecosystem services (ESS) are the benefits people receive from nature (MEA 2005). The TEEB study used a different definition and termed ESS as the direct and indirect contributions of ecosystems to human well-being. Biodiversity underpins the supply of ESS as living organisms, chiefly plants and microbes, work together to maintain the composition of the atmosphere, regulate the climate, provide clean water, control erosion, fix atmospheric nitrogen, detoxify pollutants and generally make the earth inhabitable (Thompson 2010).*

Classification. *The TEEB study proposes a typology of 22 ESS divided into four main categories: provisioning, regulating, habitat and cultural services, mainly following the MEA-classification. Provisioning services are the goods and products obtained from ecosystems such as food, water, timber or medicines. Regulating services are the benefits obtained from an ecosystem's control of natural processes, for instance pollination or climate regulation. Cultural services are the non material benefits obtained from ecosystems such as recreation in forests. Finally, habitat services are supporting the provision of other services by providing habitat.*

Ecosystem service valuation. *ESS are in essence an economic argument to protect biodiversity. Demonstrating the monetary value of ecosystems is considered useful for decision making that affects biodiversity. Often, conversion of ecosystems, for instance the cutting of forest for crop production, leads to short term economic gains for private investors, but the costs of this conversion are mostly paid by society in the long term and refer to the loss of capacity to provide useful services such as water and carbon storage and capture. Monetary valuation, also of non-marketable services such as regulating and cultural services, brings these underestimated values of ecosystems to the attention of policy and management.*

Ecosystem service indicators. *Ecosystem service indicators are information that efficiently communicate the characteristics and trends of ESS, making it possible for policy-makers to understand the condition, trends and rate of change in ecosystems (Layke 2009).*

Ecosystem service targets. *The new strategic plan aims to enhance the benefits from biodiversity and ecosystems. Parties to the CBD convention will have to report progress towards achieving the 2020 biodiversity targets:*

- *Ecosystems provide essential services and livelihoods are safeguarded and restored with equitable access.*
- *Ecosystem resilience and the contribution of biodiversity to carbon stocks are enhanced, through conservation and restoration, including 15% of degraded ecosystems.*
- *Access to genetic resources is enhanced and benefits are shared.*

2. Potentials and limitations of the concept of ecosystem services in environmental decision making: synthesis

To ensure the effective use of ecosystem services mapping, an analysis of potential trade-offs and synergies between different ecosystem services and policy measures which affect the provision of ecosystem services has been carried out. This may enhance the policy relevance of mapping activities concerning the search for better integrated policy strategies, and will align the outcomes of the analysis of the spatial dimensions of ecosystem services with an analysis of the implications of ecosystem-related decision-making along the following key questions:

- Which ESS are emphasized by decision makers in different contexts (and why)?
- Which ESS (as implicit targets) are already covered by policies/regulations?
- Which trade-offs and synergies exist between ESS?
- How do policy measures affect trade-offs or synergies between ESS?
- Which problems will occur in decision-making concerning/using the concept of ESS?
- How can spatially explicit information be used in decision-making concerning ESS?

This assessment is based on an analysis of the relevant EU documents complemented by key informant interviews with interview partners at a sub-national, i.e. policy-implementation level, in three regions: Satakunta in Finland, Saxony in Germany, and Silesia in Poland. The collected material was used for preparing a focus group discussion in Brussels with policy-makers from the EU and member states and for an online survey in the three regions mentioned above.

2.1 Ecosystem services emphasized and covered by policies

Ecosystem services are, following the Millennium Ecosystem Assessment, “the benefits people obtain from ecosystems”. This implies that human choices to a large degree determine what counts as ESS and which of them are more important than others. Figure 2.1 shows the weighted results from the regional survey, where respondents were asked to select the most important ESS for their region.

The analysis of EU policy documents as well as discussions in the focus group in Brussels and the key informant interviews revealed that, although ESS are not mentioned explicitly as such (with the exception of some conservation-related documents), most of the important ESS are targeted as well as impacted by various policies. Dominant subjects are clearly food and biotic raw materials, water purification, climate and water flow regulation, as well as recreation. Examples of the relevant documents are the Common Agricultural Policy, Floods Directive and Water Framework Directive, Habitats and Birds Directives and the Directive 2009/28/EC concerning the use of energy from renewable sources. Timber is not tackled at the European level, but rather at the national and regional levels (in terms of competences). Cultural services are often only addressed on national or sub-national levels and mainly by general strategies rather than by exact regulations. Further, participants of the focus group were concerned about conceptual vagueness of the term “ecosystem services” and the lack of knowledge especially about cultural services and non-marketable services. This poses a challenge for integrating these services into policies and makes the definition of policy targets through ESS difficult.

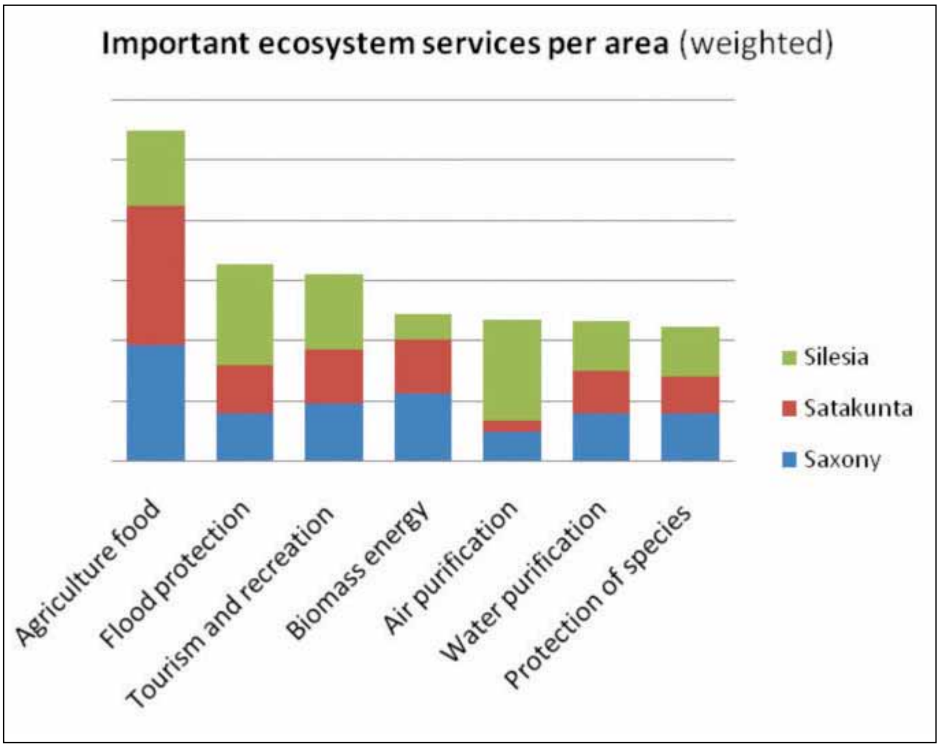


Figure 2.1 Aggregated weighted number of times the services were selected to be important. The responses are weighted since the numbers of respondents varied considerably across regions.

2.2 Policy and management trade-offs and synergies between ecosystem services

While trade-offs and synergies between different uses of ecosystem services and between policies affecting them were difficult to detect in policy documents, they became quite obvious in the empirical material. In the focus group the major synergies were seen between different services provided by forests (water and climate regulation, timber, recreation), while some trade-offs were identified between provisioning services (e.g. food) and other services (e.g. water purification). Figure 2.2 presents trade-offs and synergies identified in the online surveys.

It is not at all self-evident how different ecosystem services relate to each other and generalisations were hard to draw on European or even national and provincial levels as perceptions and knowledge of respondents varied according to geographical characteristics of the regions. Further, respondents pointed out that different forms of uses and practices lead to diverging outcomes for other services. For instance, organic farming can have very positive consequences for biodiversity in cultural landscapes, as well as for soil formation or erosion prevention. In contrast, conventional (industrial) farming can result in very negative impacts on all of the mentioned ecosystem services.

The appearance of synergies and trade-offs and their relevance for decision-making is strongly dependent on the scale of the discussion (in particular between levels of policy formulation – European and member state level – and levels of policy implementation – mostly regional or local) and on the specific ways in which ecosystems are managed (e.g. different forestry and agricultural practices). This means that policies have a great potential to harmonise trade-offs or conflicts between ecosystem services e.g. by supporting specific management practices.

	Silesia	Saxony	Satakunta	Silesia	Saxony	Satakunta	Silesia	Saxony	Satakunta
	Agriculturally produced food			Flood protection			Protection of valued species		
Agriculture: food				!!				!	
Aquaculture				!!			!!		
Wild food	!!								
Timber							!!	!!	
Biomass energy	!!			!!					
Biochem., med. and genetic resources			!!						
Water purification			!!						
Flood protection			!!					!	
Erosion prevention		!							
Air purification			!!						
Protection of species		!							
Climate regulation			!!	!!					
Aesthetic and spiritual			!!						
Tourism and recreation		!	!!	!!					
Science and education			!!						

Figure 2.2. Impacts of provisioning of selected ESS (in the columns) on provisioning of other ESS (in the rows). The green and red colours show the average of positive and negative relationships respectively, with more intense colours implying stronger synergies and trade-offs. The boxes containing exclamation marks indicate controversy among opinions within the respective region.

The relationship of ecosystem services with other types of land use was addressed several times. All types of land uses cannot be captured with the concept of ecosystem services, but they are closely linked with many ecosystem services – often having a negative influence. Some respondents even pointed out that the conflicts and trade-offs between ecosystem services are rather unimportant in comparison with conflicts between ecosystem services and other types of land-use (e.g. energy or transport infrastructure).

2.3 Mapping for decision support

Example maps showing trade-offs between the services of water purification and agricultural food production were used to discuss potentials and limitations of maps for decision-makers on a regional and EU level. The following potentials and challenges of ESS mapping were identified:

Potentials of maps:

- Maps are useful in *problem identification and framing:* they help to identify conflicts and synergies and indicate areas where particular ESS or biodiversity aspects are threatened (e.g. aquatic ecosystem endangered to lose its good ecological status).
- Maps are *heuristically useful* for initiating discussions about solutions and as visualisation for alternatives (simulations).
- Maps can be used as a *scientific basis* for decision-makers for identifying potential policy measures, improving the targeting of measures, and demonstrating/evaluating benefits of policy measures in relation to the costs.
- Maps are already used extensively and represent indispensable instruments in sub-national *planning activities* e.g. for biodiversity protection areas and showing relationships, especially for potential conflicts between different land uses. Maps could be extended to *show the potential* of a spatially explicit landscape *to provide services not yet covered.*
- Maps have a *pedagogic value* by explaining the relevance of biodiversity and ESS to the public.

Challenges identified:

- Some ESS (cultural and regulating ESS) cannot be easily presented on maps.
- The spatial and temporal scales of maps of ESS and the scales of decision-making are not necessarily identical, e.g. seasonal events are difficult to visualise on maps.
- The production of maps with a high resolution is costly, and even maps with high spatial details are often contested from the local level as inaccurate.
- Existing databases for maps, scientific expertise, and modelling work might be too scattered and heterogeneous to serve currently as a base for EU-level decision-making.
- Identifying problematic areas on a map can result in stigmatisation of the regions which appear to provide only few ESS or can indicate high potential areas where exploitation can be increased.

Mapping of ecosystem services will remain a prior tool to support the new biodiversity strategy of the European Union and the Member States. Target 2 of the EU's biodiversity strategy aims to maintain and restore ecosystems and their services and a particular action addresses mapping of ecosystem services. Ecosystem service maps, either at individual service level or at landscape level with metrics accounting for the delivery of multiple services, will be essential tools to prioritize investments in Green Infrastructure, which will be developed as a network of natural areas, semi-natural areas and green spaces that contribute to biodiversity conservation and enhancement of ecosystem services.

2.4 Conclusions

The analysis of EU policy documents and interviews revealed that there is high potential for integrating ESS into policies and for supporting this with mapping exercises. Even though ESS are hardly mentioned explicitly, many regulations (e.g. the EU Water Framework Directive and in this context the forthcoming Blueprint to Safeguard Europe's Water) and policies implicitly refer to them, or are relevant for them. While the identification of trade-offs and synergies between policies turned out to be too complex to be tackled thoroughly, many tensions between different governance levels of policy formulation (EU and national states) vs. the implementation level (local-regional) were detected.

The analysis of trade-offs and synergies clearly shows that the concept of ESS bears the risk of over-simplification. Many conflicts and synergies depend on the forms and uses of the ESS, as explained using the example of agricultural practices. Further, many conflicts and synergies only become apparent at the regional level, and therefore the inclusion of the rich regional and local knowledge and perspectives, as demonstrated in the online survey, needs to be included in policy development and decision-making. This would greatly contribute to designing policies which are sensitive to scale issues and to the differences in practices of ESS use. This also includes non-marketable and non-map-able ESS, which are otherwise at risk to be overseen.

Such an inclusive approach would, however, pose the need to develop a common understanding of ESS, make agreements and define targets in the context where many different actors, their perceptions and interests are involved. This is especially difficult in the case of land use, as land is usually a scarce resource and conflicts are likely to increase. Further, such an approach would require (policy) tools suitable for addressing ESS, mainstreaming them in all relevant policy fields, reconciling different needs for ESS and linking the policy levels to management levels. Last but not least, there is still the challenge to ensure that the inclusion of the concept of ESS in policy-making leads to positive consequences not only for human beings but also for the environment and that other (non-utilitarian) values of nature and biodiversity are not neglected.

While this study clearly demonstrates that the concept of ESS and the mapping of ESS have a great potential to support policy-making, the number of crucial open questions and concerns also shows that further research is urgently needed.

3. Spatial assessment of ecosystem services: synthesis

The inclusion of ecosystem services in biodiversity and land use planning policies requires more detailed knowledge of the services that are produced by ecosystems, thereby recognizing that several ecosystem services including cultural and regulating services may be delivered by semi-natural and agricultural ecosystems as well. Furthermore, one of the outcomes of the TEEB study (The Economics of Ecosystems and Biodiversity) is that ecosystem service assessments must generate better maps showing where ecosystem services are produced at what quantities taking into account the spatial scale of assessment. Such an assessment necessitates the development of ecosystem services maps and models in order to estimate where ecosystem services are produced, to quantify the changes in service provision over time, to describe the production of ecosystem services as a function of patterns of land use, climate and environmental variation.

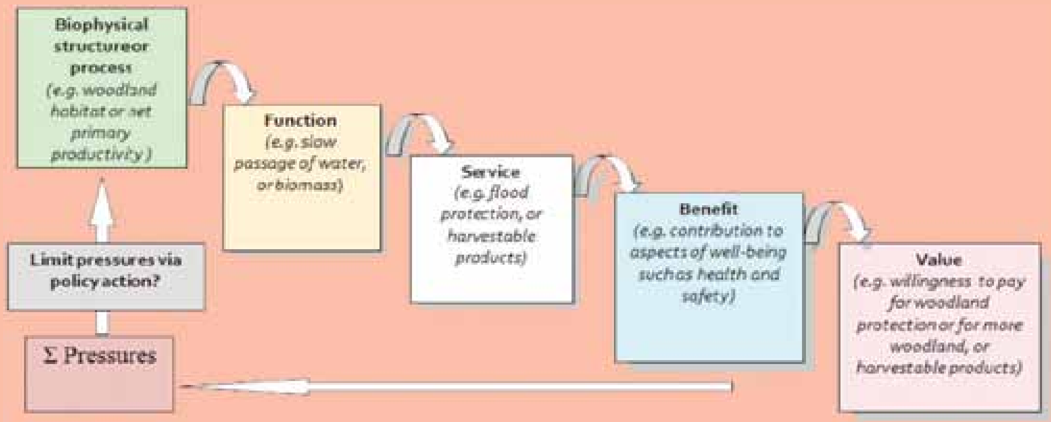
This section demonstrates the results of biophysical mapping of ecosystem services. In particular, three case studies served as examples of how to combine existing information present in statistics and models to derive new maps of ecosystem services. Special attention was given to multi-scale assessments. The policy analysis showed already that synergies and trade-offs between ecosystem services may differ depending on spatial scale. It is expected that biophysical assessments at different spatial scale will reveal similar changes in synergies and trade-offs. Model and maps at high spatial resolution for local areas may contain useful information on the management of ecosystems which lead to local synergies in service provision. Such detail is often not available for regional assessments. In this study, we have attempted to address differences between mapping approaches that relate to scale.

The first case study addressed the cleaning capacity of aquatic ecosystems as they remove pollutants and contribute to the supply of fresh and clean water. In particular, nitrogen is used as an indicator. A second case study has elaborated on methodologies for mapping the potential of nature to provide recreation to humans. A third case study has focused on the role of forests in providing services.

Box 2. Ecosystem service cascades as a frame for mapping ecosystem services

A way of representing the logic that underlies the ecosystem service paradigm and the debates that have developed around it is shown in the figure below (Hains-Young and Potchin 2010). The diagram makes a distinction between ecological structures and processes created or generated by living organisms and the benefits that people eventually derive. In the real world the links are not as simple and linear as this. However, the key point is that there is a kind of cascade linking the two ends of a 'production chain'. Defining ecosystem functions, services and benefits, and the context for CICES (Haines-Young and Potschin, 2010).

The cascade model contains also the notion of stocks and flows. Layke (2009) defines stocks of ecosystem services as the capacity of an ecosystem to deliver a service while the flow corresponds to the benefits people receive. Stocks may be expressed in total size area or the total biomass whereas the associated ecosystem service flow or output must have units per time period.



The capacity of an ecosystem to provide a flow is not necessarily measured in hectares or ton since the capacity does not only contain a quantity aspect but also a quality aspect. Given the quantity, an ecosystem may provide more output if it is in a healthy state. As a result, the capacity of such system to produce services will be higher. Ecosystems in a healthy state are considered resilient systems which are able to recover after disturbance.

We have used this cascade model for framing the indicators that we developed for mapping ecosystem services. Ideally, ecosystem services are modelled following the cascade from the left to the right. At least, indicators are developed capturing both the biodiversity and ecosystem stocks that generate the services and the final benefits as flows of goods and services.

Applying the cascade model to water purification, this report presents maps of ecosystem service indicators that measure the capacity of wet ecosystems to retain nutrients and pollutants as well as the associated flow of services and benefits in terms of the amount of pollutants removed and the effect on water quality. In the recreation services application, we assessed that capacity of ecosystems to provide recreation as a service (potential recreation map) and we estimated the number of citizens that have access to daily recreational opportunities.

3.1 Multi-scale assessment of water purification services by ecosystems

Ecosystems provide clean water by retaining, storing and regulating precipitated water in lakes, rivers and soils. Furthermore, water which is polluted by heavy metals, excess nutrients or pesticides is filtered as it moves through wetlands, rivers and streams, floodplains and riparian zones, and estuaries and coastal marshes. These services that natural ecosystems provide have been classified as water provision, water regulation and water purification and they secure water resources for human use and consumption. Water purification services delivered by aquatic ecosystems are based on particular physical and biological ecosystem properties and functions, in particular the prolonged residence time of the water and a rich and healthy aquatic biodiversity which enables aquatic ecosystems to process and remove pollutants.

At the scale of aquatic ecosystems, wetlands, lakes and slow running rivers and estuaries are characterized by extended residence time of water which enables micro-organisms such as bacteria and plankton and higher water plants such as reed to take up, process and mineralize pollutants, organic matter or excess nutrients. Floodplains, estuarine and coastal marshes and vegetation buffer strips enhance this functioning by storing water temporarily or by obstructing increased runoff and hence, increase the time during which organisms can break off pollutants. At the catchment scale, forests, grasslands and riparian areas buffer the runoff of precipitated water and the unwanted chemicals that are transported to surface waters preventing downstream nutrient enrichment and pollution. At the larger river basin scale, pollutants are retained and processed over tens of years in soils and aquifers before entering surface waters.

The collective functioning of these different ecosystems at various spatial and temporal scales leads to the immobilization of pollutants or in some case the removal from the environment. In turn, water purification results in the provision of clean water that can serve multiple uses: habitat for species and different uses for humans.
In this case study, we applied different modelling approaches in order to map water purification as ecosystem service at different spatial scales. The case study focused mainly on the removal of nitrogen by surface waters, but also other pollutants (organic carbon) and other ecosystems (soils and floodplains) were considered.

Water purification in rivers and lakes

We estimated the contribution of rivers, streams and lakes to purifying water through the removal of nutrient pollutants from runoff water. The methodology is based on models that calculate a nitrogen budget within the boundaries of watersheds, catchments or river basins. We used the processes that are addressed by these models to infer how much nitrogen is retained by surface waters as a service of water purification. This ecosystem service was mapped at three spatial scales: the Yorkshire Ouse catchment (UK) (Figure 3.1), the German Elbe river basin (Figure 3.2) and the Europe (Figure 3.3) including all river basins that drain into European seas. More case studies are available in the technical report.

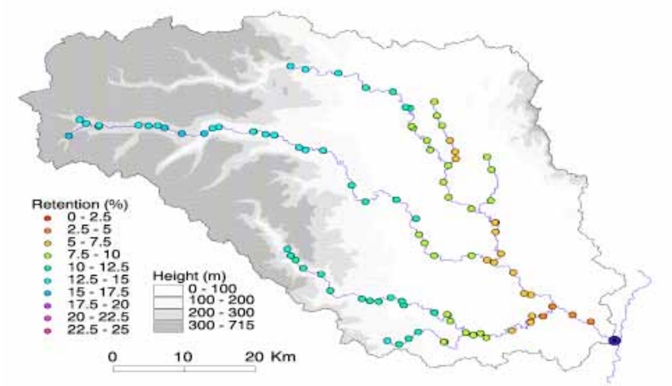


Figure 3.1. Nitrogen retention capacity at river basin scale in the Ouse catchment at median river flow rates

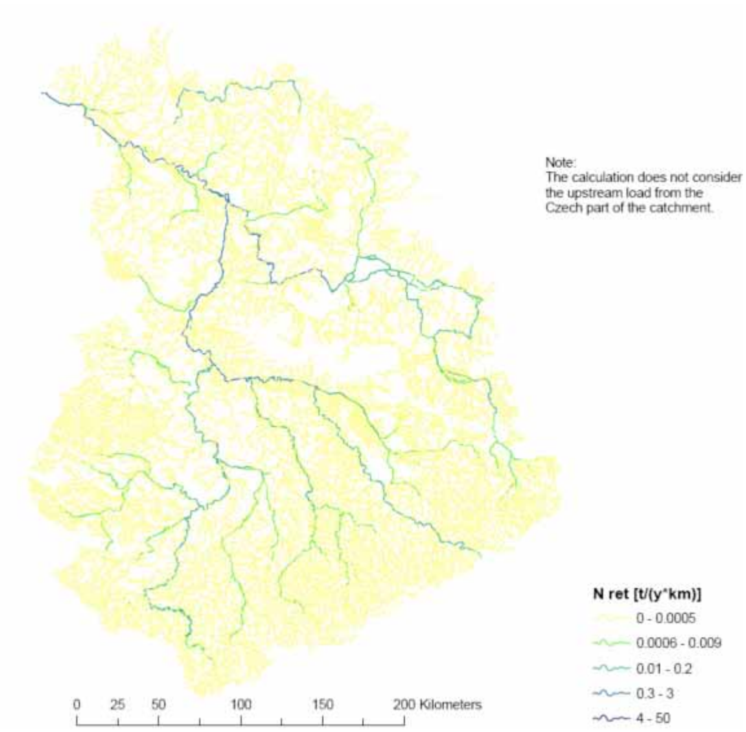


Figure 3.2. Nitrogen removal (ton km^{-1} $year^{-1}$) at basin scale for the Elbe river network

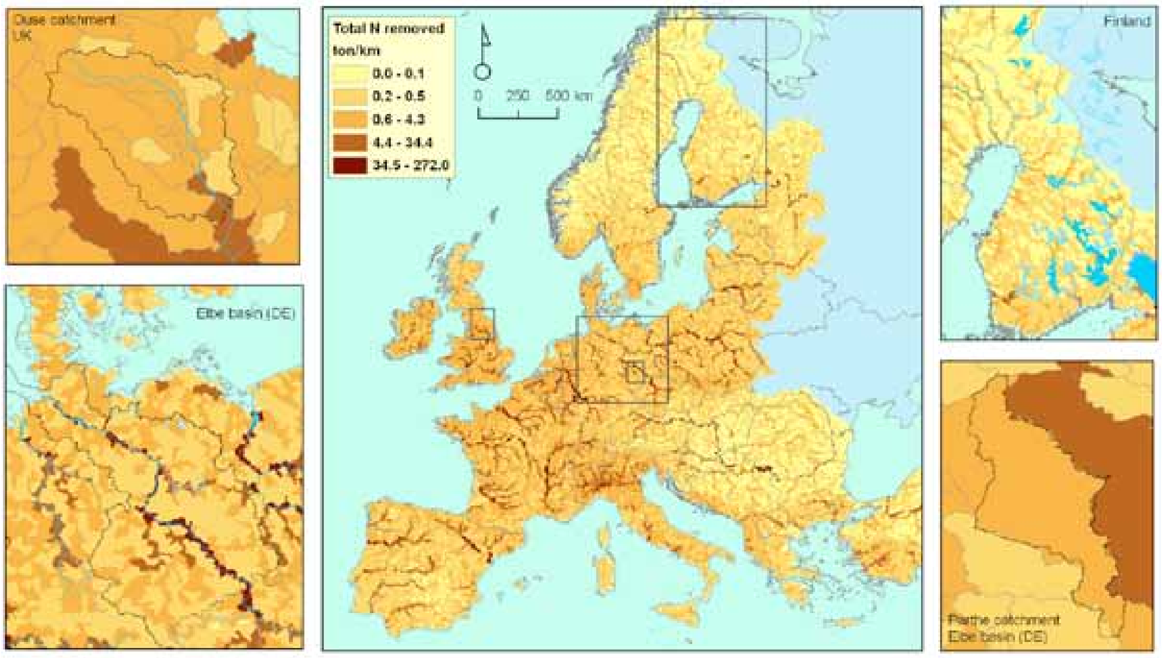


Figure 3.3. Total nitrogen removed (ton km^{-2} year^{-1}) by river networks at European scale

This report showed that rivers, streams and lakes have an important function in nitrogen removal. All methods used show that total nitrogen retention equals roughly the combined input of all point sources coming from industry and households to the river network. For Europe, the river and stream network was estimated to remove 1.5 million ton of nitrogen from surface waters (Table 3.1).
Where this service results in a relatively small improvement of the water quality, on average, benefits increase in downstream direction and in-stream retention results in a 50% reduction of the nitrogen concentration.

Table 3.1. Water purification services as indicated by nitrogen retention

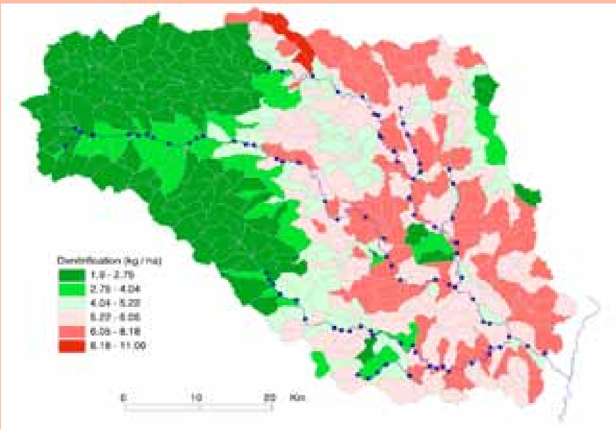
	Europe 10^6 ton year^{-1}	Elbe (DE) 10^3 ton year^{-1}		Ouse (UK) 10^3 ton year^{-1}	
Scale of the model:	**European**	**European**	**Basin**	**European**	**Catchment**
Total point sources	1.4	26	23	0.96	0.62
Total diffuse sources	45.0	1100	748	45.9	54.9
Nitrogen removal by rivers and lakes	**1.5**	**57**	**48**	**0.86**	**0.67**

Separate models were used for Europe (GREEN), for the Ouse and for the Elbe (MONERIS). The table reports the total input of point and diffuse sources estimated by the different methodologies and compares the removal of nitrogen over the different models used. For the European case, only totals are reported.

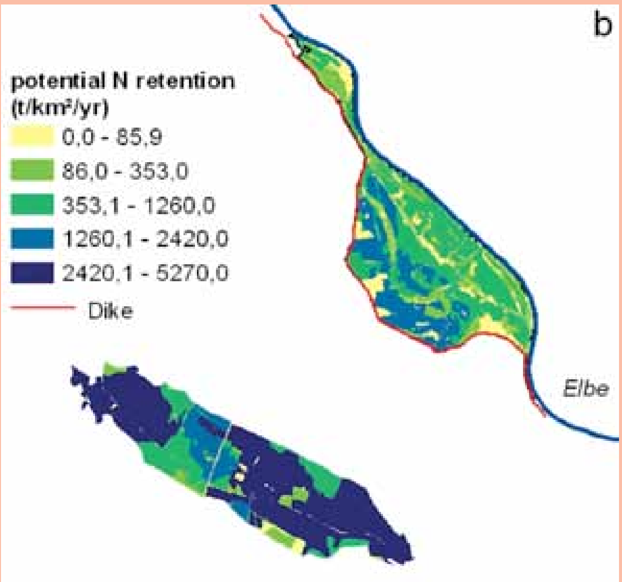
The equations used to model the retention of nitrogen in rivers are scalable. This has successfully resulted in consistency of comparative outcomes over the spatial scales that have been considered, as illustrated by fluxes derived from European-, basin- and catchment-scale modelling methods (Table 3.1). In particular for the Ouse river catchment, the regional model developed for the Ouse yielded estimates of the same order of magnitude of retention, nitrogen removal and effect on water quality as a European approach. A similar conclusion was made for the Elbe catchment while for the Finnish case differences were observed, in particular in relation to the retention capacity of lakes and peat lands, which were underestimated following a European based approach. So, although catchment or basin based estimates or nitrogen removal from runoff water are comparable, there remain notable differences between regional and local assessments depending on the resolution of the input data. Lakes and wetlands increase the residence time of water resulting in increased retention. In Finland, an increase with 1% of the lake surface area, relative to other land cover, resulted in an increase of 7% of the retention, with a maximum of about 60% retention. Likewise, including detailed river and stream network maps with smaller sized water courses of high order results will cause an increased water residence time not accounted for in a continental scaled approach. The result is that the latter methodology overestimates the biophysical ecosystem service flows that can be attributed to rivers and streams of low order and of large lakes at the cost of small sized streams, wetlands, ponds and lakes.

Box 3. Soils and floodplains as suppliers of water purification services

Organic and wet soils remove nitrogen from runoff water before it enters the river while floodplains further enhance nitrogen removal of river networks by storing water temporarily or by obstructing increased runoff and, hence, increase the time during which organisms can degrade pollutants.
Two examples at local scale show the potential of soils and floodplains to remove nitrogen and demonstrate how this service can be mapped.



Soil denitrification in the River Ouse catchment (UK).
Denitrification is a process whereby bacteria under oxygen poor conditions convert nitrates to atmospheric nitrogen. This service prevents downstream eutrophication and increases water quality. Soil nitrogen removal maps can be made using data on soil moisture, soil temperature and soil organic carbon content. These three factors define suitable conditions for soil biodiversity to act as a sink for nitrogen.



Nitrogen retention in floodplains along the Elbe
Floodplains store excess water and help prevent downstream areas from being flooded. This service results in a positive knock on effect on water quality. The temporary water storage slows down the runoff of water to the sea and increases the time that the ecosystem needs to remove nitrogen from the water. This report contains a methodology explaining how this service can be mapped and quantified.

3.2 Recreational services

Cultural ecosystem services are defined as the nonmaterial benefits obtained from ecosystems, among these the recreational pleasure that people derive from natural or managed ecosystems is defined as recreation service. Natural and semi natural ecosystems as well as cultural landscapes provide a source of recreation for mankind. People enjoy forests, lakes or mountains for hiking, camping, hunting, fishing, bird watching or just for being there.

Contrarily to other services, such as provisioning and regulating, that are providing their flow of benefits independently from the presence of human beings, recreation has the peculiarity of requiring a human agent who performs the action of recreating. We call the associated flow of benefits "fruition" which may be measured by performance indicators such as the number of visitors that annually visit a site or the appreciation of sites based on questionnaires. The relation between capacity and fruition is likely to be positive and is influenced by the accessibility of ecosystems to humans and the infrastructure that is in place to host or to guide visitors. The capacity of ecosystems to provide recreational services was assessed taking into account the degree of naturalness, the presence of protected areas, coastlines, the quality of bathing water and accessibility to the place where the service is provided. The analysis on population data allowed estimating the quality of recreation provision to the European citizens.

Comparison across scales was possible for Finland and the Netherlands, where the availability of high resolution data allowed a more detailed analysis. The Recreation Potential Index and the Recreation Opportunity Spectrum (ROS) have been calculated on the basis of the characteristics of Finnish ecosystems and a detailed road network. Results show that compared to the EU average, a higher share of Finnish population has easier access to areas where the recreation provision is medium. A case for the Netherlands was based on data on the stated preference of people where they like to go for recreation. Particular attention was given to the potential of natural sites in attracting recreants on bikes.

A map of the European Recreation Opportunity Spectrum.

The capacity of ecosystems to provide recreational services was mapped at European level with the assumption that it is positively correlated to the degree of naturalness, to the presence of protected areas (following the assumption that they have been identified as holding a higher degree of naturalness, and as providers of recreation services and facilities), to the presence of coastlines (lakes and sea) and to the quality of bathing water. Accessibility is mapped on the basis of the distance from roads and urban centres, and is added to the frame via the Recreation Opportunity Spectrum (ROS) concept. The purpose of the ROS inventory is to identify, delineate, classify and record areas within a region or country into recreation opportunity classes based on their current state of remoteness, naturalness and expected social experience. In the current application a ROS has been established for Europe (Figure 3.4), adapting overseas experiences to the peculiarities of the European continent, providing a zoning of the EU in terms of proximity vs. remoteness. The resulting layer has then been merged with the Recreation Potential Index, in order to obtain information both on the quality of recreation provision and its accessibility in nine different ROS zones. The analysis on population data allows estimating the quality of recreation provision to the European citizens. The current exercise addresses recreation and not tourism, and in particular daily recreation, ranging from a short walk or a bicycle ride to a car displacement for a Sunday trip. This can be estimated at EU, national or regional level. Population pressure is calculated on the basis of population density, assuming that in daily recreation the maximum travelled distance is 60 km. Final results show that the great majority of population has easy access to areas where quality of provision is medium (44.6%), while 23.7% have easy access to areas where provision is low, and 26.8% to areas where provision is high.

The availability of high resolution data on Finland allowed a more detailed analysis. The Recreation Potential Index and the ROS have been calculated on the basis of the characteristics of Finnish ecosystems

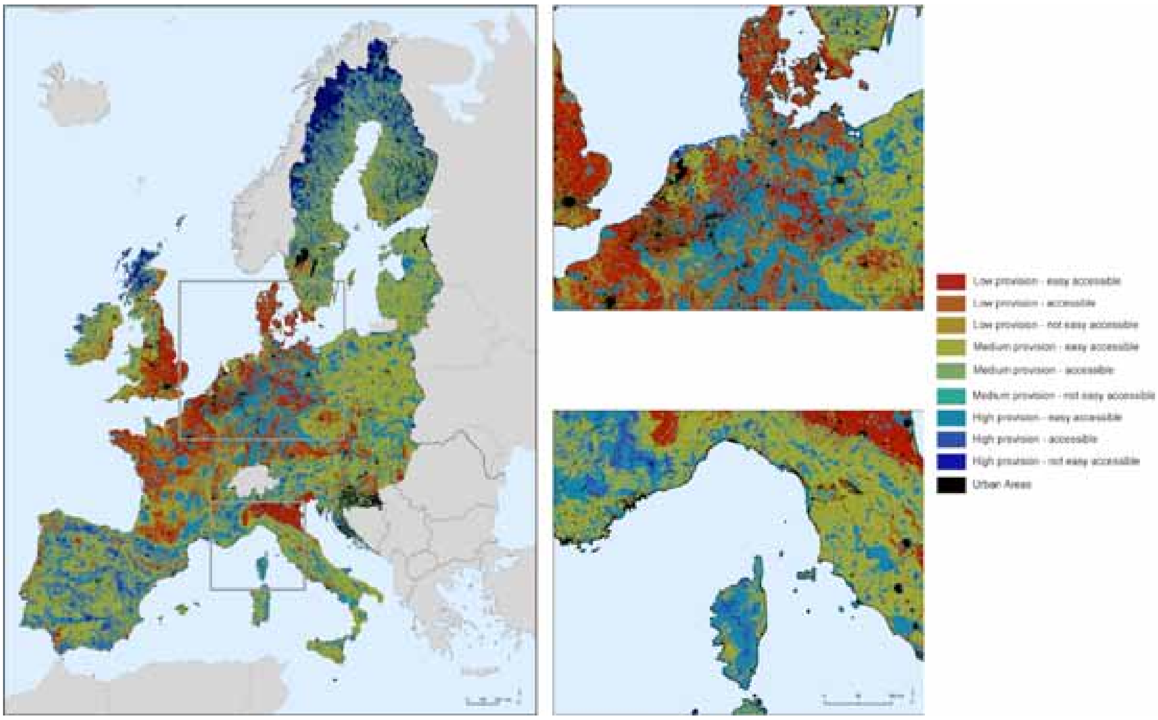


Figure 3.4. Recreation Opportunity Spectrum for the EU24

and road network. Results show that 17.6% of the population has easy access to areas where the quality of provision is low, but 77% of the population has easy access to areas where the quality of provision is medium, while 3% of the population can easily access areas where the provision is high. Furthermore, the provision of recreational services and trails in State owned land could be calculated on the basis of data on infrastructures such as hiking trails, camping grounds, skiing centres, wilderness cabins etc. On the basis of this information a map of the aggregated attractivity of recreation facilities in State owned land was obtained on the basis of attraction distances assigned to each category (Figure 3.5). This variable was used to build an explanatory model for the distribution of summer cottages in Finland, together with accessibility from road network, recreation potential and distance from coast. The model explains 28 to 32% of the distribution of summer cottages.

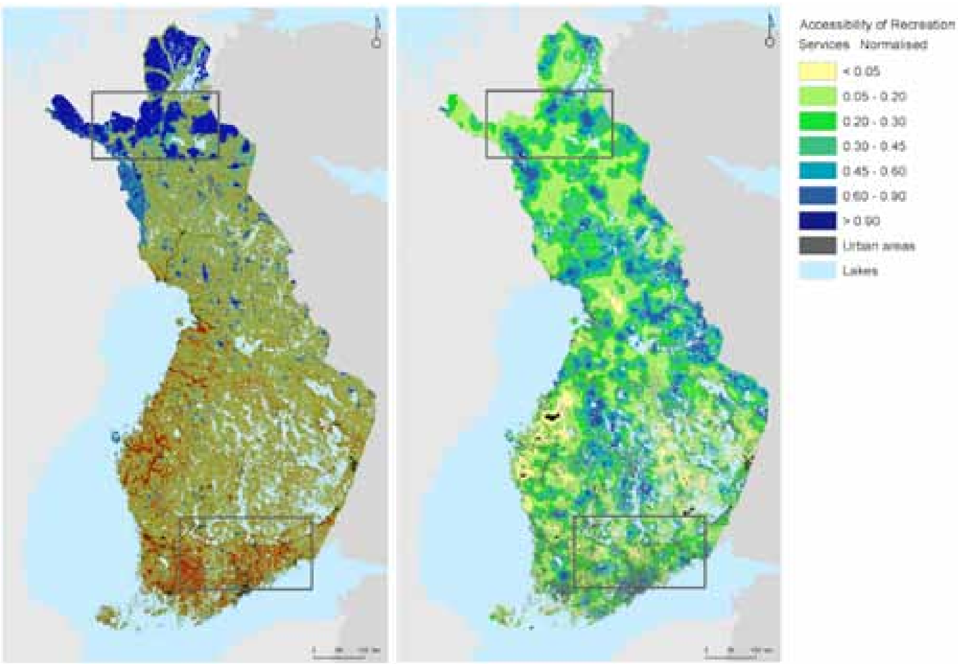


Figure 3.5. Recreation Opportunity Spectrum for Finland (left) and aggregated attractivity of recreation facilities in State owned land

A map of the Dutch recreation opportunity spectrum for cycling

For The Netherlands, the potential use of ecosystems for recreational cycling was mapped using data of the cycling infrastructure, the geographical distribution of Dutch citizens, and the cycling preferences of Dutch recreationists. A final map obtained by combining the indicator on preferences of recreation services and the zoning of The Netherlands on the basis of the potential pressure of recreation cycling is given in Figure 3.6.

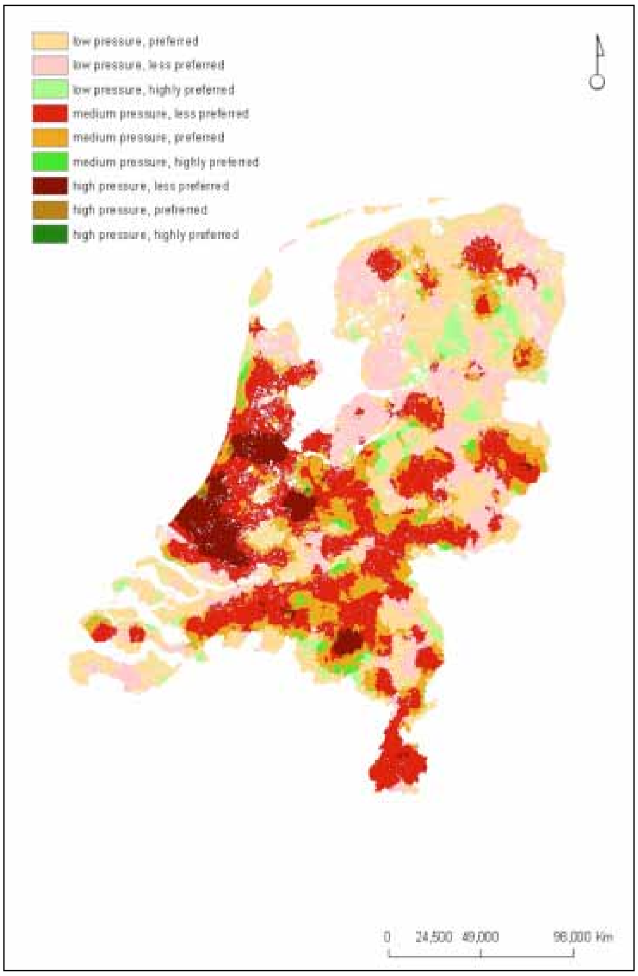


Figure 3.6. The Recreation Opportunity Spectrum for cycling for The Netherlands

3.3 Forest ecosystem services

Forests represent a combination of important ecosystems that provide habitat for numerous species, regulate water-cycles, clean air and provide timber for economic use. In addition to these well recognised regulating and provisioning services, also other forest ecosystem services generate benefits for humans locally and more generally such as fuel-wood, game, berries, mushrooms and flowers. Among less extractive services, the traditionally recognised amenities, such as recreation and nature tourism, have recently been supplemented with carbon sequestration, health enhancing impacts, aesthetic value and identity, the last mentioned ones representing cultural services.
The case study on forest ecosystem services has explored different approaches to mapping forest features and explicit ecological functions and services to the economies of the EU, at the European, national and regional scale. Each of the approaches combined different techniques of data collection and data handling and combined data from forest stand level with data from higher spatial scales.

The overall impression of the case study results was that in the forestry sector data are plentiful, but ecosystem services have not yet been much of an organizing principle in collecting or presenting the data. Maps of forest features are available at all geographical scales but few of them can readily be used to develop ecosystem services maps from.

The national level case shows that in the past few years the attention for ecosystem services in the Netherlands has steadily increased, but so far very few official mapping attempts have been done. There are a few explorative studies, for instance for ecosystem services of soils, but not specifically for forest ecosystems. Data availability is not so much the problem, as forest statistics are readily available, both for the state owned and private forests.

At the regional level, the case in France illustrates that socio-ecosystem services mapping requires to combine three kinds of information: ecosystem characteristics, defining a potential service, socio-economic system characteristics, characterizing the demand (users) and constraints maps, playing a role of filters and structuring the relation between supply and demand. National aggregations have to be done from local studies at a fine scale, in close cooperation with national forest inventories (NFI). The existing local data of NFIs, oriented towards statistical results at large scale, are often insufficient for an appropriate cartography. A lot of progress may be done by the improvement of these local data on ecosystems, through remote-sensing methods boosted by the LIDAR development.

At the habitat level, despite increasing attention to the human dimension of conservation projects, a rigorous, systematic methodology for planning for ecosystem services has not been developed. This is in part because flows of ecosystem services remain poorly characterized at local-to-regional scales, and their protection has not generally been made a priority.

4. Methodological issues in mapping ecosystem services: synthesis

4.1 Mapping ecosystem services indicators

Ecosystem services are the flows of biomass, energy and information from ecosystems to humans and represent actual work performed by ecosystems, affecting environmental conditions for humans. These flows are hard to observe and measure, but they can be inferred from observations or measurements of changes over time in stocks, structure and spatial patterns. In turn, these are the types of ecosystem service indicators which seem most likely to be useful for mapping, either **directly** from aerial photos and remote sensing data or **indirectly** from databases. This study has examined the map-ability of ecosystem services for further applications.

Table 4.1. Map-ability of ecosystem services indicators

PROVISIONING SERVICES		MAP-ABILITY
Food Sustainably produced crops, fruit, wild berries, mushrooms, nuts, livestock, semi-domestic animals, game, fish and other aquatic resources	Production from sustainable sources (ton ha^{-1}) Wild animal and plant production from sustainable sources in tones	Crops can be mapped directly. For other services, only indirect mapping can be performed
Water	Total freshwater resources in m^3	Direct (surface water) and indirect (surface and groundwater) mapping
Raw materials Sustainably produced wool, skins, leather, plant fibre, timber, cork, firewood and biomass	Forest growing stock, increment and fallings; Industrial round wood in m^3 from natural or sustainable managed forests; Pulp and paper production in tonnes from natural or sustainable managed forests; Cotton production from sustainable resources in ton ha^{-1}; Forest biomass for bio-energy in ton of oil equivalent (Mtoe) from different resources (e.g. wood, residues) from natural or sustainably managed forests	Forests are mapped directly but several indicators for products can only be mapped indirectly. Crops for fibre can be mapped directly or models are available
Genetic resources Protection of local and endemic breeds and varieties, maintenance of game species gene pool	Number of crop varieties for production; Livestock breed variety; Number of fish varieties for production	No direct mapping possible. Data for varieties available in many EU countries for crops and livestock
Medicinal resources Sustainably produced medical natural products (flowers, roots, leaves, seeds, sap, animal products etc.); ingredients or components of biochemical or pharmaceutical products	Number of species from which natural medicines have been derived; Number of drugs using natural compounds	No direct mapping possible
Ornamental resources Sustainably produced ornamental wild plants, wood for handcraft, seashells	Number of species used for handcraft work; Amount of ornamental plant species used for gardening from sustainable sources	No direct mapping possible

REGULATING SERVICES		
Air purification Regulation of air quality through exchange of air pollutants with vegetation	Atmospheric cleansing capacity in ton of pollutants removed per hectare	No direct mapping possible
Climate/climate change regulation Carbon sequestration, maintaining and controlling temperature and precipitation	Total amount of carbon sequestered / stored (sequestration / storage capacity per hectare x total area ($Gt\ CO2$)	No direct mapping possible
Moderation of extreme events Avalanche control, storm damage control, fire regulation (i.e. preventing fires and regulating fire intensity)	Trends in number of damaging natural disasters; Probability of incident	Direct mapping for fires and floods; direct mapping of protective functions
Regulation of water flows Regulating surface water runoff, aquifer recharge etc.	Infiltration capacity/rate (e.g. amount of water/ surface area) - volume through unit area/per time; Soil water storage capacity in mm m^{-1}; Floodplain water storage capacity in mm m^{-1}	No direct mapping possible; Maps based on models and soil maps
Waste treatment and water purification Capture and removal of nutrients and contaminants	Removal of nutrients by aquatic ecosystems (ton or percentage); Water quality in aquatic ecosystems (sediment, turbidity, phosphorous, nutrients)	Maps based on models and field data
Erosion control / prevention Maintenance of nutrients and soil cover and preventing negative effects of erosion (e.g. impoverishing of soil, increased sedimentation of water bodies)	Soil erosion rate by land use type	Maps based on models and field data
Pollination Maintenance of natural pollinators and seed dispersal agents (e.g. birds and mammals)	Abundance and species richness of wild pollinators; Range of wild pollinators (km^2)	Maps based on field work and mapping of landscape elements
Biological control Seed dispersal, maintenance of natural enemies of plant and animal pests, regulating the populations of plant and animal disease vectors	Abundance and species richness of biological control agents (e.g. predators, insects); Range of biological control agents (km^2); Changes in disease burden as a result of changing ecosystems	Maps based on field work and mapping of landscape elements
CULTURAL SERVICES		
Aesthetic information Amenities provided by the ecosystem or its components	Abundance and score of objects; landscape types	Maps based on landscape features (direct and indirect maps); survey scores (photo-based)
Recreation and ecotourism Hiking, camping, nature walks, jogging, skiing, canoeing, rafting, diving, recreational fishing, animal watching	Abundance or area of recreation sites; recreational opportunity spectrum	Direct mapping of recreational facilities; Indirect mapping by aggregation of spatial indicators
Cultural values and inspirational services Education, art and research	Abundance or score of objects and areas; landscape types	Maps based on classes of objects; land use; archaeological, natural monuments

Sustainability of ecosystem service indicators

Some authors suggest that ecosystem service indicators need to take account of the sustainability of ecosystem services over time, to ensure that the long-term benefit flow of services is represented. High economic values may arise from over-exploitation of ecosystems, which then may lead to erroneous conclusions about land use and beneficial investments. These phenomena may occur both with provisioning services (e.g. overexploitation of fish stocks), cultural services (e.g. degradation of nature areas due to high tourist densities) and regulating services (e.g. palm oil plantations instead of natural tropical forests). Indicators referring to those services therefore need to reflect (the actual distance from) sustainable production rates. This calls for a clear definition of what sustainability actually means with regard to those services.

The loss of biodiversity and increasing pressures from drivers of ecosystem change increase the likelihood of non-linear changes. While 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. GBO3 (Secretariat of the Convention on Biological Diversity, 2010) documents clearly a great number of such cases. It is therefore crucial to develop a baseline in order to determine where critical thresholds (e.g. population of fish stock within safe biological limits, soil critical loads) and alternative future pathways under different policy scenarios (e.g. fisheries subsidies reform, subsidies in the agriculture sector) may lie. Not all ecosystem service indicators can easily be quantified. To avoid risks of creating a policy bias by focusing on a subset of indicators high on the political agenda or the agenda of vested interests, complementary (not-yet-quantified) indicators must be developed. In parallel, ecosystem service valuations that focus on a single service should be systematically cross-checked to assess the capacity of ecosystems to continue delivering the full variety of other services potentially of interest.

Applications of ecosystem service indicators

In environmental and resource policy, the development of ecosystem services indicators will inevitably have to be accompanied by a clear definition of relevant policy goals to ensure the effectiveness of such indicators as an integration tool. A streamlined set of headline indicators would be sufficient for high level target setting and communication by policy makers, politicians, the press and business, but must be supported by wider sets for measurement and monitoring. In the business world ecosystem services indicators can also be included in corporate reporting standards to communicate the impacts of lost services on company performance and the impacts of companies on provision of these services (e.g. Global Reporting Initiative).

4.2 Issues of scale

The biological processes underlying the ecosystem services determine to a large extent whether the service providing unit is primarily local – regional (for instance pollination), or are without physical boundaries (climate regulation, as defined by sequestration of free CO_2). This means that mapping ecosystem services requires a clear definition of the spatial scale of the measurements (primary data), to be able to trace the data manipulations such as aggregation and disaggregation. In the selection of ecosystem service indicators these aspects of data and data handling need to be made explicit. Some of the indicators may easily be upscaled (or vice versa down-scaled), because their physical dimensions are expressed per unit area. This would suggest looking for indicators with such dimensions, but we realise that for some ecosystem services that may not be so easy or appropriate.

4.3 Valuation

In its first phase, this project has focused on biophysical ESS mapping. Ultimately indicators and maps of ecosystem services should not only be presented in biophysical units but also in terms of economic values to reflect human attitudes and preferences. Two broad approaches can be distinguished: (1) conversion of bio-physical (potential) ecosystem services maps to potential economic value maps, under assumptions that unit values can be placed on the flow of services based on values derived from individual valuation case studies, and (2) spatially explicit mapping of values taking into account social and economic as well as environmental characteristics of individual locations. One of the key challenges in consistent value mapping is that economic values are always derived explicitly or implicitly from comparisons of alternative policy options. Therefore, ESS valuation cannot be separated from the policy context. The working assumption is that in due time monetary values will be assigned to all selected ecosystem services according to defined policy scenarios using consistent methodologies.

4.4 Biodiversity and ecosystem services

Recent biodiversity policies introduce the concept of ecosystem services as a means of mainstreaming biodiversity into other policies, notably agriculture, fisheries and forestry. The argument is that these policies are dependent on biodiversity resources and are therefore partly responsible for some of the declines that are observed in biodiversity. The assumption is that the provision of ecosystem services is underpinned by and hence, correlated to biodiversity. As a consequence, maintaining ecosystem services is assumed to contribute to conservation of habitats and species.

Although it is evident the biodiversity underpins ecosystem services, the exact mechanisms remain poorly understood. Studies based on experiments, maps overlaying indicators for biodiversity with indicators for ecosystem services, field observations or meta-analysis of published data often report weak correlations between biodiversity and ecosystem services. The dominance of few species in ecological communities which are consuming and transferring the bulk of the energy and material flows in ecosystems may result in weak correlations between ecosystem services and biodiversity, often taking the form of an asymptotic relation whereby increasing biodiversity does not result in increasing ecosystem functioning once a plateau is reached. As a result, ecosystem service and nature conservation priorities may not always overlap.

5. Conclusions

Europe has launched a new biodiversity strategy, reinforcing the global commitments that European countries and the EU made at Nagoya. The strategy calls for conservation of biodiversity and restoration of ecosystems, and hence of ecosystem services, and sets targets for 2020.

The results of this report and the experiences achieved so far during the PRESS initiative are expected to endorse the methodological development of harmonised spatially explicit assessments of ecosystem services, which will follow the implementation of the post 2010 biodiversity policies at EU, Member State and regional scales. Clearly, an ambitious research agenda is needed to move beyond the preliminary analysis of ecosystem services that is presented in this report.

The policy analysis of potential synergies and conflicts between services showed that results are always scale and site specific and that local management decisions are critically important for enhancing synergies between services. Provisioning services were found to trade off with other cultural or regulating services and the two latter services are therefore often misrepresented in decision making. We conclude that there is a need to include much better regional and local knowledge and differing societal perspectives in policy development and decision making. The concept of ecosystem services may result in over-simplification, especially when data and crucial regional details are hidden due to a high level of aggregation. A more inclusive approach would also require (policy) tools suitable for addressing ecosystem services, mainstreaming them in all relevant policy fields, reconciling different needs for ESS and linking the policy levels to management levels.

There is high potential for integrating ESS into policies and for supporting this with mapping exercises. Stakeholders appreciate the powerful communication opportunities of ecosystem service maps, but several challenges remain. In particular, not all ecosystem services are easily represented on maps risking again the under representation of some services. Methodological challenges also remain on mapping the diverse valuing of the ecosystems in place and time by various groups of people.

The PRESS initiative has shown that research capacity is present in Europe in order to make spatially explicit assessments of ecosystem services with a view of reporting indicators, prioritisation of restoration areas and inclusion into land use planning and cost benefit analysis. The experiences based on three case studies revealed pragmatic mapping approaches which depend essentially on data availability and pre-existing knowledge and adopted research practises that differ for the different environmental disciplines involved.

The forest case study shows that although many resources are available to map forests, it remains challenging to derive a set of forest services based on these maps. Data are often biased towards timber statistics and the role of forest in providing other services than timber such as berries are game and recreational opportunities remains difficult to assess. The water purification case study shows that substantial efforts have gone into modelling the fate and transport of pollutants in the aquatic ecosystems, but the challenge is to translate this knowledge of ecosystem functioning into a set of scalable and harmonized ESS indicators. In particular, the sustainability question needs to be addressed further. What is the maximum self cleaning capacity of systems and what are the negative effects for biodiversity of this threshold remains to be investigated. In both the forest and water quality case study, researchers have been focusing much on continuing ongoing assessments instead of looking at the knowledge from a different angle in order to extract and present information on ecosystem services. For example, by presenting maps of where ecosystem services are produced at what quantities and where ecosystem services are at risk. Mapping recreation services is new and adopts better the concepts of ecosystem services which may facilitate the translation of ESS science to stakeholders. A remaining weak point is the lack of validation of EU wide recreational ESS maps against observations such as visitor statistics.

At present, several ongoing initiatives aim to propose a set of ecosystem service indicators for reporting progress in achieving the new 2020 biodiversity targets of the convention of biodiversity. The development of such indicators is expected to face several challenges: difficulties to measure and map flows (which ecosystem services are in essence), the contribution of natural and man-made inputs into the system, the identification of sustainable thresholds, illustrating and integrating the spectrum of stakeholder values in maps and indicators, and the inclusion of more advanced economic thinking such as knowledge based on economic models accounting for scarcity, demand and interdependencies.

All in all, the policy analysis, the case studies and the issues that this report raises with respect to indicator development and quantitative ecosystem service assessment provide much useful material and constitute a basis for further development with respect to scenario assessment and monetary valuation. In particular, the following steps are considered a way forward endorsing ecosystem services research.

There is a need for the development of hierarchical sets of ecosystem service indicators, following the European SEBI example (streamlining European biodiversity indicators), but geographically explicit and linked to the EU-2020 Biodiversity Strategy.

Ecosystem service maps are ideally framed in the ecosystem services cascade model and require mapping of biophysical flows, as demonstrated in this study but also of mapping demand, uses, benefits and values. Both indicators and maps should be used in order to prioritize areas where restoration of ecosystems and their services yields maximal gains with beneficial effects for local and regional users.

A more model-based approach of mapping ecosystem services will result in a better exploration of scenarios and policy alternatives and, if coupled to the policy assessment, will reveal potential future synergies and conflicts by mapping EU policy domains on EU multiple use maps. Therefore, PRESS will use in a follow up study, this pilot experience to complete a set of examples of how the spatially explicit distribution of ecosystem services is influenced by changing land use and climate through an assessment of scenarios and what is the associated monetary value.

Finally, the PRESS project team calls for the broad collaboration of all stakeholders involved. The policy analysis showed that the concept of ecosystem services can be used to support urgently needed collaborative processes between scientists, stakeholder groups and local citizens in identifying important ecosystem services and evaluating potential conflicts and win-win situations in specific political and socio-economic contexts.

PART 2. TECHNICAL REPORT

This part of the report represents the scientific material upon which the synthesis report is based. It reports in detail the different methodologies that have been used and provides all the background material for readers who prefer a more in-depth analysis of the PRESS results.

Chapter 6 introduces the policy assessment. It examines how the introduction of ecosystem services in the post 2010 biodiversity policy will affect other policies such as agriculture or regional development by assessing synergies and trade-offs between ecosystem services. Both EU and regional policies have been included in this assessment.

Chapter 7 presents three case studies where ecosystem services have been mapped. The maps show the capacity and flow of ecosystem services generated by various ecosystems at various spatial scales. Examples are available for water purification services with a focus on nitrogen pollution, recreation services exploring the capacity of nature to provide recreation and forest services, in particular timber production and management of forest resources,.

Chapter 8 reports on methodological issues in mapping ecosystem services. At present many efforts go to the development of ecosystem services indicators. This section contributes to this debate by examining indicators that reflect ecosystem services adequately and effectively (policy criterion, and which can be put on maps (map-ability).

6. Policy analysis: Potentials and limitations of the concept of ecosystem services in environmental decision making

6.1 Introduction

Policy making concerning ecosystem services (ESS) is extremely difficult for several reasons. First, the concept of ecosystem services is relatively new in environmental policy. Even though it in part addresses several topics already well known, such as agricultural production and water purification, these topics currently are mostly dealt without using the new term and its conceptual context. Decision making on ecosystem services, thus, not only may pose problems of understanding (with strategic and ethical implications), but, more than that, also represents a broad cross-cutting issue with manifold, often non-linear interactions, trade-offs and potential synergies – both with respect to their biophysical aspects as with respect to values and policies (Ring et al. 2010; Rodrigues et al. 2006). Furthermore, ecosystem services do not exist independently from societal processes and decision making. Only what is valued, needed or required by humans can be called an ecosystem service (Jax 2010). In reverse, they are affected by previous decisions and related to several kinds of human uses in the past. Ecosystem service trade-offs arise from management choices made by humans, which can change the type, magnitude, and relative mix of services provided by ecosystems. Trade-offs can be deliberate and conscious, but in many cases they are unintentional, resulting from lack of knowledge or understanding of the interactions between ecosystem services, or a systematic misrepresentation within economic processes or public discourses (e.g. ecosystem services which have no explicit markets are systematically undervalued in decision making; Millennium Ecosystem Assessment, MA 2005). In other words: “a key challenge of ecosystem management is determining how to manage multiple ecosystem services across landscapes. Actions to enhance the supply of some ecosystem services have led to declines in many other ecosystems” (Raudsepp-Hearne et al. 2010). The challenges increase when the decision making level and the level of policy implementation or the level of the actual use of services differ, or when humans on different levels value ecosystem services and trade-offs between them differently, or when values change over time (Vermeulen and Koziell 2002).

These scale issues also arise in connection with spatial explicit information on ESS. For sound decision making it is important to not only take the spatial existence of specific ecosystem services into account, but also the policy level or the scalar dimension of socioeconomic processes and/or political institutions which affect ecosystem services (e.g. global markets, European or national regulations, local or regional perceptions). A further difficulty exists with respect to the spatial dimensions of ecosystem services themselves, namely that production and utilisation of ecosystem services are not necessarily geographically proximal or even at the same spatial scale (e.g. global benefits at the detriment of local people; MA 2005). This is a challenge for management as well as for attempts to represent ecosystem services and their utilisation in spatial terms (Görg and Rauschmayer 2009).

The goal of this policy assessment was to ensure the effective use of ecosystem services mapping exercises by providing an analysis of potential synergies and trade-offs between different ecosystem service types (provisioning, regulating, and cultural services, in the terminology of the MA) and policy measures that affect the provision of ecosystem services. This kind of analysis will enhance the policy relevance of mapping activities concerning the search for better integrated policy strategies. It will align the outcomes of an analysis of the spatial dimensions of ecosystem services with an analysis of the implications of ecosystem-related decision making.

In order to provide information concerning the relevance of ecosystem services for existing and future policies in various fields, events and activities at different levels were organised within the work package. Especially at the EU level, but also at national levels, institutional regulations and strategic approaches in several political sectors were assessed and discussed, concerning the question how ecosystem services are addressed (directly or indirectly). These discussions and the resulting agreements set the broader societal context of ecosystem services use.

To learn more about these processes, a workshop in Brussels was organized, focusing on important levels for policy development and elaboration of policy measures. To also address the level of policy implementation and to complement the EU and national level, in addition key informant interviews and an online survey were carried out in three regions of Europe, in order to strengthen the understanding of relationships between different ecosystem services at the regional level. The regions were Satakunta in Finland, Saxony in Germany, and Silesia in Poland. They all represent areas of multiple uses of natural resources. The empirical material provided a bottom-up perspective, showing the importance that the stakeholders ascribe to different ecosystem services. It was further designed to improve our understanding of synergies and trade-offs between ecosystem services at the regional level, which is the implementation level and thus especially important for direct impacts on ecosystem services. This analysis, thus, adds an important dimension to the process of integrating ecosystem services into different policy levels.

6.2 Approach

6.2.1. Preliminary remarks

Ecosystem services assessments not only affect environmental policy but also influence policies related to agriculture, forestry, infrastructure, rural development, tourism, etc. For this reason the relations and interactions between different ecosystem services connected to these manifold policies must be analysed. Our basic assumption was that, although the expression “ecosystem services” is not currently used in many policy fields, ecosystem services such as the provision of food or clean drinking water, climate regulation or opportunities for recreation are already covered by policy directives and measurements. However, if these services are not recognized explicitly, the risk is rather high that promoting a service which is specifically protected might have negative consequences on other services (e.g. promoting the use of biomass for fuel vs. educational value of wildlands that may be converted for the purpose of biomass production). Making services explicit provides an opportunity to better identify conflicts and synergies between them and rank the importance of services to humanity (or specific groups of human societies).

6.2.2. Key questions

The following research questions were formulated:

- Which ESS are emphasized by decision makers in different contexts (and why)?

- Which ESS (as implicit targets) are already covered by policies/regulations?
- Which trade-offs and synergies exist between ESS?
- How do policy measures affect trade-offs or synergies between ESS?
- How can spatially explicit information be used in decision making concerning ESS?
- Which problems will occur in decision making concerning/using the concept of ESS?

6.2.3. Methodological approaches

Within WP3, several social-science methods were used, namely document analysis, expert interviews, focus group discussions, and an online survey. At the initial stage an analysis of the most relevant documents of the EU helped to get an overview of the extent to which ESS are already implicitly covered by EU policies. This analysis was complemented by six key informant interviews at a sub-national, i.e. policy-implementation, level in three regions: Satakunta in Finland, Saxony in Germany, and Silesia in Poland. The collected material was used for preparing a focus group discussion in Brussels with policy-makers from the European Commission and member states, and an online survey in the three regions mentioned above. In the following the approaches are described in more detail.

Document analysis. To get an overview of the extent to which ESS are already implicitly covered at the EU level and what the most relevant policy regulations are, a document analysis reviewing documents of the European Commission was conducted. This document analysis was restricted to the fields that we perceived as the most relevant for our analysis, namely environmental policy, agriculture and forestry, transport, regional development, and tourism. The goal of the analysis was not to provide an in-depth analysis of the policy fields but to prepare the focus group discussion later in the project. The task was thus to clarify the competencies of EU institutions or member states in certain policy fields and to identify some of the most relevant EU or member state regulations important for the ESS addressed.

Key informant interviews. While the interviews also provided input for the focus group discussion on the EU level, their main purpose was to support the preparation of the online survey. The concept of ecosystem services is meanwhile well known in the scientific world. However, it was not clear to what extent policy makers or planners on regional levels are familiar with the new concept. The interviews were used to find out what kind of language would be useful in a regional online questionnaire. We assumed that even respondents who were not familiar with the concept of ESS could still answer the questionnaire if the concept was properly explained and illustrated with important and well defined ESS. A list of ESS was prepared for the interviews, based on a refined list of ESS from the TEEB project, and sent to the key informants before the interviews, so that the interview partners had some time to prepare for the interview. Further, the interviews served to identify the relevant stakeholders in order to complement the list of potential respondents for the online survey. In order to prepare the focus group discussion, the key informants were asked about the implicit coverage of ESS by current EU, national and regional level policies. The interviews ended with a discussion of example maps showing possibilities of presenting trade-offs on a map, such as trade-offs between the services of water purification and agricultural food production. The maps were used to discuss potentials and limitations of maps for decision makers on a regional level.

In Germany two interviews were conducted, one with a regional planner (regional planners in general were assumed to have a good overview of ESS in their planning region), and one interview with two respondents from the Saxon State Ministry of the Environment and Agriculture involved in landscape planning. In Finland two interviews were conducted with four key informants. One of the interviews was with two persons working in a regional planning authority, while the other interview was held with representatives of regional units of agriculture and forestry producers organisations and the forest owners organisation. In Poland two interviews were conducted, one with a member of the Centre of

Natural Heritage of the Marshal Office of Silesia and one with a member from the spatial planning unit of the Silesian Voivodship Office.
From the information gathered in these key informant interviews a list of 15 ESS with short, easy-to-understand explanations (Annex 1) was developed, on which the questionnaire (see 2.3.4 below) was based.

Focus group discussion. Another important methodological tool was a focus group discussion with members of several DGs of the European Commission and representatives of some EU member states. This event took place on September 13, 2010 in Brussels. The focus group discussion was meant to explore the current implicit and/or explicit inclusion and importance of ecosystems services in various policy fields. Further, synergies and/or conflicts between policies were explored and the potential of maps for decision making was discussed. The focus group method was chosen because it provides an opportunity to discuss questions of understanding of the ESS concept as well as the more explicit questions of trade-offs between different European regulations and the relevance of spatial explicit information and mapping activities for resolving potential conflicts.
Invited participants from the EU level came from DG Environment (biodiversity and water units) and DG Agriculture (forestry and agriculture units). Participants from member states represented the national ministries of the UK, Finland and Poland, covering the fields environment, forestry, agriculture and regional development. Even though other policy fields (and of course nations) are certainly relevant for our questions, the number of participants was restricted to promote more in-depth discussions among the participants. The method is not meant to produce representative results but rather to explore the field.

Online survey. The survey was meant to view ecosystem services in the context of real landscapes, including their uses and benefits obtained from them. It should support a better understanding of synergies and conflicts between ecosystem services in the three abovementioned regions in Germany, Finland and Poland. The questionnaire for the online survey was first prepared in English and then translated into the three languages before submitting it. In total seven questions were asked, including the respondents' organisational background, familiarity with the ESS concept, identification of the ESS missing from the prepared list, a selection of important ESS for the region, the number and kinds of trade-offs and synergies between selected ESS and other ESS, as well as additional comments of any kind. The finalised questionnaire was prepared and sent out to the respondents using the software Webropol (www.webropol.com).
The survey was sent as an e-mail link to 156 persons in Saxony, 148 in Satakunta and 108 in Silesia. The response rates were rather low. We received 16 valid responses from Saxony (two responses were rejected as they showed that the questions had been misunderstood), which gives a response rate of 10.3%. In Satakunta the response rate was 19.6% with 29 respondents, but in Silesia only seven persons responded (response rate 6.5%).
In the following, the three regions in Germany, Finland, and Poland are described. These regions were chosen for the online survey as some information about ESS stakeholders was already available, including a number of contact points for an online survey. Further, all of the three regions are characterised by a high diversity of ESS.
Saxony, in Germany, covers an area of 18 416 km2 and had 4 168 732 residents in 2009. Agriculture, forestry, and fisheries made up only 1%, i.e. 729 Mio. €, of the gross value in Saxony in 2009, and only 2% of the population find employment in these sectors. In contrast, these three sectors cover 85% of the Saxonian territory. The area covered by agriculture makes 56% and the area covered by forestry 27%. Of the 56% of agricultural area only a very low percentage of 3.5 % is used for organic farming. Saxony has 270 FFH areas and 77 bird sanctuaries, which together make up almost 15% of Saxonian

territory. Tourism in Saxony exists but is predominantly characterised by urban tourism to Dresden and Leipzig. However, there are some very attractive and well known landscapes for nature tourism and hiking such as the Elb-Sandsteingebirge, Sächsische Schweiz or the Erzgebirge, which are partly also nature conservation sites, including a National Park (SMUL 2009).

The region of Satakunta, on the west coast of Finland, covers 8 412 km2 (excluding the marine area). In 2010 there were 230 000 inhabitants. More than half of the population lives in the central Pori sub-region. In 2009 the migration pattern was positive, for the first time since the 1970s, but the population trend is decreasing (Satakuntaliitto 2010a; Satakuntaliitto 2010b). In comparative terms, Satakunta is the most industrialised region in Finland. Particularly the metal and machinery industries, wood processing, leather and food production industries are significant. Industry, services for businesses, welfare and recreational services are recognized as the potential growth sectors (Satakuntaliitto 2010c). Primary production is a small sector on the scale of the whole region. In 2007 agriculture produced 1.5% of value added of the region and employed 5.5% of the workforce. Forests, when forestry, wood processing and pulp and paper industry are combined, play a more important role for the region's economy. Value added gained from these activities was 9.3%, while employment was provided for 4.4% of the workforce. Other industries were far more important, though: 24.1% of the value added and 19.5% of the workforce. The region offers a healthy and variable environment. Rural landscapes and nature areas are easily accessible to all citizens. The state of the environment is generally good except for the quality of surface waters. Although primary production (agriculture and forestry) are not economically the most important activities, they are the main forms of land use. Agriculture land covers 20% of the land surface, while forests take over 74%. Built and industrial areas take only 4.5% of the land areas (Statistics Finland 2011; HERTTA 2011).

The Silesian Voivodship, with the capital of Katowice, is located in the south of Poland and occupies an area of 12 333 km2. Its population comprised 4 640 725 people in 2009 and is highly urbanised, with over 3 624 400 people residing in urban areas (Statistics Poland 2009a).

In Silesia, agricultural land covers 646 076 ha, of which 463 371 ha represent arable area (440 ha - ecological arable area). Forest areas occupy another 399 592 ha, while built-up and urbanised areas stretch over 141 196 ha (Statistics Poland 2010). Only 4.3% of the inhabitants are employed in agriculture, forestry and fishing, whereas 37.9% of the population work in industry and construction (Statistics Poland 2009a). 22.1% of the Voivodship area are designated as legally protected areas possessing unique environmental value, out of which 18.4% represent landscape parks, 3% protected landscape areas, and 0.3% natural reserves (Statistics Poland 2009b). Some of the popular touristic destinations in the region are the Beskid mountains in the south of Silesia, providing opportunities for skiing and hiking, as well as the Krakow-Czestochowska Upland (Polish Jura) in the north of the Voivodship (Silesia 2010).

6.3 Results

6.3.1. Important ecosystem services

Ecosystems services are not something given by "nature" (or ecosystems) as such, but they are the consequences of some processes and components which humans consider to be valuable for their needs and desires. They are, in the words of the Millennium Ecosystem Assessment, "the benefits people obtain from ecosystems." That means what counts as an ESS, and which of these are more important than others, is (to a large degree) dependent on human choices and specific societal contexts. In the following the findings from the online survey concerning the prioritisation of ESS by the respondents are presented. However, because this prioritisation might be influenced by the respondents' background and knowledge of the concept of ESS, these details are described first and the survey results are interpreted against this background.

In total 52 people responded to the questionnaire. The 16 responses from Saxony came from the agricultural, forestry, nature conservation, planning and water sectors. The seven responses from Poland came from the sectors of agriculture, education and research, nature conservation and social development. From the Finnish region we received 29 responses from all of the mentioned sectors, except for the social development and the water sectors.

Most of the respondents have their institutional background in nature conservation. However, this category also includes respondents working for organisations involved in the conservation of cultural landscapes, e.g. protected by the UNESCO Man and the Biosphere Programme (MAB), and for environmental protection agencies (e.g. water quality). Otherwise the respondents had very different backgrounds, as shown in Figure 6.1.

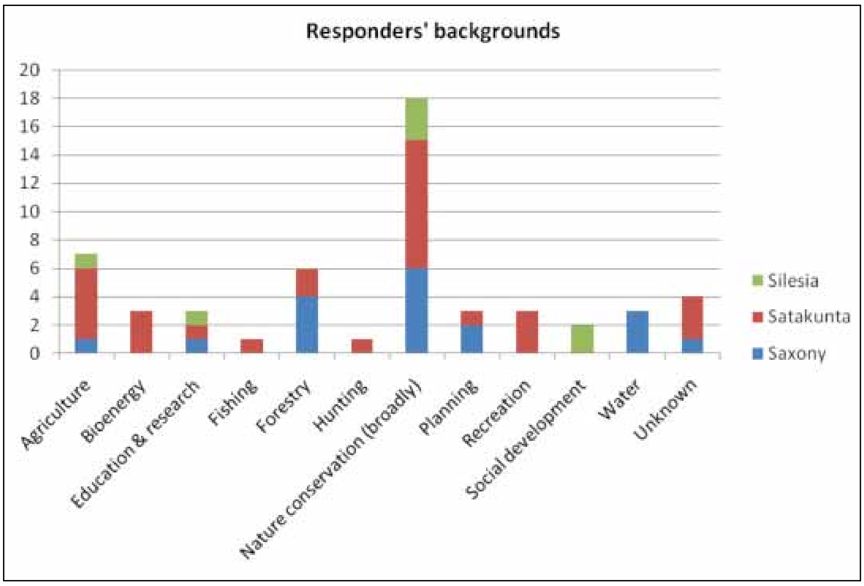


Figure 6.1. Respondents' backgrounds according to regions

Familiarity with the ESS concept varied strongly among the respondents (Figure 6.2). From the respondents in Saxony especially the nature conservation group stated that they have a rather good knowledge of the concept of ESS. Research and water representatives as well as two of the three foresters also perceived themselves to be familiar with the concept. In the other groups familiarity was rather low. In Satakunta, similarly to Saxony, the group of respondents from the agricultural sector demonstrated a medium to low level of familiarity. The same applied to the representatives of the bioenergy sector. The two foresters

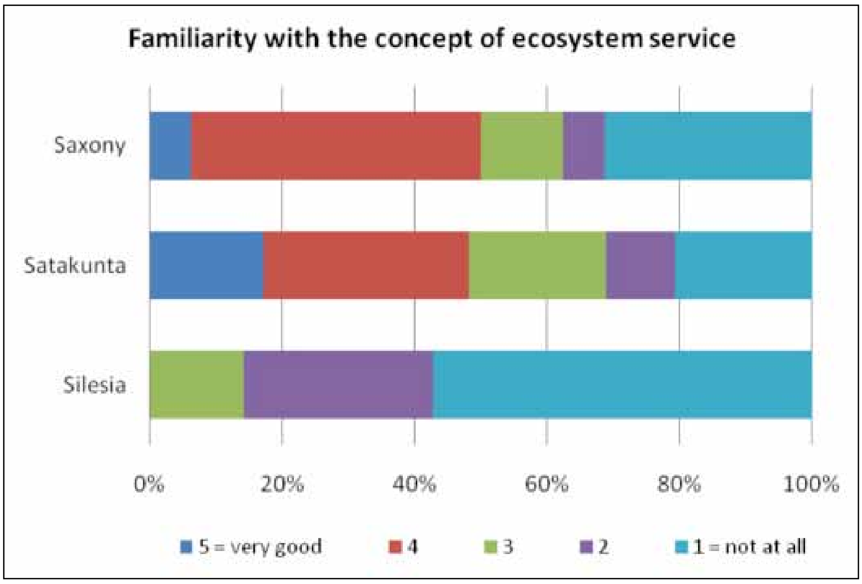


Figure 6.2. The respondents' familiarity with the concept of 'ecosystem services'.

from Satakunta showed high familiarity with the concept, like the respondents involved in planning and tourism. In the nature conservation group the respondents considered themselves quite familiar with the concept, with one exception. Respondents from Silesia overall had low familiarity with the concept, which might be one reason why the response rate was low as well.

The survey asked the respondents to name three ecosystem services the saw as most important in their region. They were also given a possibility to explain their selection. In the following, the respondents' prioritisation of ESS is presented according to the region.

In Saxony the service "agriculturally produced food" was ten times mentioned as important and thus became the most frequently selected ESS. Reasons provided for the selection of the service were its importance for the provision of food for the region, its high productivity and cultural values, and the fact pointed out above, namely that it constitutes the dominant land use.

Timber, tourism and recreation, and erosion prevention were all chosen five times as important services. Explanations for selecting timber were also its dominating land use; after all, 27% of Saxonian territory is covered by forests. Unfortunately, no further explanations were given for the selection of tourism and recreation, and erosion prevention.

Services selected four times as being important were water purification, flood prevention, and protection of species. Clean drinking water was pointed out to be indispensable for human life, natural flood protection reduces the costs for technical flood protection, and flood areas also provide special habitats. Interestingly, although many of the respondents had a background in nature and landscape conservation, the provision of habitats for species conservation did not become the most often selected service, but was only selected four times as being important for the region.

In Satakunta agricultural food production was, like in Saxony, by far selected most often, namely 23 times, not only due to its importance for the region but also because the region is an important link in the national food production of Finland, and agriculture is very typical for the region. Timber was ten times selected as an important service because it represents, like agricultural food production, an important economic sector and employer. As forests cover 74% of the land, and forestry is practised intensively in the region, this choice does not come as a surprise. Timber is followed by tourism and recreation, and biomass for energy production, which were both selected nine times. It is interesting to note that of the respondents who chose biomass energy as an important ecosystem service many emphasised its importance in the future, not necessarily its present importance. Biomass energy has gained a lot of interest in the region, for instance in a recently launched energy strategy for Satakunta. Eight respondents also selected flood prevention, as people in the region are affected by floods and anticipate an increase in their frequency.

The services most often selected as being important in Silesia were flood prevention and air purification. Unfortunately, hardly any explanations were given regarding the choice of the various services. Still, one of the respondents stated that polluted air is a major problem of life quality in Silesia. Agriculturally produced food was, together with tourism and recreation, only the second most selected service. One respondent explained that the region has a high population density, and recreation sites in close distances are very important for the inhabitants of the region. The third place is occupied by water purification and protection of species. The former is a huge problem due to the heavy industrialisation of the region and because the quantity of municipal wastewater is growing.

Taken the weighted results from all the three regions together (Figure 6.3), it becomes evident that the service of agricultural food production clearly dominates the other services. Food production is followed by natural flood protection, based on the experience with catastrophic flood events in the last decade in Saxony and Silesia, and the expectation that their frequency and intensity will increase. The third most important ESS is tourism and recreation, based on its high importance for Silesia. While there are some differences concerning the importance of ESS among the regions, respondents from all three regions agree that the services of aquaculture, biochemicals, medical and genetic resources, aesthetic and spiritual use of landscapes, and the gain of scientific knowledge and education are of very low importance.

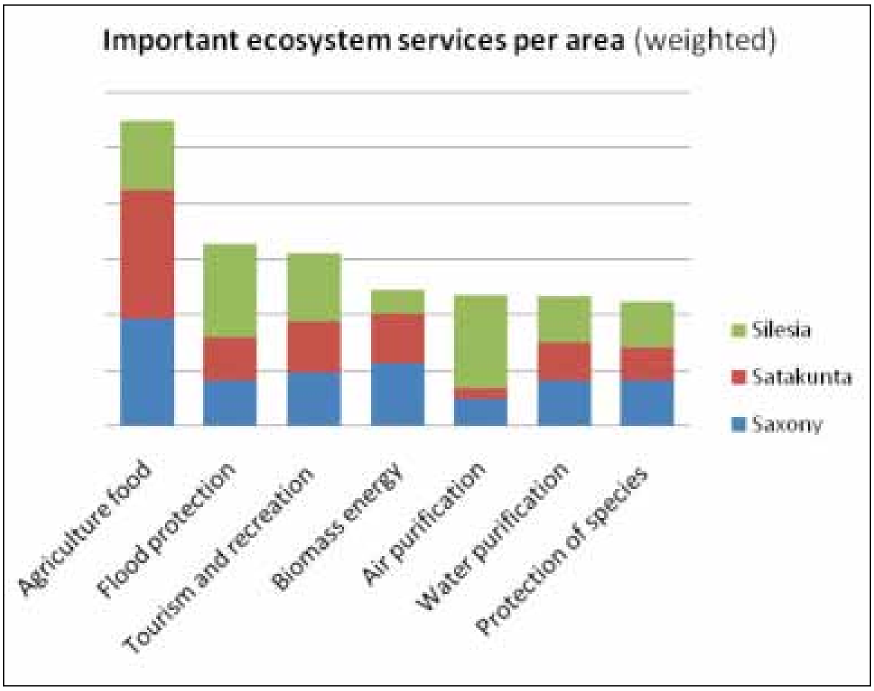


Figure 6.3. Aggregated, weighted number of times the services were selected to be important. The responses are weighted since the numbers of respondents varied considerably across regions.

6.3.2. Ecosystem services covered by policies

The document analysis of EU policy documents, the discussions in the focus group in Brussels, as well as the key informant interviews revealed that, although ESS were not mentioned explicitly as such (with the exception of a few conservation-related documents), many of them were targeted as well as impacted by the policies. In the following, the results of the focus group discussion and the key informant interviews are presented together, highlighting important services from different perspectives.

Agricultural policies. The service of agricultural food production is dominantly addressed by the Common Agricultural Policy (CAP). However, agricultural food production is not the only service addressed implicitly. Other services subject especially to the agri-environmental programmes are erosion prevention, aesthetic and spiritual uses of landscapes, provisioning of water and habitats, genetic resources, as well as tourism and recreation. While the EU level is clearly the dominating policy level for agricultural food production, there are complementing programs on national levels, such as the Joint Task for the Improvement of Agricultural Structures and Coastal Protection (GAK) in Germany and even support programmes on the federal state level.
Examples of conflicts between policies can be seen between the targets of water policies (e.g. WFD) and some targets of agricultural policies (CAP). Increased agricultural production via increased agricultural inputs such as fertilisers can endanger water quality targets.

Water policies. Several EU policies such as the Floods Directive, the Bathing Water Directive or the Water Framework Directive (WFD) address ESS implicitly. Examples given were water purification, water provisioning, flood protection, or tourism, albeit they are not explicitly called “ESS”. In the context of the WFD, ESS become relevant when costs and benefits are thought of in relation to achieving the good status of water. Moreover, the WFD necessitates new land management practices that are beneficial for many ESS. Beyond the EU level, of course, national and federal state concepts, programs and other measures exist that address the above mentioned services, not the least flood protection.

Forestry policies. Forestry policies can mainly be found on the national level, where they usually implicitly address not only timber production, but also tourism and recreation, climate regulation, maintenance of soil fertility, water and habitat provisioning, as well as protection against noise and pollution. As pointed out above, in Satakunta, where forestry is an important economic sector and form of land use, the respondents emphasised the importance of the national sustainable forestry policy and its financing instrument as a means to level and mitigate trade-offs. Although there is no common EU forestry policy, the new EU Forestry Strategy and Action Plan will include the concept of ESS. The example of payment for the ESS “water purification” to forest owners by private companies was given as one option for using the ESS concept.

Biodiversity and nature conservation policies. It comes as no surprise that in biodiversity and nature conservation policies, such as the Natura 2000/FFH or the Birds Directive and especially the new EU-Biodiversity strategy, ecosystem services are prominent.
The policies on EU level are complemented by a number of measures and strategies on national and federal state level, such as national nature protection laws, national biodiversity strategies or the support funding programmes of the Free State of Saxony. In Satakunta a new economic instrument named ‘METSO’ was mentioned as a process to protect forest biodiversity. Under the METSO scheme forest owners can offer parts of their forests with high conservation values as protected areas. If their offer is accepted, they are compensated.

Other policies. Other policy fields mentioned, but not discussed in detail, were energy policies, especially those concerning the use of energy from renewable sources, such as the Directive 2009/28/EC, marine and fisheries policies, as well as strategies and programmes concerning tourism and recreation via rural development, such as the integrated rural development (ILE) or the LEADER program, or national and regional activities.
In Satakunta a regional energy strategy, which is currently being prepared, emphasises bioenergy as an important energy source in the near future. In the regional interviews and the survey it was mentioned that the strategy will make both agriculture and forestry more important sources of biomass energy than they are today.
During the empirical investigation it became clear that while ESS are often addressed implicitly, explicit inclusion in policy formulation and target setting remains an exception and a challenge. This is due to the fact that the precise understanding of ESS is lacking, and participants of the focus group were concerned about conceptual vagueness of the term “ecosystem services”. Additionally, concerns about the lack of knowledge especially about cultural services and non-marketable services were expressed. Further, already the rather limited research activities possible within the project, namely the group discussion and key informant interviews, revealed a rather broad range of conflicts between the policies. This suggests that an in-depth study of the policies is necessary for sound policy formulation.

6.3.3. Trade-offs and synergies between ecosystem services

6.3.3.1. Trade-offs and synergies between ecosystem services from a regional level perspective

While trade-offs and synergies between different uses of ESS and between policies affecting them are rarely visible in policy documents, they became quite obvious in the empirical material. Especially the online survey revealed interesting insights, which are described below first according to the regions and then ecosystem services- wise, in order to compare differences between the regions.
The analysis of trade-offs and synergies is based on the questions of how the selected important ESS affect other services, positively or negatively. Figure 6.4 shows how the important ESS in the columns affect the other ESS listed in the rows. The green colours show positive relationships, with the darker

shade implying strong synergies. The red colours show negative relationships, with the darker shade implying strong trade-offs. White fields indicate no impacts of ESS on each other. However, not all respondents had the same perceptions of synergies and trade-offs. While the colours represent the average of the different responses, boxes marked with one exclamation mark indicate that, despite the resulting colour, some respondents saw a different relationship than the majority of respondents. Boxes marked with two exclamation marks indicate an even stronger controversy among opinions. Differences in perceptions can reflect different knowledge bases but also differing experiences.

	Agriculturally produced food	Timber	Water purification	Flood protection	Erosion prevention and soil fertility	Protection of species	Tourism and recreation
Agriculturally produced food			!		!	!	!
Aquaculture							
Wild food							
Timber						!!	!
Biomass energy							
Biochem., med. and gene resources							
Water purification							
Flood protection						!	
Erosion prevention and soil fertility	!						
Air purification							
Protection of species	!						
Climate regulation		!					
Aesthetic and spiritual							
Tourism and recreation	!	!					
Science and education							

Figure 6.4. Impacts of provisioning of selected ESS (in the columns) on provisioning of other ESS (in the rows) for Saxony. The green and red colours show the average of positive and negative relationships respectively, with more intense colours implying stronger synergies and trade-offs. The boxes containing exclamation marks indicate controversy among opinions within the respective region.

Saxony. It comes as no surprise that the production of agricultural food is perceived as having many negative consequences on other services. In most cases the area for agricultural food production competes with other land uses, such as timber production, floodplains, or nature conservation areas. Problems of water pollution and soil erosion resulting from agriculture are likewise well-known. However, not all respondents automatically assumed negative consequences and pointed out that the consequences for other services depend very much on the type of agriculture. While negative consequences mostly result from conventional and industrial farming, organic farming can have very positive consequences for biodiversity in cultural landscapes such as Saxony. Organic farming can even support soil formation or prevent erosion at the very least. The same applies to synergies with tourism and recreation. Many people enjoy the beauty of structurally diverse cultural landscapes, which can be destroyed by large areas of monocultures.

Timber production has, as expected, much less trade-offs than agriculture, but as some respondents pointed out again, this also depends on the specific forestry practices. E.g. clear-cutting can have negative effects for recreation as it has a very negative impact of the landscape, and of course clear-cutting is strongly negative for climate regulation. While fertile and rich soils are obviously good for agricultural

production, the protection measures for this service can have at least short-term negative consequences for agricultural production if it has to be changed into one with more gentle production methods. Those who selected the service of ecosystems to provide habitats for valued species also pointed out that the service can have negative consequences on agriculture as agricultural land becomes converted into protected areas. Similarly for timber, conservation areas often forbid timber harvest and the larger the conservation areas are, the more the forestry sector is loosing. Like with many other services, respondents who selected tourism and recreation as important, pointed out that whether the consequences of this service are negative of positive depends on the kind of tourism and recreation activities.
While it was selected as an important service, respondents assume that the production of biomass for energy will or could have negative consequences in the future. Apart from competition for land, as well as pollution and soil erosion problems, the consequences for climate regulation are not clear.

Satakunta. Unlike in Saxony, most of the respondents found more synergies between agricultural food production and other ESS; however, as the exclamation marks suggest, not all respondents agreed – pointing to the same trade-offs described for Saxony (Figure 6.4). The views emphasising conflicting aspects of agriculture were presented by nature conservation, forestry and recreational sectors. Agriculture was seen as harmful for biodiversity, but at the same time as creating aesthetic landscapes. It was noted, for instance, that fallow land might have a lower aesthetic value than cultivated fields and that the present practice of keeping animals inside instead of outside at meadows has reduced the aesthetic value of rural landscapes.
Again, unlike in Saxony, the use of timber was seen as rather harmful for many other ESS. A probable explanation is that forests cover most of Satakunta and forestry is practiced on a large scale. Conflicts between timber and other ecosystem services produced by forest ecosystems were perceived by nature conservation and recreation sectors, but also by people working in the forestry sector. Some respondents explained that industrial forestry and especially clear-cut forestry has negative consequences on biodiversity and contributes to the degradation of watersheds and natural flood protection. However, other respondents pointed out that different forestry practices have different influences on erosion and flood protection, yields of berries and mushrooms, cleaning of groundwater, and natural carbon sinks.

	Agriculturally produced food	Timber	Biomass energy	Flood protection	Tourism and recreation
Agriculturally produced food					
Aquaculture					
Wild food		!			
Timber					!!
Biomass energy					
Biochem., med. and gene resources	!!	!			
Water purification	!!				
Flood protection	!!	!!			
Erosion prevention and soil fertility	!!	!!			
Air purification		!			
Protection of species	!!	!	!		
Climate regulation	!!	!			
Aesthetic and spiritual	!!	!			
Tourism and recreation	!!	!			
Science and education					

Figure 6.5. Impacts of provisioning of selected ESS (in the columns) on provisioning of other ESS (in the rows) for Satakunta. The green and red colours show the average of positive and negative relationships respectively, with more intense colours implying stronger synergies and trade-offs. The boxes containing exclamation marks indicate controversy among opinions within the respective region.

Nature conservation was seen as necessary in many comments and as being compromised by important economic uses. One person commented though that protection of some species, such as large carnivores, might reduce ecosystem services because they are dangerous to humans (a wolf pack of approximately ten wolves has recently settled in Eastern Satakunta). One respondent also pointed out that some of the relationships between ecosystem services may vary from location to location. While being harmful for other services in an upstream area, they can be synergetic in a downstream area.

Silesia. In the case of Silesia the interpretation of synergies and trade-offs is much more difficult. First, there were only very few respondents. Apart from that, the data contain an even stronger bias than in the two other cases, as only few sectors were represented by the respondents and the respondents did not provide explanations or examples for the synergies and trade-offs they selected. Therefore, the results from Silesia are not discussed separately but only in the comparative analysis in the following.
However, it is important to mention that some respondents from the online survey and from the key informant interviews pointed out that trade-offs and conflicts between ESS are of minor importance when compared to the conflicts between ESS and other land uses, such as infrastructure developments (e.g. highways).

Aggregated analysis of the findings from the regional level. Overlaps and differences become best visible when responses from each region concerning a single ecosystem service are analysed together. Figure 6.6 shows the impact of agricultural food production on other services in Silesia, Saxony and Satakunta. It is surprising to see that the impact of the service agriculturally produced food on other services is judged very differently. For example, while the respondents in Satakunta see the impact of agriculture on timber to be very synergetic, respondents in Saxony indicate a rather negative impact. The same applies to the impact of agriculture on the service of water purification, although not all respondents in Satakunta valued the impact of agriculturally produced food on water purification positively. Both Silesia and Saxony had been affected by flooding of a catastrophic extent. Areas for natural flood protection are scarce and often occupied by farmers, who hold on to their land, so that the expansion of flood protection areas is difficult. This is only one example of the trade-offs between agriculturally produced food and natural flood protection indicated by respondents in Silesia and Saxony. Although in Satakunta some respondents also realised certain trade-offs, the majority of the respondents saw the relationship as synergetic.

	Silesia	Saxony	Satakunta
	Agriculturally produced food		
Agriculturally produced food			
Aquaculture			
Wild food	!!		
Timber			
Biomass energy	!!		
Biochem., med. and gene resources			!!
Water purification			!!
Flood protection			!!
Erosion prevention and soil fertility		!	
Air purification			!!
Protection of species		!	
Climate regulation			!!
Aesthetic and spiritual			!!
Tourism and recreation		!	!!
Science and education			!!

Figure 6.6. Impacts of agriculturally produced food on other ESS as perceived in all three regions. The green and red colours show the average of positive and negative relationships respectively, with more intense colours implying stronger synergies and trade-offs. The boxes containing exclamation marks indicate controversy among opinions within the respective region.

The respondents of all three areas seemed to agree on the positive effects that agricultural food production has on aesthetic and spiritual use of landscapes and tourism and recreation, the services which are closely linked. This reflects the deep roots that agriculture has in the tradition and self-image of the regions, which are valued positively. This is also reflected in EU policies that aim to protect cultural landscapes via special agricultural practices. However, some respondents also pointed out that industrial farming with large fields and monocultures can destroy the positive picture.

Another example of heterogeneity in perceptions is the influence of natural flood protection on other services, as shown in Figure 6.7. The impact of flood protection on agricultural food production was perceived as ambiguous as the impact on timber, in the light of the competition for land. In the case of timber, it is interesting to note that while in Silesia and Satakunta respondents pointed towards trade-offs, respondents in Saxony saw synergetic effects. While flood protection and agricultural food production by and large exclude each other, flood protection areas can take the form of riparian forests. However, the impact on water purification, erosion prevention, air purification and climate regulation, as well as on the protection of valued species, was identified as consistently positive over the three regions and indicated that this service has a high synergetic potential. This can be an important argument when comparing the costs and benefits of natural and technical flood protection, which have often been ignored up to now. What is not visible in our results but must be taken into account is that flood risk and flood protection vary across regions. Silesia and Saxony have witnessed several disastrous floods while Satakunta, although being a flood-prone area on a Finnish scale, is not even closely exposed to flood risks of the magnitude as in Silesia and Saxony. The same applies to the prospects of natural and technical flood protection.

	Silesia	Saxony	Satakunta
	Flood protection		
Agriculturally produced food	!!		
Aquaculture	!!		
Wild food			
Timber			
Biomass energy	!!		
Biochem., med. and gene resources			
Water purification			
Flood protection			
Erosion prevention and soil fertility			
Air purification			
Protection of species			
Climate regulation	!!		
Aesthetic and spiritual			
Tourism and recreation	!!		
Science and education			

Figure 6.7. Impacts of flood protection on other ESS as perceived in all three regions. The green and red colours show the average of positive and negative relationships respectively, with more intense colours implying stronger synergies and trade-offs. The boxes containing exclamation marks indicate controversy among opinions within the respective region.

The protection of valued species (Figure 6.8) has likewise a lot of synergetic potential. However, like agriculture, nature conservation can be a very exclusive form of land-use, and controversies occur when the protection of species excludes even non-extractive uses. This is especially problematic in such nature conservation sites which protect species very sensitive to disturbance, as they are then out of bounds for humans and are not available anymore for human uses such as recreation. In most cases there is also a conflict between the protection of valued species and agricultural food production; however, as respondents rightly pointed out, without agriculture many cultural landscapes with their specific diversity would not exist and therefore the two services do not necessarily exclude each other. It also depends by and large on the specific agricultural practices, as mentioned above. The same applies for timber.

	Silesia	Saxony	Satakunta
	Protection of valued species		
Agriculturally produced food		!	
Aquaculture	!!		
Wild food			
Timber	!!	!!	
Biomass energy			
Biochem., med. and gene resources			
Water purification			
Flood protection		!	
Erosion prevention and soil fertility			
Air purification			
Protection of species			
Climate regulation			
Aesthetic and spiritual			
Tourism and recreation			
Science and education			

Figure 6.8. Impacts of the protection of valued species on other ESS as perceived in all three regions. The green and red colours show the average of positive and negative relationships respectively, with more intense colours implying stronger synergies and trade-offs. The boxes containing exclamation marks indicate controversy among opinions within the respective region.

The views on synergies and trade-offs between ecosystem services, although being variable between regions and even within them, do have an interesting common pattern. It is that most of the negative relationships pertain to – being either produced or experienced by – provisioning services (from agriculture to biochemical, medicinal and genetic resources in the list presented in Annex 1). Most of the time the red colour is linked to provisioning services in the trade-offs tables presented above. One reason for this could be the fact that provisioning services are utilised by extractive uses of landscapes – and are often practiced as rather exclusive types of land use, where land use conflicts are pre-programmed. Another reason for the high number of trade-offs is that these services are much more visible, while supporting, regulating and cultural services are not always directly detectable – which might also be true for the associated conflicts. An analysis and visualisation of the trade-offs between easily detectable and more abstract ecosystem services is important for mitigation of trade-offs in land-use management, but it is not an easy task.

Apart from a few obvious patterns the analysis mainly supports the fact that it is not at all self-evident how different ESS relate to each other and that generalisations are hard to draw on the European or even national and regional levels. In addition to the reasons pointed out above, there are several more explanations for these differences and similarities in the relations of ESS, ranging from varying perceptions and knowledge of respondents to geographical characteristics of the regions (e.g. climate, soils) and demographic factors (e.g. population density). Further, respondents pointed out that different forms of uses and practices lead to diverging outcomes for other services. For example, organic farming can have very positive consequences for biodiversity in cultural landscapes, as well as for soil formation or erosion prevention. In contrast, conventional (industrial) farming can result in very negative impacts on all of the mentioned ESS. This notion emphasises the need of contextual knowledge and also sensitivity to scale issues: on larger scales certain generalisation may be valid for elucidating synergies and trade-offs, but the final outcomes of relationships between ecosystems services are produced on regional or local levels. This implies that policies that aim to find synergies or at least avoid conflicts between ecosystem services must have a capacity to operate on various scales. The differing perceptions on the trade-offs have yet another implication for policy formulation. They suggest that fair and just policies must go beyond average or most common preferences and take preferences of all stakeholders and societal groups into account.

6.3.3.2. Trade-offs and synergies between ecosystem services from EU and national level perspectives

Participants of the focus group perceived trade-offs and potential conflicts as unavoidable. They identified trade-offs in particular between biodiversity and forestry, between ESS and infrastructure, tensions within water management between navigation and biodiversity, between food production and biomass production for energy, or between biomass production for energy and biodiversity. It was stressed that conflicts between ESS on the local level are often not considered in EU policy making. However, participants emphasised that sound local implementation should be more important than conflicting targets on higher levels (policy formulation level: EU, national). In practical terms, participants mentioned rural development programs and strategic approaches at the local level, including stakeholder consultations to reconcile different targets. Concerning stakeholder involvement at the local level, participants emphasised that it is important to identify ESS important for the local level and find solutions for conflicting targets. In this context, participants also mentioned problematic issues such as the flows of services/benefits and aspects of justice (e.g. between those who help to provide ESS and those who benefit from them). Further, they mentioned the concerns of the conservation community about the utilitarian focus of ESS approaches, which might lead to a neglect of other value categories. A sound science base for political discussion and political impact assessment was deemed necessary, and also as needed to develop a common understanding of ESS. However, the participants also stressed that the political target setting as such remains a political process, which science can inform but not determine.

The analysis of the assembled empirical material shows that the appearance of synergies and/or trade-offs and their relevance for decision-making is strongly dependent on the scale of the discussion (in particular between levels of policy formulation – European and member state level – and levels of policy implementation – mostly regional or local) and on the specific ways in which ecosystems are managed (e.g. different forestry and agricultural practices). This means that policies have a great potential to harmonise trade-offs/conflicts between ESS e.g. by supporting specific management practices. The trade-offs between ecosystems services cannot be resolved in high level policies, but could be substantially supported by 'framework policies' that set explicit requirements for lower level policy implementation to be sensitive to multiple trade-offs between ecosystem services. Furthermore, the high level policies should also allocate means to lower level decision-making for practicing such requirements.

6.3.4. Mapping for decision support

We will now address the question of how spatially explicit information can be used in decision-making concerning ESS, drawing on results from the key informant interviews and the focus group discussion. Example maps indicating trade-offs between the services were shown to the interview partners and participants. These maps were intended as a basis for discussing the potentials and limitations of maps for decision makers on regional, national and EU levels. Both potentials and challenges of maps could be indentified.

Potentials of maps. One frequently mentioned potential of maps was that they are useful in problem identification and framing: they help to identify conflicts and synergies and indicate places or areas where particular ESS or biodiversity aspects are threatened (thresholds e.g. aquatic ecosystem endangered to loose its good ecological status).

Maps are also heuristically useful for initiating discussions about solutions and as visualisations for alternatives (simulations). Some of the participants considered this heuristic value of maps to be even more important than high quantitative details.

Nevertheless, maps are certainly already indispensable instruments in planning activities regarding ESS and biodiversity protection areas, e.g. for minimising conflicts and for developing sound spatial management plans in general. On the regional level, maps would especially be helpful in the case of those ecosystem services which are not yet covered by regional plans, e.g. those related to adaptation to climate change, climate regulation, biomass, etc.

As a further use, maps can show the potential of a spatially explicit landscape and its ecosystems to provide a service. Maps can hint at synergies (or conflicts) with other ESS depending on the specific form and use of an ESS, e.g. different crop and/or farming practices. On the regional level, the areas where actual or potential provision of ESS is high can also be presented in relation to various forms of land use. Many forms of land use are not themselves using ESS, but can importantly support or hinder provisioning of ESS. With respect to the former uses, maps thus can serve as a scientific contribution for improving decision making; they can assist decision makers in identifying potential policy measures, improve targeting of measures and demonstrate/evaluate benefits of policy measures in relation to costs. Finally, there is a pedagogic value of maps: they can explain the relevance of biodiversity and ESS to the public.

Challenges with using maps of ecosystem services. Maps can be an important tool for supporting policies on protecting and managing ESS and biodiversity. However, they should be used carefully and consciously, as there are a couple of challenges and potential problems with respect to their use.

In many cases non-marketable ESS (cultural and regulating ESS) are much less visible and sound information on them is scarce. This requires extra effort in future analysis and mapping. Also, not all ESS can be presented easily (if at all) on maps. Temporality of events is a challenge for mapping; for example: how can seasonal events (or demands) be presented on a map?

Even if ESS can be mapped, the scale of ESS maps and the scale of decision making are not necessarily identical. The same applies to the scales and boundaries of administrative units (borders) and ecological units (e.g. ecosystem delimited by watersheds). As already mentioned above, there are tensions between different policy levels, as the level of policy formulation (EU and member states, also based on maps on EU or national level) does not fit well with the implementation level (local-regional).

A further problem may arise with respect to the costs of producing appropriate ESS maps: even detailed maps are often contested from the local level as still inaccurate. A high level of detail and accuracy in maps is therefore necessary, however, very cost intensive. Further, it would be necessary to bring together all data, scientific expertise and modelling work that provide spatial information about ESS, to provide maps as a basis for decision making. This is very cost intensive as well. A solution to the cost factor would be to map e.g. only the most important drivers and ESS. However, the challenge would be to develop criteria to select and visualise such drivers and services and possibly find proxies that relate to other services which are not displayed.

Beyond technical and financial problems a caveat mentioned was that the identification of problematic areas on a map can result in stigmatisation of regions which appear to provide only few ESS. Another challenge emerges from the possibility that ESS maps may indicate areas where exploitation can be increased (e.g. where rivers have the capacity to retain more nutrients). The possibility of such interpretations of maps, which may even be counterproductive with regard to protecting ESS and biodiversity, should be kept in mind and possibly accounted for by accompanying comments.

On the same line with undesired “side effects”, it should be taken into account that, as maps usually show only partial areas, impacts of ecosystem services presented on the map on other services outside the map might be overlooked (“transregional effects”), which may even lead into questions of social and/or international justice. The same applies to the temporal scale. Maps only depict one point in time or an average.

Furthermore, the maps that are presented to decision makers are often kept simple in order to really be understandable and useful. However, often the issues at stake are much more complex and maps oversimplify and even trivialize problems.

All of the problems and challenges above should not be read as an argument against the use of ESS maps for decision making but rather for their careful and conscious use. There is a need for much additional scientific work on how to improve the use of maps in decision making, in particular on how to use them in a discoursive setting and not as an "objective" scientific fact that requires no interpretation.

6.4 Conclusions

The analysis of EU policy documents, focus group discussions, and interviews revealed that there is high potential for integrating ESS into policies and for supporting this with mapping exercises. Even though ESS are hardly mentioned explicitly, many regulations (e.g. the EU Water Framework Directive) and policies implicitly refer to them, or are relevant for them. While many tensions between different governance levels of policy formulation (EU and national states) vs. the implementation level (local-regional) were detected, the identification of trade-offs and synergies between policies turned out to be too complex, to be tackled by our project thoroughly. A more refined in-depth policy analysis in this respect is therefore highly recommended and a desideratum for the future.

The analysis of trade-offs and synergies between ecosystems services themselves, however, clearly shows that the concept of ESS bears the risk of over-simplification, especially when data and crucial regional details are hidden due to a high level of aggregation. Many conflicts and synergies depend on the forms and uses of the ESS, as explained at the example of agricultural practices, where practices such as organic farming can reduce conflicts and increase synergetic effects.

Further, many conflicts and synergies only become apparent at the regional level, and therefore the rich regional and local knowledge and differing societal perspectives, as demonstrated in the online survey, need to be included in policy development and decision making. This would greatly contribute to designing policies which are sensitive to scale issues and to the differences in practices of ESS use. This also includes non-marketable and non-mappable ESS, which are otherwise at risk to be overseen. Based on our findings, we can support the argument that in particular provisioning services, often traded by markets, cause trade-offs with other services, whereas cultural or regulating services are often misrepresented in decision making. While it was not an object of this study, one topic that came up several times regards the conflicts between ESS and other land uses. Some respondents even pointed out that the conflicts and trade-offs between ESS are rather unimportant in comparison to the conflicts between e.g. ESS and infrastructure development.

An inclusive approach would, however, pose the need to develop a common understanding of ESS, make agreements and define targets in the context when many different actors, their perceptions and interests are involved. This is especially difficult in the case of land use, as land is usually a scarce resource and conflicts are likely to increase. By emphasising the need for a common understanding, we do not mean that there must be a common „one fits all" – definition of ESS. On the contrary, definitions will need to be targeted for specific audiences, depending on the specific complexity required for specific purposes (Fisher et al. 2009), and thus be different for communicating the general idea that nature provides multiple services to humanity or when aiming at specific accounting or planning processes.

An inclusive approach would also require (policy) tools suitable for addressing ESS, mainstreaming them in all relevant policy fields, reconciling different needs for ESS and linking the policy levels to management levels. Thought would also need to be given to proper compensation strategies to those who greatly contribute to the support of ecosystems to provide ecosystem services, e.g. as done in organic farming. Likewise, the polluter-pays-principle would need to be extended to cover the range of ecosystem services. Last but not least, there is still the challenge to ensure that the inclusion of the concept of ESS in policy making leads to positive consequences not only for human beings but also for the environment, and that other (non-utilitarian) values of nature and biodiversity are not neglected.

While this study clearly demonstrates that the concept of ESS and the mapping of ESS have a great potential to support policy making, the number of crucial open questions and concerns also shows that further research is urgently needed.

7. Spatial assessment of ecosystem services

The inclusion of ecosystem services in biodiversity and land use planning policies requires more detailed knowledge of the services that are produced by ecosystems, hereby recognizing that several ecosystem services including cultural and regulating services may be delivered by semi-natural and agricultural ecosystems as well. Such an assessment necessitates the development of ecosystem services maps and models in order to estimate where ecosystem services are produced, to quantify the changes in service provision over time, to describe the production of ecosystem services as a function of patterns of land use, climate and environmental variation.

This section demonstrates the results of biophysical mapping of ecosystem services. In particular, three case studies served as examples of how to combine existing information present in statistics and models to derive new maps of ecosystem services. Special attention was given to multi-scale assessments. The policy analysis showed already that synergies and trade-offs between ecosystem services may differ depending on spatial scale. It is expected that biophysical assessments at different spatial scale will reveal similar changes in synergies and trade-offs. Model and maps at high spatial resolution for local areas may contain useful information on the management of ecosystems which lead to local synergies in service provision. Such detail is often not available for regional assessments. In this study, we have attempted to address differences between mapping approaches that relate to scale.

The first case study addressed the cleaning capacity of aquatic ecosystems as they remove pollutants and contribute to the supply of fresh and clean water. In particular, nitrogen is used as an indicator. A second case study has elaborated on methodologies for mapping the potential of nature to provide recreation to humans. A third case study has focused on the role of forests in providing services.

7.1 Multi-scale assessment of water purification services by ecosystems

7.1.1. Introduction

Ecosystems contribute in supplying clean water by absorbing or filtering pollutants such as heavy metals, excess nutrients, and pesticides processed as water moves through wetland areas, rivers and streams, floodplains and riparian zones, estuaries and coastal marshes. This is an important ecosystem service because it reduces water treatment costs, increases the aesthetics of water and supports native species that people like to view or harvest (Loomis et al. 2000).

The water purification service delivered by wet ecosystems is based on particular physical and biological ecosystem properties and functions. Two ingredients are necessary: prolonged residence time of the water and a rich and healthy aquatic biodiversity, able to process pollutants as they perform their function in the aquatic food web.

At the scale of aquatic ecosystems, wetlands, lakes and slow running rivers and estuaries are characterized by extended residence time of water which enables micro-organisms such as bacteria and plankton and macrophytes such as reed to take up, process and mineralize pollutants, organic matter or excess nutrients. Floodplains, estuarine and coastal marshes and vegetation buffer strips enhance this functioning by storing water temporarily or by obstructing increased runoff and hence, increase the time during which organisms can degrade pollutants. At the catchment scale, forests, grasslands and riparian areas buffer the runoff of precipitated water and the unwanted chemicals that are transported to surface waters preventing downstream nutrient enrichment and pollution. At the larger river basin scale pollutants are retained and processed over tens of years in soils and aquifers before entering surface waters.

The collective functioning of these different ecosystems at various spatial and temporal scales leads to the immobilization of pollutants or in some case the removal from the environment which is considered an important service. In turn, water purification results in the provision of clean water that can serve multiple uses: habitat for species and different uses for humans.

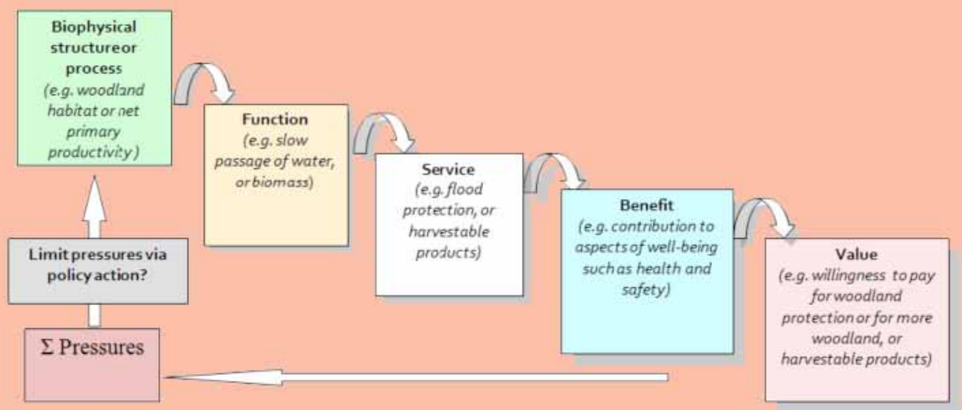
Retention of nutrients and pesticides does not always result in benefits. Eutrophication of lakes and coastal zones is a clear example that an ecosystem service can become a disservice if anthropogenic pressure is too high. Under specific aquatic conditions, enriched levels of nutrients are used by algae for photosynthesis, causing blooms that disrupt normal ecosystem functioning through the consumption of the available oxygen or via the production of harmful toxins. A second example is the production of N_2O gas from the denitrification process; a particularly important nitrogen retention mechanism in wetlands but also active in other ecosystems. The dominant gas produced in wetlands as a result of denitrification is N_2, which is abundantly present in the atmosphere. Nitrous oxide N_2O, however, is a greenhouse gas. Consequently, high production of this gas following high anthropogenic nitrogen inputs result in a cost, not in a benefit. For now, these two trade-offs have not been considered in this assessment but could be considered in a follow up study.

In this case study, we applied different modeling approaches in order to map water purification as ecosystem service at different spatial scales. In particular, we address the retention of nitrogen in floodplains and soils of river basins at local scale, the retention of nitrogen in surface waters at catchment, national and European scale and the retention of organic matter at European scale.

In several of these cases, the ecosystem service cascade model was used as a frame for mapping (Box 2). As such, maps are provided that assess the capacity of systems to provide a service and the concomitant service and benefits flow.

Box 2. Ecosystem service cascades as a frame for mapping ecosystem services

A way of representing the logic that underlies the ecosystem service paradigm and the debates that have developed around it is shown in the figure below (Hains-Young and Potchin 2010). The diagram makes a distinction between ecological structures and processes created or generated by living organisms and the benefits that people eventually derive. In the real world the links are not as simple and linear as this. However, the key point is that there is a kind of cascade linking the two ends of a 'production chain'.



Defining ecosystem functions, services and benefits, and the context for CICES (Haines-Young and Potschin, 2010).
The cascade model contains also the notion of stocks and flows. Layke (2009) defines stocks of ecosystem services as the capacity of an ecosystem to deliver a service while the flow corresponds to the benefits people receive. Stocks may be expressed in total size area or the total biomass whereas the associated ecosystem service flow or output must have units per time period.
The capacity of an ecosystem to provide a flow is not necessarily measured in hectares or ton since the capacity does not only contain a quantity aspect but also a quality aspect. Given the quantity, an ecosystem system may provide more output if it is in a healthy state. As a result, the capacity of such system to produce services will be higher. Ecosystems in a healthy state are considered resilient systems which are able to recover after disturbance and they are characterized by high species diversity and a balanced trophic community. We have used this cascade model for framing the indicators that we developed for mapping ecosystem services. Ideally, ecosystem services are modeled following the cascade from the left to the right. At least, indicators are developed capturing both the biodiversity and ecosystem stocks that generate the services and the final benefits as flows of goods and services.
Applied on water purification as an ecosystem service, this report presents maps of ecosystem service indicators that measure the capacity of wet ecosystems to retain nutrients and pollutants as well as the associated flow of services and benefits in terms of the amount of pollutants removed and the effect on water quality.

7.1.2. Mapping methodology and study sites

The focus of this report is largely on the retention and removal of nitrogen from surface waters. The removal of nitrogen through denitrification takes place in wet ecosystems (Figure 7.1). Both terrestrial as aquatic ecosystems contribute to nitrogen retention. Wet soil systems, floodplains and wetlands receive nitrogen via diffuse sources such as atmospheric deposition or the application of fertilizer in agricultural areas. Nitrogen leaks into the soils and further into the aquifers where denitrification takes place. Eventually, the remaining nitrogen reaches rivers and is transported downstream. Rivers, streams and lakes remove dissolved nitrogen from the surface water by plant uptake and denitrification, hereafter collectively called in-stream retention. This ecosystem service, provided by freshwater ecosystems, contributes to maintaining or improving downstream water quality. Ultimately, the supply of clear surface water provides several benefits such as water for drinking and recreation but also for maintaining economic activities as agriculture and industry.

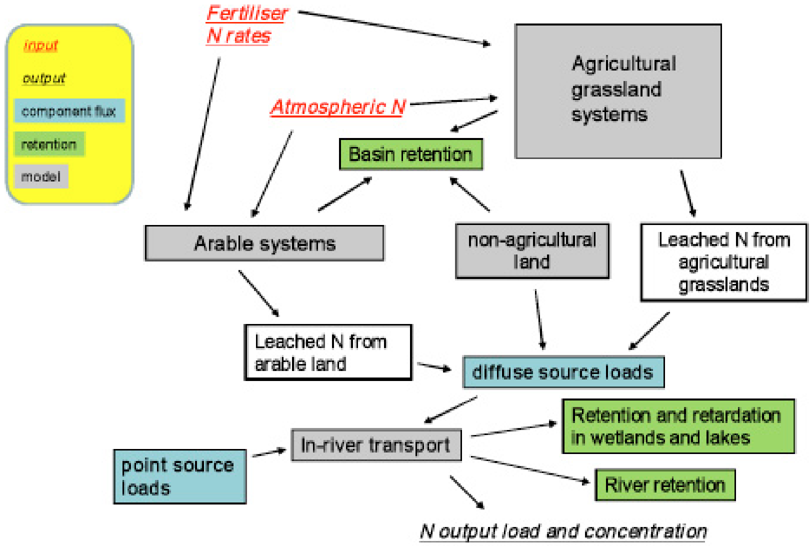


Figure 7.1. Conceptual diagram of nitrogen cycling in a river basin from a hydrological perspective.

Here we summarize the results of local, regional and continental case studies examining in more depth the nitrogen retention processes that take place in the terrestrial and aquatic habitats of the river basin. In particular we propose methodologies for mapping potential and actual nitrogen retention in floodplains along river Elbe (Germany) and soils of the River Ouse catchment (UK). Subsequently, the role of rivers, lakes and streams is considered at different spatial scales with maps illustrating the retention of nitrogen.

A second substance that was considered is organic matter. Organic matter can increase the extent and risk of eutrophication and oxygen depletion which dramatically reduce the use of water for biodiversity conservation, fisheries, recreation, and water supply. A similar mapping approach was used to estimate the removal of organic matter from European water bodies by natural processes and the water quality improvements that result from this.

7.1.2.1. Study sites

Retention maps were made for 6 study sites covering local to continental spatial scales (Table 7.1).

Table 7.1. Study sites included in this study assorted by ecosystem and scale.

Scale	Soil	Floodplains	River networks
Local		N retention in Elbe floodplains (DE); UNESCO Biosphere Reserve Middle Elbe	
Catchment	N retention in River Ouse (UK)		N retention in River Ouse (UK)
River basin			N retention in the Elbe river basin (DE)
Country			N retention in Finland
Continental			N retention in Europe BOD retention in Europe

N: Nitrogen retention; BOD: Biological Oxygen Demand

7.1.2.2. Methods

Mapping soil denitrification. The approach adopted to assess the denitrification in soils of the Ouse river catchments was made specific for three broad categories of land-use: non-agricultural land, agricultural grassland and arable land. Land-use was defined using CEH landcover map (LCM2000) available at 25m spatial resolution, and the Defra Agricultural Census from 2004 available on a 2km × 2km grid.

Denitrification values in non-agricultural land are low. A constant value for denitrification was used based on the wide range of observations reported from worldwide studies (Barton et al. 1999). Rates for agricultural grassland were calculated using the NCYCLE model (Scholefield, 1991). Assumptions about grassland system and fertilisation levels were made based on existing methods (Hutchins et al. 2010a). Values of nitrate from atmospheric deposition were calculated for each sub-catchment based on a 5×5 km resolution gridded dataset based on a modelled interpolation from 32 UK monitoring stations (Fowler et al. 2005).

For arable land, the model used well-founded concepts (Boyer et al. 2006) to simulate topsoil denitrification. Coefficients defining the van't Hoff temperature relationship are taken from mean values of a review of model structures and their parameterisation (Heinen, 2006). The model is of the form used in the GLEAMS and EPIC models:

$$DN_{soil} = NO_3\left[1 - \exp\left(-w\lambda_T C_{org} t\right)\right]$$

where DN_{soil} = monthly soil denitrification rate N (kg ha^{-1}); NO_3 = monthly available soil nitrate-N (kg ha^{-1}); w = dimensionless reduction factor for water content (w = 0 at field capacity, w =1 at saturation); λ = van't Hoff expression representing exponential increase in denitrification with temperature = $Q10^{(T-Tref)/10}$ (where Q10 = 2.28; Tref =21 and T = observed soil temperature); C_{org} = topsoil organic carbon (%); t = 1 (i.e. a single monthly time step).

A fundamental component to calculating denitrification is determination of monthly available soil nitrate. Manure and atmospheric contributions to soil nitrate were taken from the catchment-specific values in the existing soil nitrate model (Hutchins et al. 2010a). Likewise fast mineralisation, determined from crop residual nitrogen, was determined on an HRU-specific basis (Hutchins et al. 2010a). Slow mineralisation of recalcitrant organic matter was calculated using soil properties based on HOST classification (at a 1 km^2 resolution) (Boorman et al. 1995). Organic carbon content, for quantifying slow mineralisation was also derived using this source of soil information.

The modifiers to denitrification rate based on temperature, water content and organic carbon were calculated as follows. A mean monthly reduction factor on denitrification due to water content was assigned on an HRU basis. All 21 gauging stations in the catchment have a PROPWET value (proportion of time when soil moisture deficits are less than 6mm) which can be used as an index of soil wetness. In the current application all 21 values were used although there is not much change on a small spatial scale, values being in part derived from MORECS 40 x 40 km cells. After soils become "wet" it is assumed that the reduction factor changes from 0 to a maximum value over a 30 day period. The maximum value was determined from soil properties of the dominant soil HOST class in each of the 21 sub-catchments, being the ratio of water content at field capacity to water content at saturation. Likewise the reverse change is seen at the same rate at the end of the "wet" period. Mean monthly values of soil temperature in the topsoil were derived (Green and Harding 1979). Organic carbon content was calculated as above.

Mapping nitrogen retention in flood plains. The study areas addressing nitrogen retention in floodplains are situated in the UNESCO Biosphere Reserve Middle Elbe. Lödderitzer Forst has a size of about 1000 ha and is characterized as a largely alluvial forest, which is separated in a flooded (400 ha) and non-flooded (600ha) part by a constructed dyke. A second study site Wulfener Bruch is characterised as a drained depression of about 1000 ha, which is situated in the former floodplain in a distant of about 6 km from the Elbe River itself.

Following (Maltby et al. 1996; Maltby et al. 2006; Brinson 1993; Brinson 1996; Matlby 2009)) the study site was divided into areas of similar environmental characteristics (HGMU). The results of these projects have led to a Wetland Evaluation Decision Support System (Wedss) which mainly gives qualitative information about the assessment of nitrogen retention in wetlands. To come to a quantitative assessment we additionally used estimation values of nitrogen retention potentials as reviewed in the literature. Denitrification in wetlands averages a conservative estimation value of 100kg N ha-1 yr^1), which probably underestimates but definitely does not overestimate the N-Retention potential in wetlands. This conservative value was then modified taking into account the size of the flooded area, the duration of floods and the nutrient loads in the river (Noe and Hupp 2009) yielding possible nitrogen retention values per HGMU varying from 50 to 250kg N yr^1. By taking into account the area of the several HGMUs it is possible to come to nitrogen retention potentials for single HGMUs.

Mapping nitrogen retention in river networks. Fractional nutrient removal in river networks is determined by the strength of biological processes relative to hydrological conditions (residence time, discharge, width, volume). The product of the in-stream retention efficiency and the total nitrogen river loading yields the total amount of nitrogen that is retained per unit time.

The removal of nitrogen by ecosystems can be modelled using a simple mass balance. The mass balance used to calculate changes in the nitrogen stock in a river segment equals the inflow of nitrogen due to upstream and basin loading to the outflows resulting from downstream transport and retention by uptake and denitrification. Based on this mass balance, the in-stream retention, the total amount of nitrogen removed by surface water and the associated changes in water quality were mapped. The effect of in-stream retention on water quality, expressed as the ratio between nitrogen concentration based on nominal model runs and nitrogen concentration calculated for zero-retention model runs according to $[1 - C/C_0]$ where C is the nitrogen concentration with in-stream retention and C_0 without in-stream retention. This set of indicators in used for methodological comparisons between the study areas included in this case study. Five areas are selected to map nitrogen services provided by surface waters: Europe, Finland, the Elbe river basin, the Ouse catchment (UK) and the Parthe Catchment (DE) (table). For each assessment, different models have been applied.

The European wide assessment was calculated using the model GREEN (Geospatial Regression Equation for European Nutrient losses) (Grizzetti et al. 2005; Grizzetti and Bouraoui, 2006; Grizzetti et al. 2008; Bouraoui et al. 2009). GREEN calculates a nitrogen budget for about 33 thousand sub-catchments based on nitrogen inputs from diffuse sources and point sources as well as upstream loading.

The nitrogen assessment for Finland is based on the application of the N_EXRET model (Lepistö et al. 2001, 2006) for 30 river basins.

The Elbe case study builds on results compiled during the Elbe-DSS project (Berlekamp et al. 2007; Lautenbach et al. 2009). The results have been calculated using an integrated model that consists of MONERIS, GREAT-ER and LFBilanz. This model system was used to calculate nitrogen surplus on agricultural land, nitrogen input into the river system as well as nitrogen retention in the catchment and in the river system.

The Ouse catchment case study applied an empirical model based on a world-wide database of observations (Seitzinger et al. 2002) to calculate river retention of nitrate on a reach-by-reach basis. Here, the percentage of nitrogen removed is related to the hydraulic load of the river reach.

Mapping the retention of organic carbon in river networks. As a measure of organic matter concentration, the 5-day biological oxygen demand (BOD_5) was modeled using the GWAVA model. Biochemical oxygen demand or BOD is the amount of dissolved oxygen needed by aerobic biological organisms in a body of water to break down organic material over a period of time. BOD is frequently used to measure the degree of organic pollution of water.

GWAVA is a model for prediction of water resources scarcity at continental and global scales. It was developed by Meigh et al. (1999) with funding from the UK Department for International Development. GWAVA estimates water scarcity on a cell-by-cell basis by comparing modelled river flows with modelled human demand for water. GWAVA has been further developed to include a water quality module (GWAVA-WQ). GWAVA-WQ produces monthly gridded maps of 5-arc-minute resolution of BOD_5 levels across Europe. It does this by modelling levels and pathways of BOD_5 across Europe from its sources (households, paved surfaces, industry, agriculture) to rivers, lakes, and wetlands (Dumont et al. 2010).

Three specific indicators are mapped to illustrate the spatial distribution of organic matter services: the natural aquatic BOD_5 retention, the amount of BOD_5 removed by natural aquatic retention processes and the percentage BOD_5 level reduction due to natural aquatic retention. This was calculated as $[1 - C/C_0]$. Here C is the average BOD_5 level (kg m^{-3}) modelled for 2000, and C_0 is the average modelled BOD_5 level in 2000 if there would have been no natural aquatic retention.

7.1.3. Results

7.1.3.1 Nitrogen retention in soils

Soil denitrification appears to be more significant (relative to other fluxes) in the more upland parts of the catchment (Figure 7.2) though absolute values of the flux are higher in the lowland areas where inputs of nitrogen are higher. The relationships between denitrification and geographical factors were investigated and illustrated in the cases of downstream river network length and altitude (Figures 4b). The scatter in these relationships is a consequence of geographic variability in soil characteristics such as wetness, as well as nitrogen inputs.

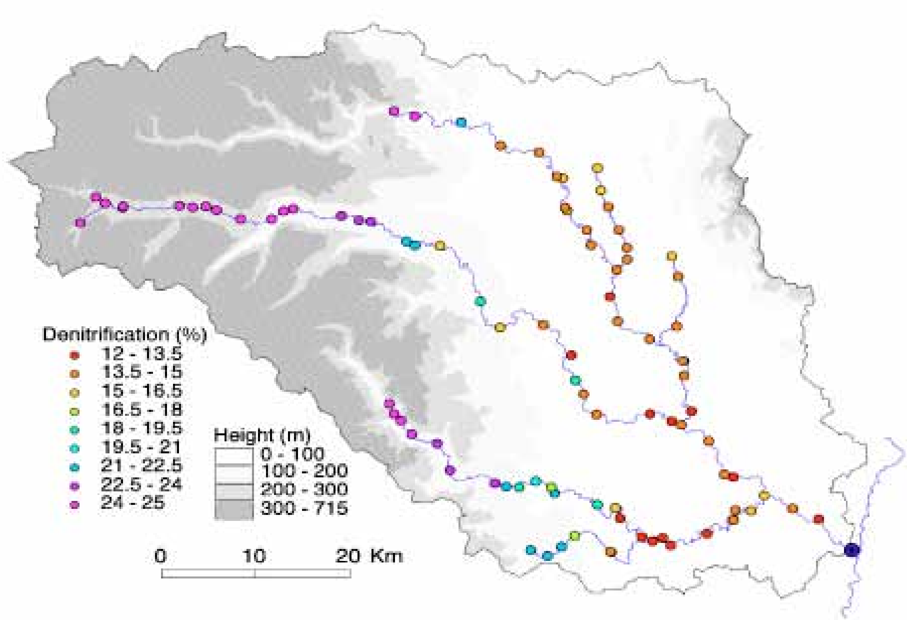


Figure 7.2. Soil denitrification as a percentage of losses from the soil in the sub-catchment areas draining to each point (where losses from soil are denitrification plus leaching, i.e. not including plant uptake and soil storage).

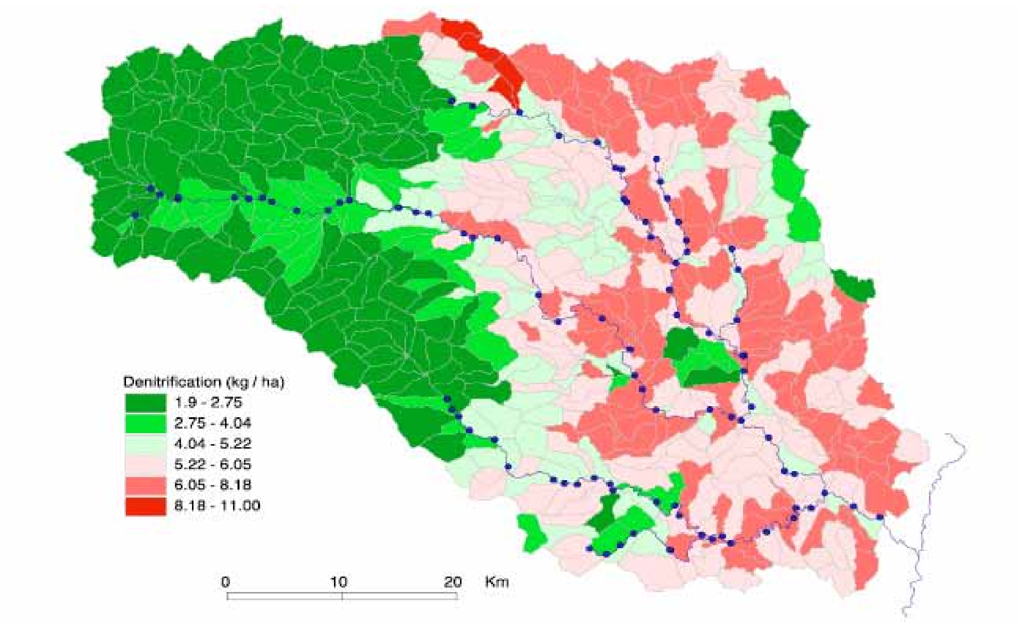


Figure 7.3. Annual soil denitrification by hydrological response unit (HRU).

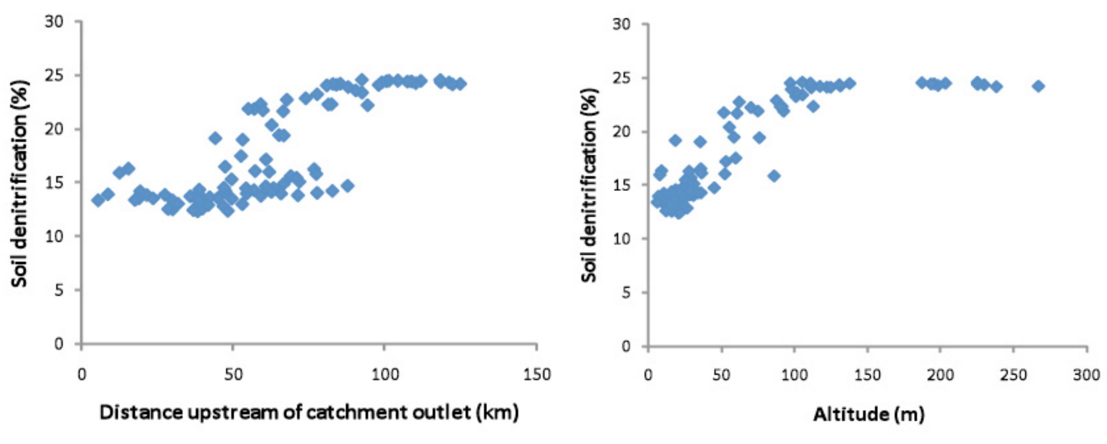


Figure 7.4. Sub-catchment-wide soil denitrification as a percentage of losses from soil plotted against attributes of the sub-catchment hydrological outlet.

7.1.3.2 Nitrogen retention in floodplains

For the 'Lödderitzer Forst' possible nitrogen retention potentials ranged from 0.01 to 8.96 ton N $year^{1}$ for the status quo (SQ) and from 0.01 to 24.2 ton N $year^{1}$ for the scenario option (SC) (Figure 7.5). In total, the Vorland retains about 38 ton N $year^{1}$ and the Hinterland about 96t N $year^{1}$. That makes an increase of N-retention potential through the dyke replacement of about 40% within the 'Lödderitzer Forst', which reflects the additional size of floodplain area and the conversion from farmland into alluvial forest which can be achieved by the dyke replacement.

The range of possible nitrogen retention in the 'Wulfener Bruch' does not vary between the status quo and the planned management option per HGMU (SQ/SC 0.01 to 52.7 ton N $year^{1}$). But changes in the nitrogen retention caused by a possible re-wetting are obvious by comparing the total nitrogen retention potential for the whole study site. For the status quo total retention value is about 120 ton N $year^{1}$ and for the scenario option about 152 ton N $year^{1}$. This increase is caused by the conversion of farmland (RF=0) into grassland (RF=1.5) which comprises an area of about 157 ha. Since the 'Wulfener Bruch' is only connected to the river via groundwater nitrogen retention only marginally benefits river water quality directly.

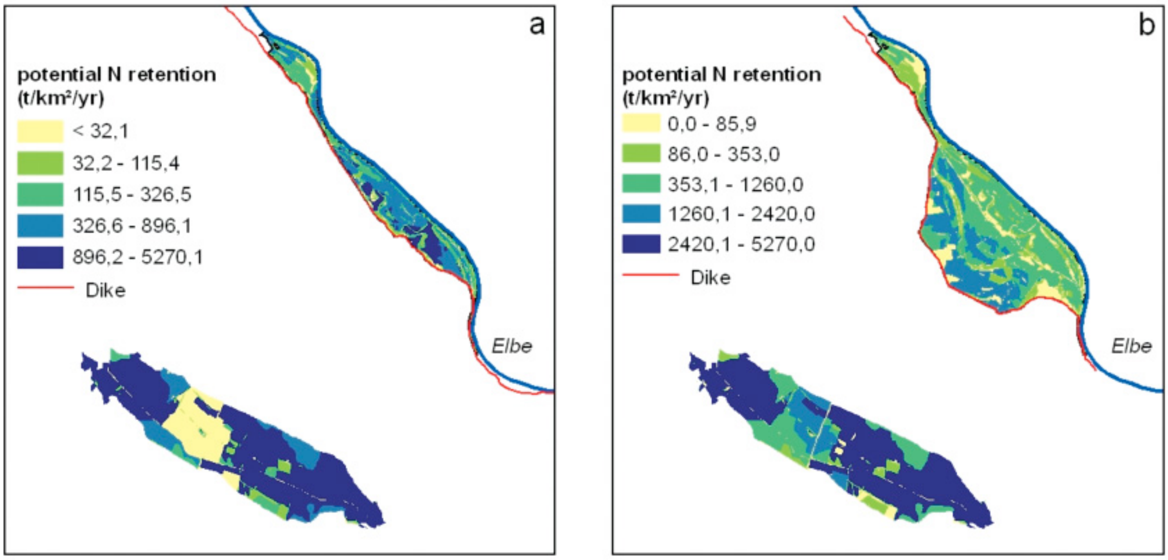


Figure 7.5. Potential nitrogen retention in the floodplains 'Lödderitzer Forst' and 'Wulfener Bruch' according to dyke displacement. The left figure shows the current potential nitrogen retention while the right figure shows the potential nitrogen retention after a potential dyke shift.

The spatial differences between distinct geomorphological units can also be mapped. However, the results still need careful interpretations and the consideration of constraining factors. So it should be kept in mind that the used estimation values are only ranges and that their assignment is strongly connected to expert knowledge. It is assumed for example that the denitrification process makes up 60-90% of the total nitrogen retention in floodplains (Jansson et al. 1994; Byström 1998). Thus, only this process has been considered for the presented results. Further it has been supposed that nutrient retention in general is substantial dependent on the hydrological situation in floodplains - but at present, measured information about the size of the flooded area as well as the flood duration only locally exist. Due to their strong influence on the overall retention process we came up with indirect appraisals via soil type and groundwater level to overcome this limitation. Beyond it, we only know very marginally how changes in the hydrological system, caused by different management options, and changes from farmland into alluvial forest or into grassland will affect ecosystem functioning. For the time being, we just can assume that the functional relationship will stay the same as before the modification. We further cannot take into account how resilient a modified landscape could be and how long such a re-development would take.

However, the presented data and results give a good overview about ranges of a possible nitrogen retention potential for the investigated floodplain sections. Beyond it, if the used basic data are available it is possible to use this method also for other floodplain sections or even in other floodplains. In the EVALUWET-project the method was tested in 5 European countries. Within the project the results are validated with measured data, which is recommended in general too.

7.1.3.3 Nitrogen retention in river networks

Nitrogen services provided by surface waters were mapped using three spatial indicators: the capacity of rivers to retain nitrogen, expressed as a proportion of potential input, the realized nitrogen removal in units of ton per year and the effect of in-stream retention on water quality.

Nitrogen retention at European scale. Based on the pan-European assessment using the GREEN model, European rivers retain annually (based on the data for 2000) 1.5 million ton nitrogen. This value represents 24% of the catchment-to-river flux, after retention of nitrogen in the soil and aquifers of the river basins (Figure 7.6). In sum, European rivers and streams clean up every year an amount of nitrogen equivalent to all the point sources emitting into the European river network. Retention varies between 0 and 20% for different sub catchments (Figure 7.7).

The removal of nitrogen is calculated as the product of retention capacity and nitrogen input. As rivers get loaded with nitrogen from different sources, the removal of nitrogen increases which explains the downstream increment on the maps (Figure 7.7).

The budget of Figure 7.6 depicts only partially the capacity of rivers to retain nitrogen because it does not take into account in an explicit manner the retention of nitrogen arriving from upstream catchments. Headwaters only receive nitrogen from point and diffuse sources and the part that is not retained discharges into medium and large rivers loading the downstream river network progressively with nitrogen.

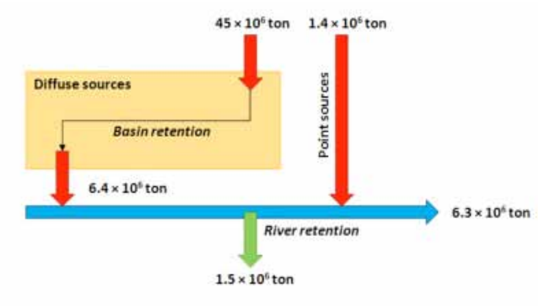
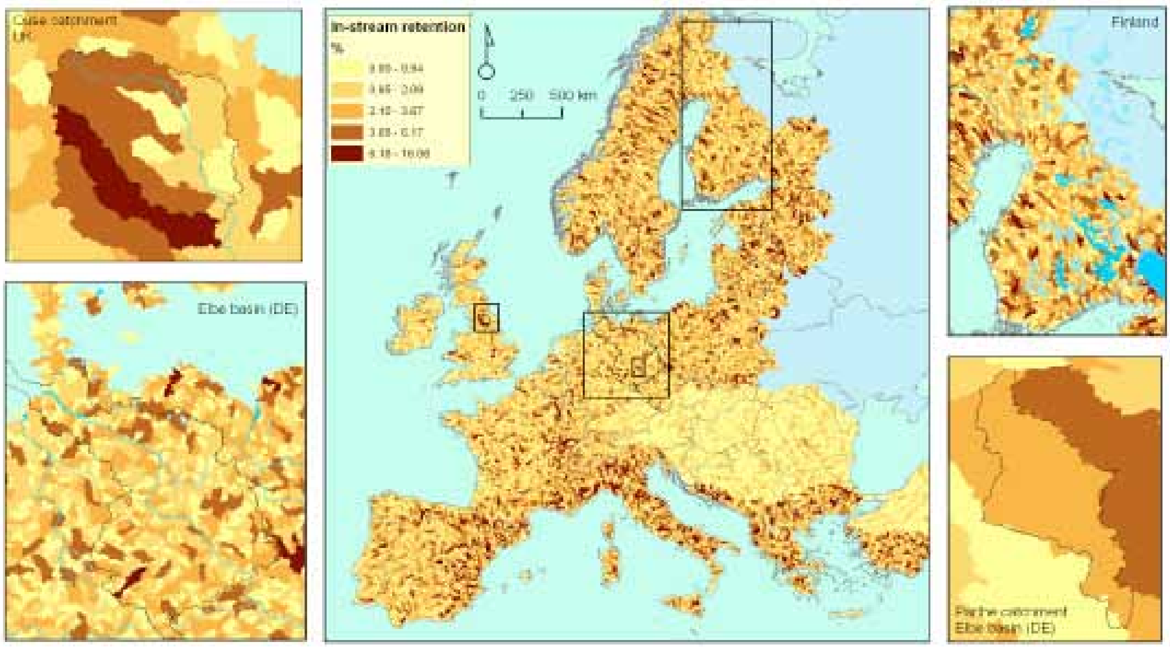


Figure 7.6. Nitrogen budget for the European river network broken down over point source inputs, diffuse inputs, catchment-to-river input, river retention and final loading at the outlet. Data based on the GREEN model for 2000.



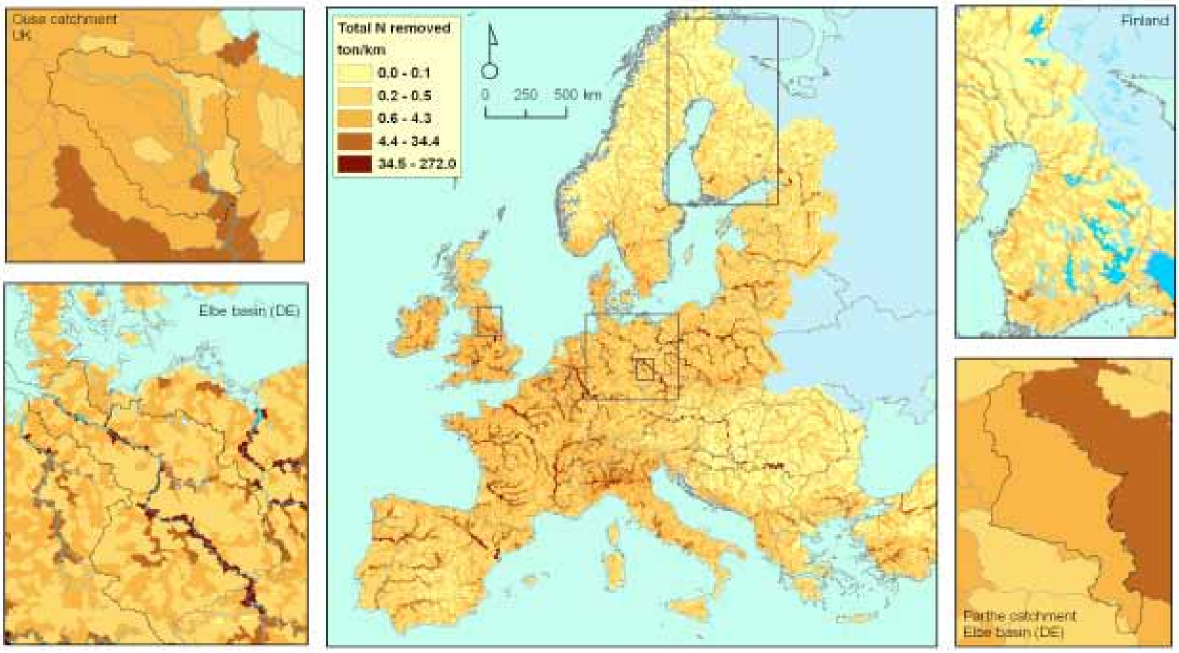
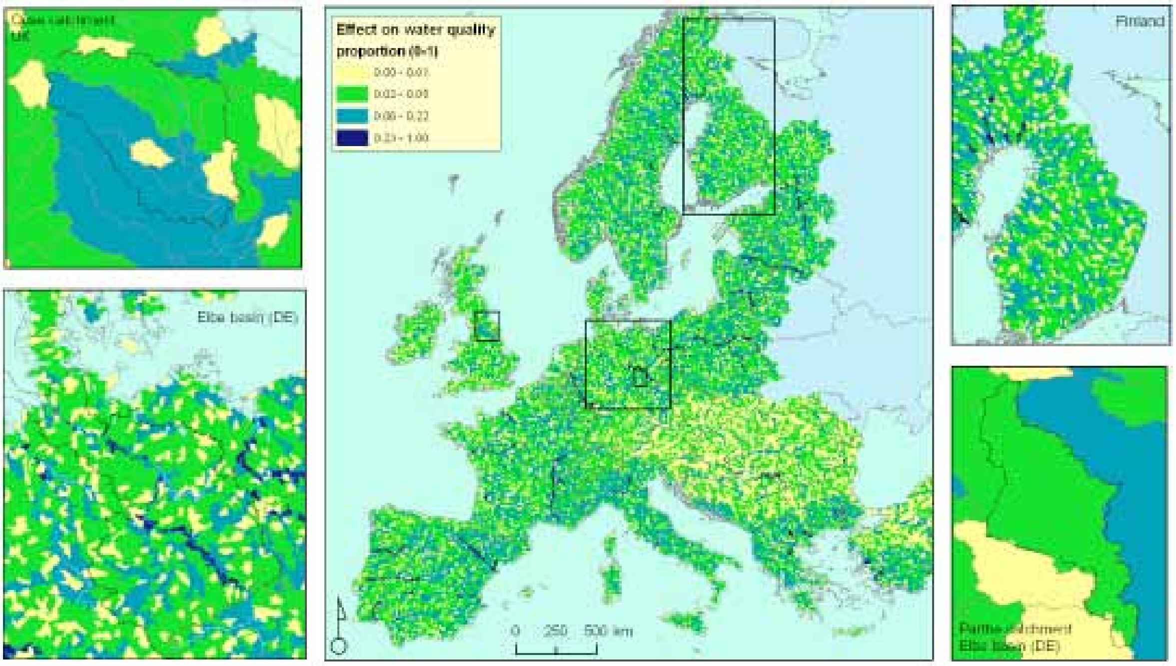


Figure 7.7. Spatial variation in in-stream nitrogen retention (top) and nitrogen removal (bottom) in Europe per sub catchment. Zoom areas for the different study areas (Ouse, UK; Elbe, DE; Parthe, DE). All results based on the GREEN model for 2000.



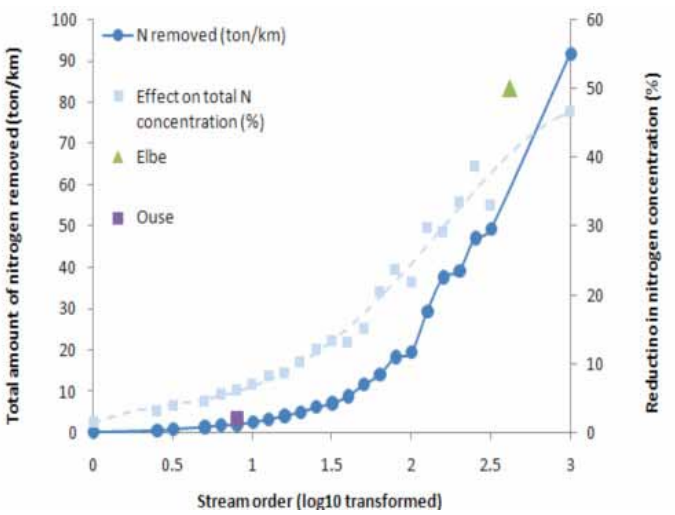


Figure 7.8. Downstream accumulation of nitrogen services. The map shows the reduction in nitrogen concentration as a result of in-stream retention. The reduction is presented as a percentage between 0 and 1. Average results are plotted on a graph as a function of stream order. The total amount of nitrogen removed annually increases with stream order. Similarly, the reduction of total nitrogen concentration in rivers due to river retention increases with stream order. Stream order was calculated using the Shreve number which assigns a value of 1 to headwaters, a value of 2 to the receiving catchment, and so on. For comparison, also the coordinates for the Ouse and the Elbe at catchment outlet, based on different catchment based modeling approaches, are plotted.

As a result, the role of nitrogen retention as ecosystem service increases in importance in downstream direction. Figure 7.8 plots the total amount of nitrogen removed per km river stretch against the stream order (log10 transformed Shreve number). Mountain rivers and headwaters, which have a low stream order pass most of the received nitrogen to downstream catchments but as nitrogen loading increases in downstream direction, the role of retention gains importance. The total amount of nitrogen that is removed per unit length of river stretch increases from nearly zero in headwaters to 90 ton per km nearby the outlet where rivers discharge into estuaries or deltas.

The benefits of nitrogen services are further demonstrated by considering the effect of in-stream retention on water quality as measured by total nitrogen concentration. GREEN was used to simulate the transport of nitrogen and the resulting total nitrogen concentrations in absence of river retention. These results were subsequently evaluated against a nominal model run. A model run without in-stream nitrogen retention results in total nitrogen concentrations that are on average 5% higher than nitrogen concentrations based on a simulation including retention. The effect of in-stream retention on total nitrogen concentration increases with stream order (Figure 7.8), following the pattern of nitrogen removal. In downstream catchments, retention of nitrogen by the river network results in total N concentrations that are between 10 and 50% lower than under the assumption of zero retention. The effect is clearly visible on a European map of sub catchments (Figure 7.8). River retention improves water quality in downstream direction with improvements of less than 1% in head waters, of between 1 and 10% in medium river catchments and of between 10 and 50% in catchments of large streams.

Nitrogen retention in Finland. A nitrogen budget including retention by surface waters was calculated for Finland using the N_EXRET model. The N mass balance, based on the calculation of N retention in river basins, suggested that up to 68% of the total N input was retained, with a mean of 22%. The lowest retention values (0-10%) were detected in coastal basins with practically no lakes (<2%). In all the large watersheds with highest lake percentages (>10%), N retention was high (36-61%) due to longer residence times (Figure 7.9). Also the relative N retention compared with N total input was higher in those watersheds (lakes >10%) than in watersheds with low lake proportion. Lake percentage clearly had an impact on N retention, explaining between 60 and 66% of the variability of N retention, depending on whether a linear or quadratic relationship was fitted (Figure 7.9).

Concerning management of N fluxes, reduction of N loads is more important in small coastal river basins – where agricultural areas are typical - than in upstream catchments. Excess N in lakes is transported downwards to the larger river basins, however, thus affecting to eutrophication of the estuaries (Lepistö et al. 2006).

Lakes were effective nitrogen sinks due to two major processes: denitrification and sedimentation. Retention rates for lakes also compared well with paleolimnological data for the large Lake Päijänne during the industrial phase (Itkonen et al. 1999). In the national N mass balance, annually 38 000 ton N year^{1} (32%) was estimated as lake retention and 4000 ton N year^{1} (3%) as retention in peat lands (Table 7.2). The rest of 77 000 ton N year^{1} (65%) was estimated to transport to estuaries. Lake retention in Finland dominates, although average lake coverage (10%) is significantly lower compared to average peat land coverage (Lepistö et al. 2006). One of the key questions is what will happen to N retention processes when N fluxes may considerably increase in the future climate.

These data were subsequently compared with output provided for the European assessment. The Finish budget was based on data for the period 1993-1998 while GREEN used a budget for 2000. There are substantial differences in input data which seem to be caused by underestimated nitrogen sources by the GREEN model. More importantly with respect to ecosystem services is the difference in retention. The national assessment yielded retention of 35% while GREEN assumed an average retention of 12%.

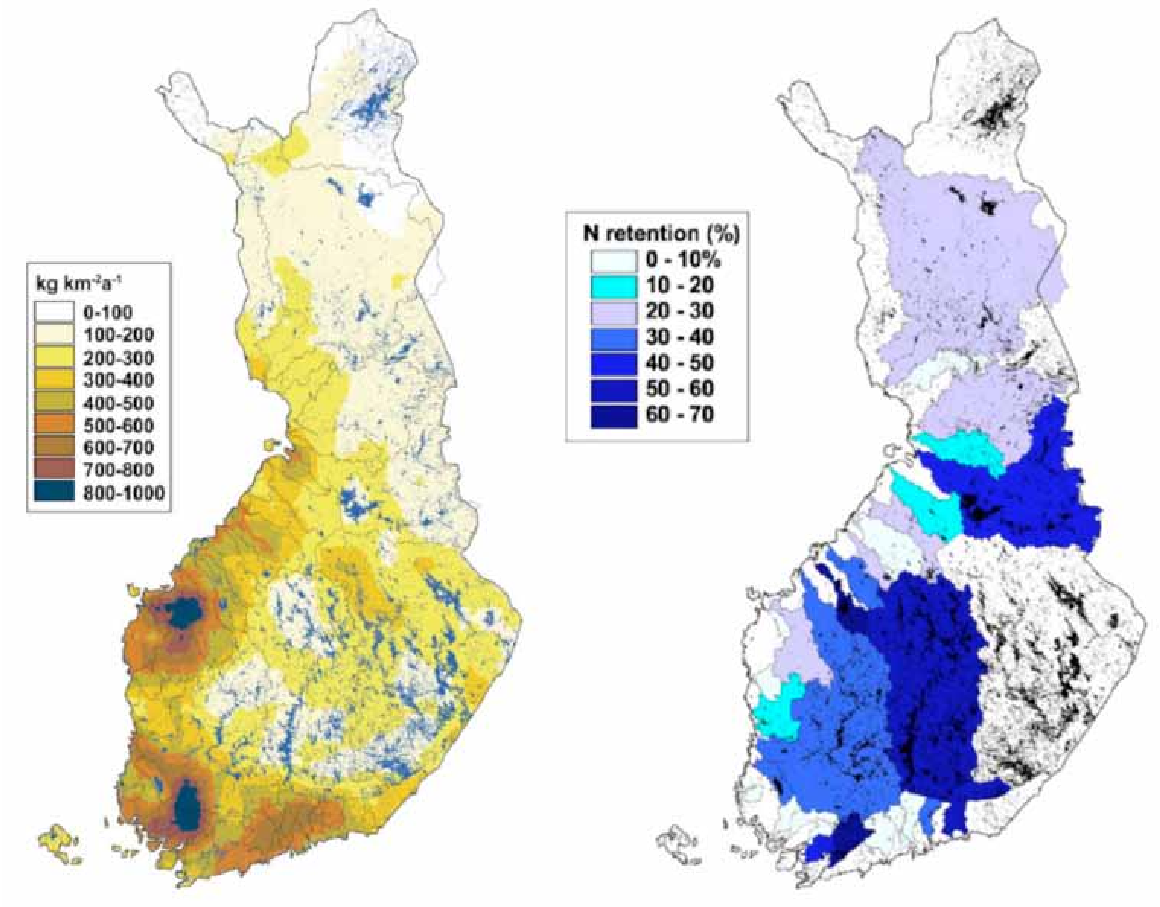


Figure 7.9. Total nitrogen loading of surface waters and in-stream nitrogen retention for Finland.

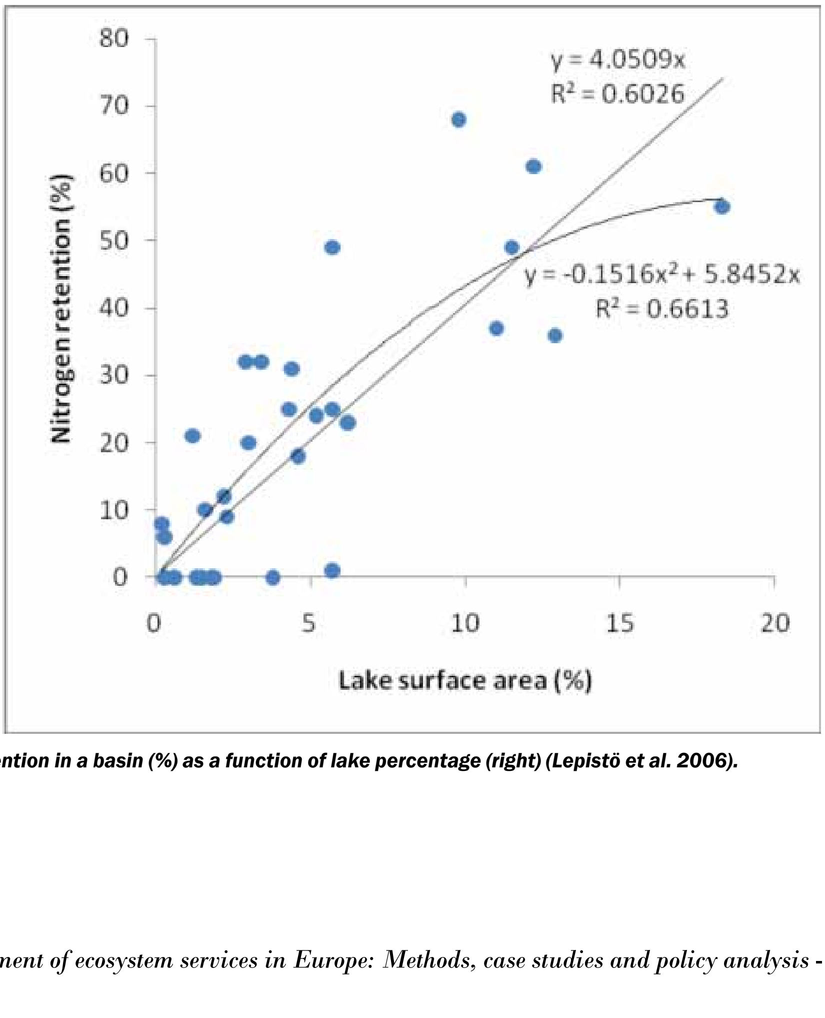


Figure 7.10. N retention in a basin (%) as a function of lake percentage (right) (Lepistö et al. 2006).

Table 7.2. Comparison between N-budgets based on GREEN and N_EXTET for Finland.

Nitrogen budget (ton year^{-1})	N_EXTET based results	GREEN based results
Total diffuse sources on the catchment		572 481
Catchment to stream fluxes:		
Agriculture	45 000	
Forestry	11 000	
Build up area	2 500	
Peat harvesting	1 000	
Background	32 000	
N deposition	11 000	
Total catchment to stream flux	102 500	75 330
Total point sources	16 500	3 134
Lake and peat land retention	42 000	
River retention		9 987

At European scale, the role of lakes and peat-lands in retaining and processing nitrogen is underestimated. Lakes are regulators of nitrogen. They increase considerably the residence time of water which is on its way to the sea. In turn, this increases the time available for uptake, sedimentation and denitrification processes.

Nitrogen retention in the Elbe basin. In-stream nitrogen retention in the Elbe basin ranges from 0 to 14 ton per stretch per year with a clear skewed distribution with a large number of stretches with low retention values. Highest values are observed in lakes due to the high residence time and low flow velocity.

The effect of in-stream retention on concentration is presented in Figure 7.11. Retention in surface waters result in considerable decreases of nitrogen concentration, especially in areas with lakes. In addition, important concentration decreases occur in smaller channels. If model runs with and without in-stream nitrogen retention are compared, the accumulated effect of in-stream retention can be estimated (Figure 7.11). Relative changes in concentration compared to a reference situation without any in-stream retention (Figure 7.11) go up to a reduction by 85% in some of the lakes but stay below 20% for river stretches.

When applied for the Elbe basin only (Table 7.3) the nitrogen fluxes as calculated by GREEN compare well with the nitrogen fluxes estimated by the MONERIS model. River retention was estimated lower than GREEN by the MONERIS model while the GREEN assumed more retention in the basin resulting in a lower catchment to stream flux. Of greater importance are the differences that arise from using a more detailed river network. The river network that was used by the GREEN model to route water from headwater catchments to the sea had a total length of almost 9000 km. It is based on the CCM River and Catchment Database, which covers the entire European continent. The river network used in the MONERIS model is much finer and sums up to nearly 43000 km, almost 5 times the size of the European based network. As a result, averaged removal of nitrogen, standardized per km river stretch, is considerable lower for the Elbe when MONERIS is used (Table 7.3). This difference is crucially important for monetary valuation when values are expressed on a per km basis. Models that conserve mass on a catchment basis by balancing inputs with outputs are useful for assessing water purification services but care needs to be taken when downscaling the catchment based values for assessing biophysical flows and for assigning monetary values on a per km basis. Models using coarse resolution river networks will evidently result in higher per km values. Put simply, the value of large rivers will be over estimated at the cost of small sized streams and water courses.

Table 7.3. Comparison between N-budgets based on GREEN and MONERIS for the Elbe river basin (10^3 ton).

Nitrogen budget (ton year-1)	GREEN based results	MONERIS based results
Total diffuse sources	1 100	748
Total point sources	26	23
Catchment to stream flux	92	105
River retention	58	48
Length of the river network (km)	8 990	42 742
Average nitrogen removal (ton per km)	6.4	1.1

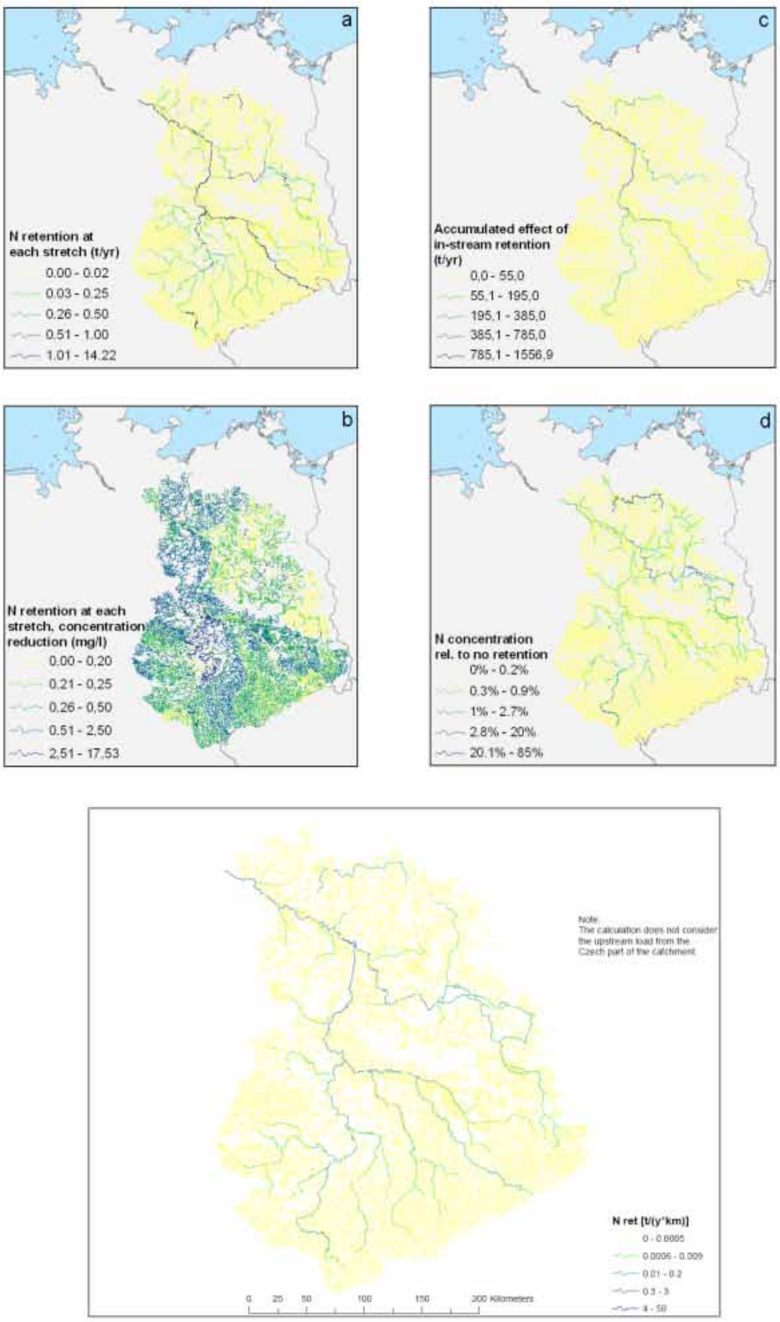


Figure 7.11. Nitrogen retention in the river system of the Elbe basin. The maps show the nitrogen retention in each stretch of the river system expressed as the load (a), the concentration effect of that load reduction (b) as well as the accumulative effect of the load reduction by in stream nitrogen removal (c) as well as for the accumulative relative reduction of nitrogen concentrations (d). A final map (e) plotted the nitrogen removal per km. For the interpretation one should keep in mind, that only the German part of the Elbe has been modeled, the Czech part (around 1/3 of the basin) is missing.

Nitrogen retention in the River Ouse catchment. A mass balance for nitrate sources and sinks for the Ouse catchment outlet was calculated (Table 7.4). The values are representative of the period 1997-2003. The balance between inputs and the diffuse source loads to rivers represents plant uptake and soil storage. Nitrogen retention in river channels was calculated on a reach-by-reach basis (189 reaches) for each of 93 spatially distinct inputs of nitrogen (tributaries and effluents) identified in the catchment. On a reach-specific basis, the QUESTOR model was used to estimate daily water flows and non-linear hydraulic relationships between flow and velocity (Leopold and Maddock, 1953). These, in conjunction with estimates of river widths taken from UK national river habitat surveys (Raven et al. 1998) allowed the calculation of travel times and depths (and hence nitrate retention) at different rates of flow deemed to be of interest. Retention at mean flow is quoted in Table 7.4. In calculating the load exported at the catchment outlet it is assumed that in-river nitrate sources (e.g. via nitrification) and other sinks are in balance. Nitrogen retention per km for sub-catchment stretches of river was calculated and compared with estimates from GREEN. This was undertaken for 14 sub-catchment units delineated by GREEN for the Ouse to investigate spatial differences within the catchment. In general, river retention fluxes calculated by the two approaches compare well (Figure 7.12). Largest differences were apparent in the lower reaches where GREEN simulates higher rates of retention. With GREEN, variability in ton km-1 N retention is closely related to catchment area. The CEH model also captures local variability in hydraulic load. The differences apparent for the smallest catchments are explained by the two models using slightly different river networks. In terms of river retention such differences are only manifested at the smallest scales. For the entire Ouse, total modelled river lengths are very similar (Table 7.4).

Mass balances for the entire catchment and three sub basins were compared (Table 7.4). Most of the differences between the two models appear to be attributable to greater basin retention by GREEN (e.g. for the Ure sub catchment). Note that there is some spatial mismatch between the spatial units of assessment. For the Ouse catchment outlet the overall impact denitrification processes have, as sinks of nitrate, on river concentrations is also shown and compared with GREEN estimates (Table 7.4).

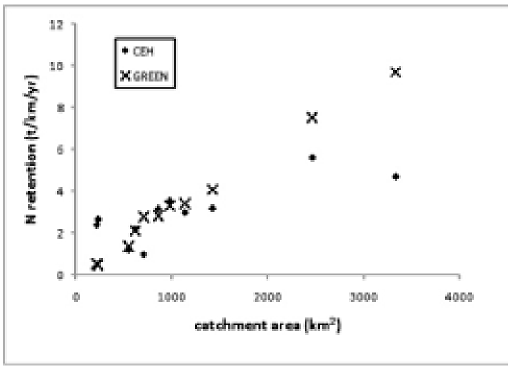
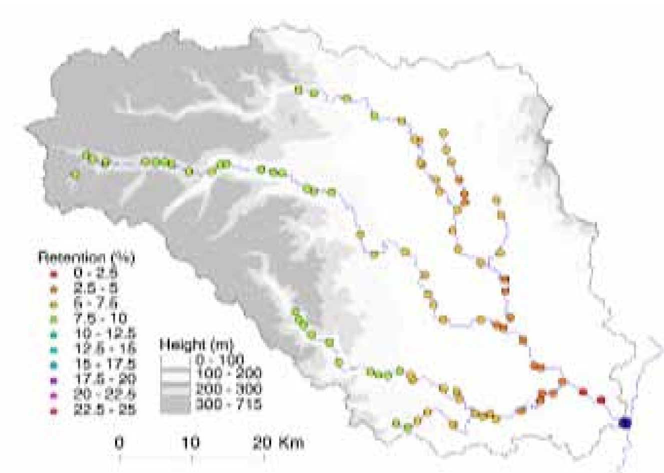
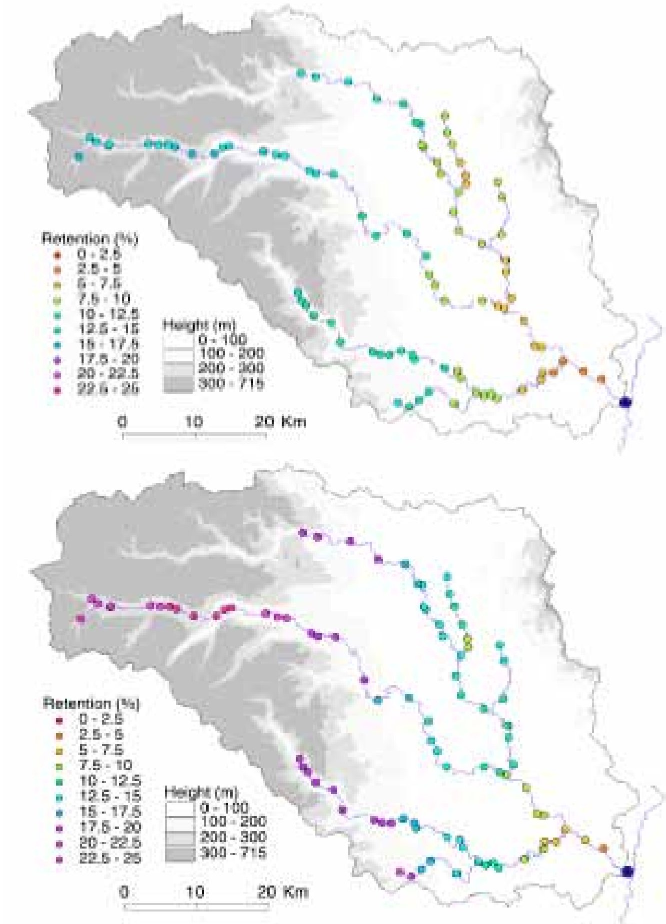


Figure 7.12. Comparison between N retention as calculated by the European (GREEN) and catchment (CEH) scale modeling approaches and catchment area.



River nitrate retention at high flow (Q10): retention defined between each point of interest (which approximate to locations of point and diffuse nitrate inputs) and the catchment outlet



River nitrate retention at median flow (Q50): retention defined between each point of interest (which approximate to locations of point and diffuse nitrate inputs) and the catchment outlet

River nitrate retention at low flow (Q95): retention defined between each point of interest (which approximate to locations of point and diffuse nitrate inputs) and the catchment outlet

Figure 7.13. Simulation of flow regime affecting in-stream nitrogen retention.

Atmospheric nitrate inputs are approximately 25% of the total input. This is significant as the atmospheric inputs are roughly in balance with outputs in this catchment. It is notable that the estimated amount of in-river denitrification exceeds the total amount of nitrate input to the river from point sources. Furthermore, of the total denitrification in the catchment, the river channel component is not dominant and accounts for about 30%.

The variability, due to flow, of nitrate sinks in the river network (Figure 7.13) indicates that the largest fluxes are observed at low flows. In these maps it can also be seen how river sinks are greatest for those inputs joining the river network farthest upstream from the catchment outlet.

Table 7.4. Comparison between N-budgets based on GREEN and the Ouse model for the Ouse catchment and for three different sub catchments.

	Ouse		Ure		Swale		Wiske*	
	Ouse Model	GREEN	Ouse Model	GREEN	Ouse Model	GREEN	Ouse Model	GREEN
Catchment area (km^2)	3315	3335	984	904	1449	1424	232	233
Modelled river length (km)	325	318	98	77	107	134	15	20
Loads (ton year^{-1})								
Nitrate-N exported output	7604	11434	1744	3713	3459	4742	743	593
River denitrification	671	855	138	214	280	267	40	10
Soil denitrification	1492		390		675		129	
Inputs (fertiliser plus atmospheric)	32995		8153		15753		3223	
Inputs (fertiliser plus manure plus atmospheric)	54889	45943	13796	11143	20338	19646	4982	3646
Point source loads to rivers	619	956	95	124	214	248	53	61
Diffuse source loads to rivers	7656	11333	1787	3803	3526	4761	729	542

Output mean nitrate-N concentrations (mg L^{-1})								
Simulated	4.01	5.25						
Simulated (if river denitrification absent)	4.35	5.46						
Simulated (if river and soil denitrification absent)	5.14							
Observed	3.61							

7.1.3.4. Organic matter retention in river networks

River networks are capable of clearing nearly all the organic matter that enters rivers trough diffuse or point sources. A budget is depicted in Figure 7.14. From the 4.2 million ton BOD that enters river networks, over 85% is decomposed in rivers.

Figure 7.15 shows the natural aquatic BOD_5 retention (%ret) for each cell as the percentage of all BOD_5 coming into the cell. %ret is generally higher in parts of Europe with less relief. This can be explained by the fact that calculated river depths are larger for such cells resulting in larger river reach volumes. Areas with extensive wetlands (especially in Sweden, Finland, and along the Dnieper river) often show low %ret. This results from an artefact in GWAVA-WQ which is that if a cell is covered by a wetland but not by the wetland outlet than the volume for that cell only consists of the river volume resulting in a locally underestimated BOD_5 retention. GWAVA-WQ assigns the reservoir volume of the wetland to the outlet cell, so all the BOD_5 retention resulting from the wetland occurs in that cell.

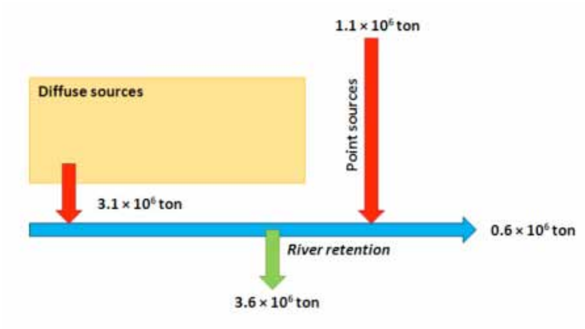


Figure 7.14. BOD5 budget for the European river network broken down over point source inputs, catchment-to-river flux, river retention and final loading at the outlet. Data based on the gwava model.

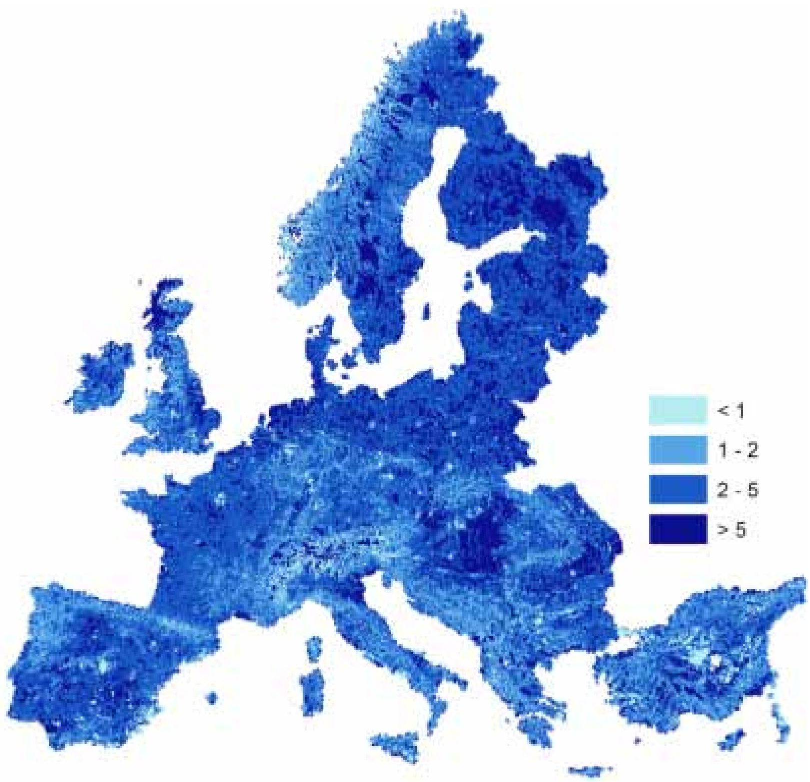


Figure 7.15. Spatial variation in natural aquatic BOD5 retention in Europe (%) modelled by GWAVA-WQ for 2000.

Figure 7.16 shows the amount of BOD_5 removed by natural aquatic retention processes in ton km^{-2} $year^{-1}$. This figure shows that the largest absolute amounts of organic matter are retained downstream in river networks. This can be explained by the fact that water volumes in rivers and reservoirs are larger further downstream in the river network, thus increasing the organic matter retention.

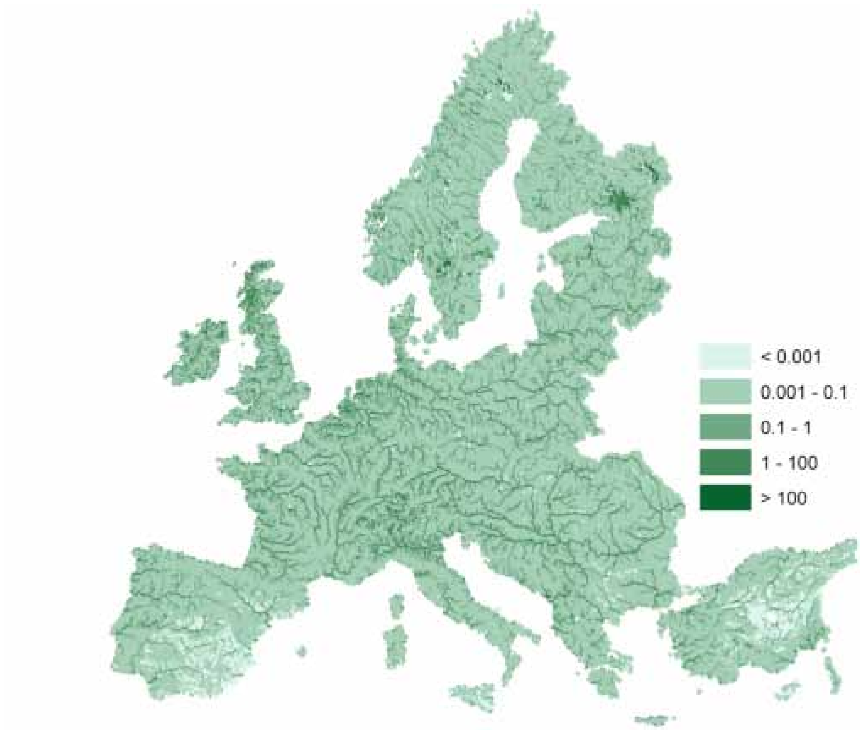


Figure 7.16. BOD_5 removed by natural aquatic retention (ton km^{-2} $year^{-1}$) modelled by GWAVA-WQ for 2000.

Figure 7.17 shows how the natural aquatic retention of anthropogenic BOD_5 (ret_{abs}) varies with upstream area. Generally it increases in downstream direction. This is especially the case when excluding cells covering lakes, reservoirs and/or wetlands. The contribution of lakes, reservoirs and wetlands on the total anthropogenic BOD_5 removal is large especially in where upstream areas are smaller.

Figure 7.17 shows also the BOD_5 level reduction due to natural aquatic retention (ΔBOD) and how it relates to the upstream area. The graph clearly shows that the effect of natural aquatic retention on BOD_5 level is larger in cells with a larger upstream area. The reason is that, in such cells a larger proportion of the aquatic organic matter has travelled over a large distance. Over that larger distance there was more opportunity for the organic matter to be retained. When impoundments cells are excluded (black line), the relation between BOD_5 level reduction and upstream area is very similar because the retention of organic matter in impoundments not only affects the BOD_5 level in the impoundment itself but also the BOD_5 levels further downstream. Thus the ecosystem service provided by impoundments has benefits over large distances in downstream direction.

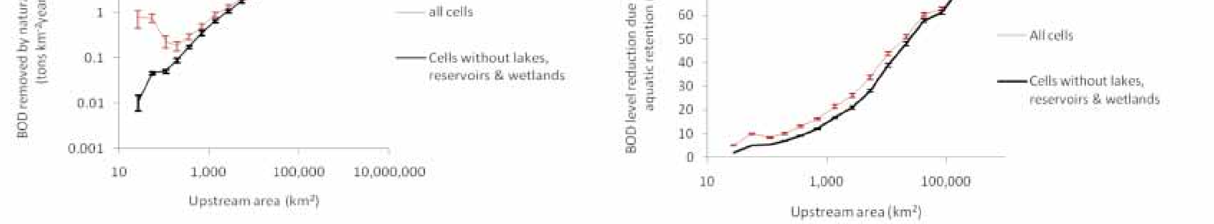


Figure 7.17. Left: water purification services along river networks. Area-specific removal of anthropogenic BOD_5 (ret_abs) in the year 2000 with increases in upstream area.The error bars show the 95% confidence interval. Right: water purification services along river networks. The reduction of BOD_5 levels due to natural aquatic processes bod increases with upstream area. The error bars show the 95% confidence interval.

Figure 7.18 shows the percentage BOD_5 level reduction due to natural aquatic retention (ΔBOD), and, like Figure 6, it clearly indicates that the BOD_5 level reduction increases in the downstream direction. Also it shows that ΔBOD is smaller in areas with a high relief (cause by lower modelled river depth reducing modelled water volume), and in areas with very low river discharge (resulting from lower estimates of river width and depth which reduce modelled water volume).

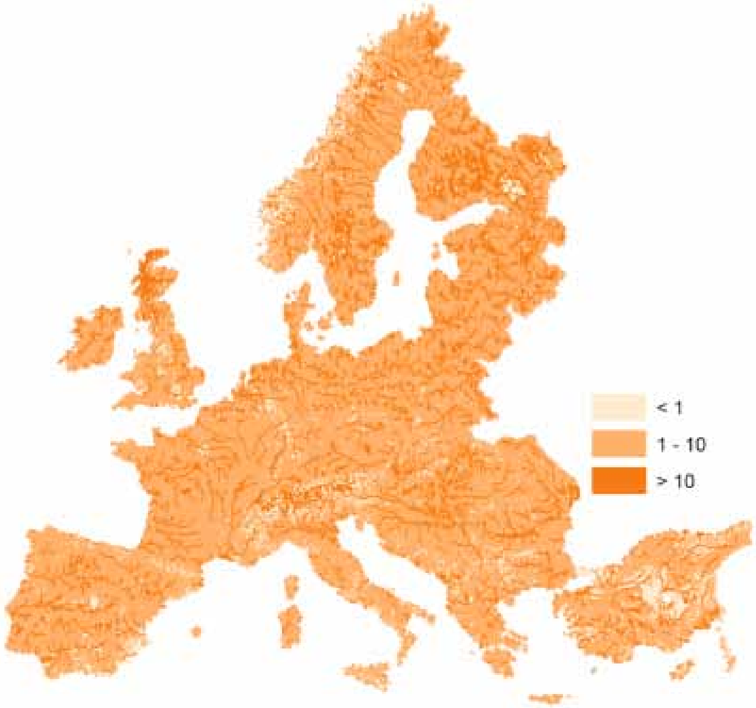


Figure 7.18. The effect of natural retention processes on BOD_5 level. The map shows the reduction in BOD_5 level (%) based on comparing a normal model run with a model run with zero natural aquatic retention.

7.1.4. Discussion and conclusions

This study case on water purification services illustrated scalable methodologies for mapping ecosystem services contributing to the removal of pollutants from surface waters and soils. The models that were applied in this mapping exercise showed that river networks considerably contribute to waste treatment. At the continental scale, over one million ton of nitrogen and over three million of BOD (as surrogate for organic matter) are removed from surface waters. This self cleaning capacity of rivers represents a service that is likely to be worth millions of euro.

Existing data and model applications show that the quantity of nitrogen retained in rivers is heavily dependent on the input nitrogen load. As inputs increase, so too does the retention flux. The process of nitrogen retention in rivers is facilitated by microbial denitrification in bed sediments and requires the interaction between water and the underlying substrate. The efficiency of retention is therefore dependent on hydraulic characteristics of the river channel. River network water quality models include representation of channel hydraulics and can potentially quantify the process. It is important to isolate and quantify this inherent and beneficial characteristic of the environment.

In general, model outputs based on the different approaches that have been used to map nitrogen retention by river networks compare well. In particular for the Ouse river catchment, the regional model developed for the Ouse yielded estimates of the same order of magnitude of retention, nitrogen removal and effect on water quality as a European approach. A similar conclusion was made for the Elbe catchment while for the Finnish case some considerable differences were observed, in particular in relation to the retention capacity of lakes and peat lands.

The different approaches at various spatial scales learned that the methods used in these studies are scalable in space. Retention coefficients based on physical properties such as river length and hydraulic load do not depend on spatial scale which means that the approaches used in the assessment of river retention are indeed scalable.

However, although catchment or basin based nitrogen budgets are comparable, there remain notable differences between regional and local assessments depending on the resolution of the input data. Lakes and wetlands increase the residence time of water resulting in increased retention. In Finland, an increase with 1% of the lake surface area, relative to other land cover, resulted in an increase of 7% of the retention, with a maximum of about 60% retention. Likewise, including detailed river and stream network maps with smaller sized water courses of high order results will result in increased water residence time which is not accounted for in a continental scaled approach. The result is that the latter methodology overestimates the biophysical ecosystem service flows that can be attributed to rivers and streams of low order and of large lakes at the cost of small sized streams, wetlands, ponds and lakes.

Rivers, streams and lakes have an important function in nitrogen retention and removal. All methods used show that total nitrogen retention equals roughly the combined input of point sources to the river network. Where this service results in a relatively small improvement of the water quality, on average, benefits increase in downstream direction and in-stream retention results in 50% N concentration reduction.

There remains a number of shortcomings in this assessment that need to be addressed in a follow up study.

None of the models takes explicitly into account the response of stream denitrification processes to anthropogenic nitrogen loading. The relation between the capacity of rivers to remove nitrogen and the total service flow, i.e. the total amount of nitrogen removed, depends on the input of nitrogen. In absence of nitrogen input the service flow is zero (no nitrogen is removed) but the benefit (good water quality and clean water) is high. Increasing nitrogen input will increase the total amount of nitrogen removed but further increments may impair the removal efficiency and hence result in a decrease of service and benefit.

Convincing evidence for streams is provided by Mullholland et al. (2008). They showed that total biotic uptake and denitrification of nitrate increased with stream nitrate concentration, but that the efficiency of biotic uptake and denitrification declined as concentration increases, reducing the proportion of in-stream nitrate that is removed from transport. This behavior is not captured by any of the models used in our assessment. However, the inflection point beyond which capacity of streams to remove nitrogen decreases is at lower nitrogen concentrations than typically seen in larger rivers. Therefore, it cannot be excluded that processes affecting retention of nitrogen include scale effects. In any case, the models used in this study rely essentially on geophysical properties for parameterizing river retention coefficients. Adding nitrogen loading or nitrogen concentration as a parameter as suggested by Mulholland et al. (2008) would allow including anthropogenic pressure which may be useful when addressing nitrogen scenarios with respect to agriculture, lifestyle and urbanization.

In-river nitrogen retention (in terms of ton km^{-1}) provides for the ecosystem service of water purification. Quantifying this requires a specific nitrogen concentration criterion for defining clean water to be chosen and defined. Then we ask, if nitrogen retention were absent, how many km less of the river network would have sufficiently clean water than is presently the case? Our models can answer this question. Any criteria of purity can be set for specific purposes and impacts then explored on the in-river nitrogen retention ecosystem service.

Another weakness is the lack of knowledge of the role of aquatic biodiversity in providing water purification services. Although it is clear the high biological activity in aquatic ecosystems results in self cleaning capacity of rivers and streams, the role of biodiversity in underpinning these services remains not fully understood. Also here additional efforts are needed to quantify how for instance good ecological quality as required by the Water Framework Directive relates to the maintenance of water quality.

7.2 Mapping recreation as a cultural ecosystem service

7.2.1. Introduction

Natural and semi natural ecosystems as well as cultural landscapes provide a source of recreation for mankind. People enjoy forests, lakes or mountains for hiking, camping, hunting, fishing or bird watching or just for being there. Recreation is also supplied by more intensively managed ecosystems such as agricultural lands and green urban areas. Despite being so important for human beings, the capacity and the flow of benefits associated with cultural services may be much more intangible and difficult to measure as compared to the provisioning and regulating services.

The capacity of ecosystems to provide recreation depends on multiple factors: their beauty, their uniqueness, the culture that generated them, the possibility for outdoor activities etc. We call the associated flow of benefits "fruition" which may be measured by performance indicators such as the number of visitors that annually visit a site or the appreciation of sites based on questionnaires. The relation between capacity and fruition is likely to be positive and is influenced by the accessibility of ecosystems to humans and the infrastructure that is in place to host or to guide visitors:

Fruition ~ Capacity × Accessibility

Ecosystems may be of extreme beauty but if they are not accessible, they will not provide a flow of cultural services. Also, ecosystems may be highly accessible but if their quality is low, the benefit flow they provide as is low as well.

Following this conceptual model, spatial indicators are needed that help to integrate the two independent variables, i.e. the capacity of ecosystems to provide recreation services and the transport infrastructure needed to access these ecosystems in order to generate the fruition or benefit flow.

The mapping of recreation services described in the following sections was carried out at EU and national level (for Finland and the Netherlands). The three exercises followed the same conceptual model but differed in their implementation considering the data available and the time limitations of the project.

7.2.2. Mapping of recreation potential

7.2.2.1 Data for mapping recreation services at EU scale

It must be underlined that data availability strongly drives the calculation of the recreation services. At the EU scale, in fact, there are no supporting data for calculating the actual fruition of the recreation service. There are neither harmonised data on accommodation facilities and touristic fluxes in non-urban areas at regional level, nor visitor fluxes of green areas in cities. Therefore this preliminary exercise is carried out on recreation potential in non-urban areas.

The final results of this exercise is a zonation of the EU into categories according to the Recreation Opportunity Spectrum (ROS) model (Clark and Stankey 1979; Joyce and Sutton 2009), and an analysis of what is the provision of the ES recreation service to the average European citizen.

The Recreation Opportunity Spectrum (ROS) was developed in the US to provide a framework for:

- Establishing outdoor recreation management goals and objectives for specific management areas.
- Trade-off analyses of available recreation opportunities as characteristic settings would be changes by other proposed resource management actions.
- Monitoring outputs in terms of established standards for experience and opportunities settings.
- Providing specific management objectives and standards for project plans.

Bullet 2 and partially bullet 3 are the scope of the present study. Furthermore, in this exercise landscape components of scenic beauty and culture are not addressed, and the provision of the service by the ecosystems in the strict sense is analysed.

Recreation potential is mapped with the assumption that it is positively correlated to a limited list of territorial features associated with attractiveness, i.e. the degree of naturalness, the presence of protected areas (following the assumption that they have been identified as holding a higher degree of naturalness, and as providers of recreation services and facilities), the presence of coastal lines (lakes and sea) and to the quality of bathing water. Such features can be mapped on the basis of the following data and indicators:

Hemeroby or degree of naturalness is an index that measures the human influence on landscapes and flora (Sukopp 1976; Wrbka et al. 2004; Fu et al. 2006). The European hemeroby map (Figure 7.19) is based on CLC land cover data; the average degree of naturalness has been attributed to each CLC class on the basis of literature, then agricultural and forested areas have been further reclassified on the basis of data concerning management, such as disaggregated data on nitrogen input and livestock density (provided by the CAPRI model) and the tree species database of the JRC (AFOLU action). CAPRI is an agro-economical model allowing regionalised impact analyses of the CAP. In Capri-Dynaspat dataset, production data of 30 crops in the European administrative regions for EU27 (from FSS statistics) have been broken down to, so-called, Homogeneous Spatial Mapping Units (HSMUs), identified by soil conditions, land cover, slope and administrative boundaries (Nuts 2 or 3) (Kempen et al. 2006). On the basis of disaggregated crop share, the model allows calculating indicators of intensity of management (as N input and livestock density) at HSMU scale. Input data for the base year are provided by the Farm Structure Survey (FSS). The AFOLU tree species datataset includes the distribution of more than 100 species in 1 km^2-cell grid layers. We used the distribution data of the 26 most abundant species in Europe and of 9 introduced species to attribute different degrees of naturalness to forest areas.

The presence of protected areas was mapped using the Natura 2000 database and the Common Database on Designated Areas (CDDA) database. The Natura 2000 database contains sites designated under the Birds Directive (Special Protection Areas, SPAs) and the Habitats Directive (Sites of Community Importance, SCIs, and Special Areas of Conservation, SACs). The CDDA or Common Dataset on Designated Areas holds information about protected sites and designation types at national or sub-national level (Figure 7.21).

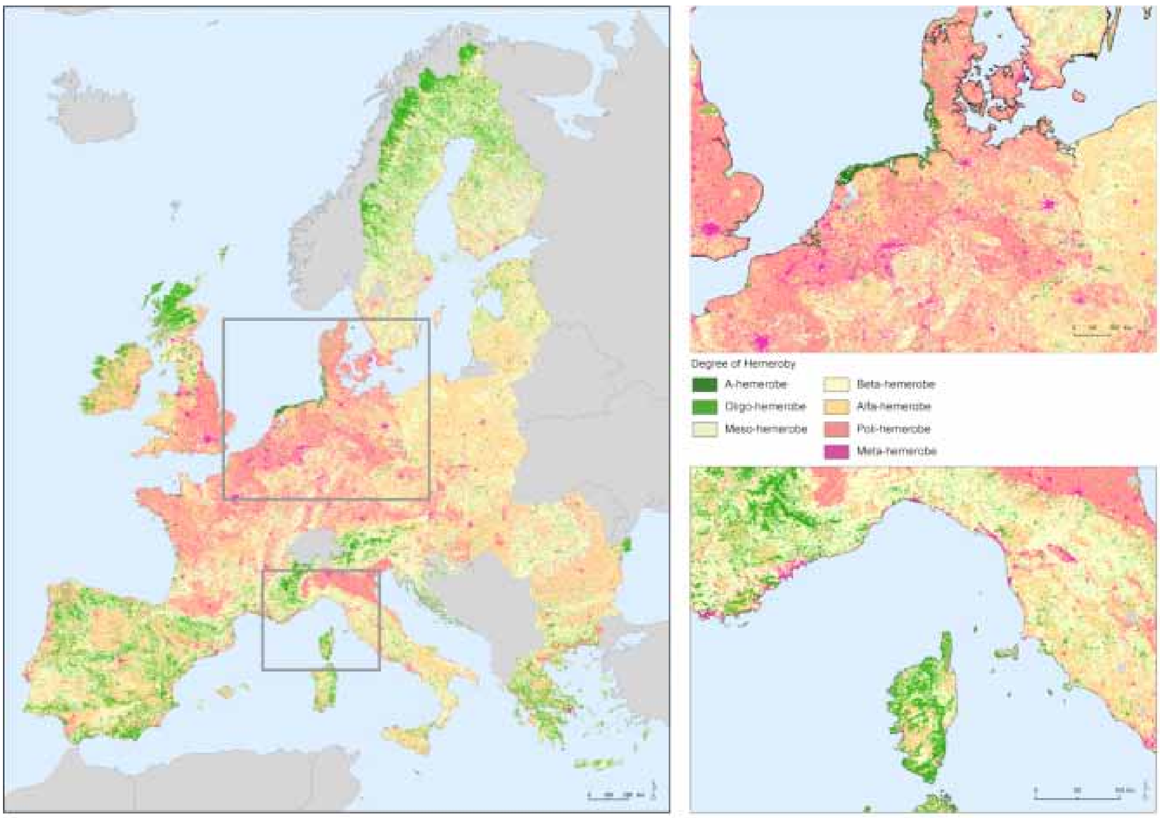


Figure 7.19. Degree of naturalness.

The CLC2000 dataset was used to extract the coastline of lakes and seas and the function represented in Figure 7.20 was used to calculate the distance from coast, with the assumption that presence of water is an attraction function for recreation.

Data on bathing water quality, as measured under the EU Bathing Waters Directive, were used to add weight to the coastline indicator. These data are annually collected by the EEA. The resulting component related to recreation provision by coasts and water is represented in Figure 7.20.

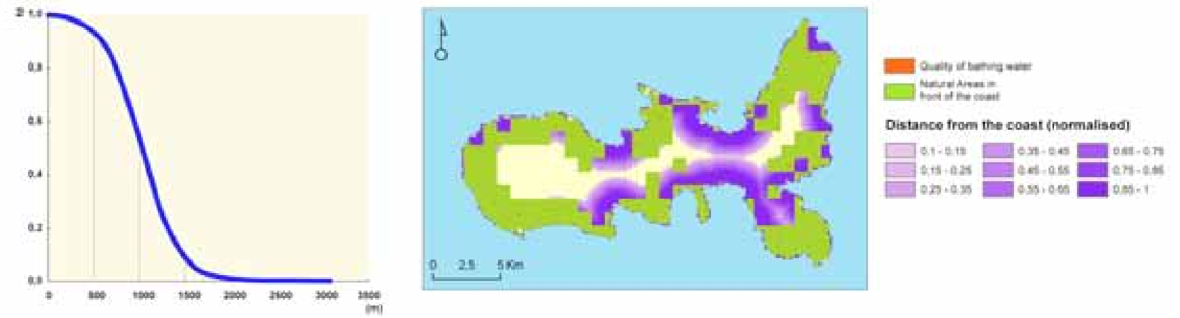


Figure 7.20. Left: function used to calculate distance from coast; Right: components of the indicator on presence of water bodies and coastal areas.

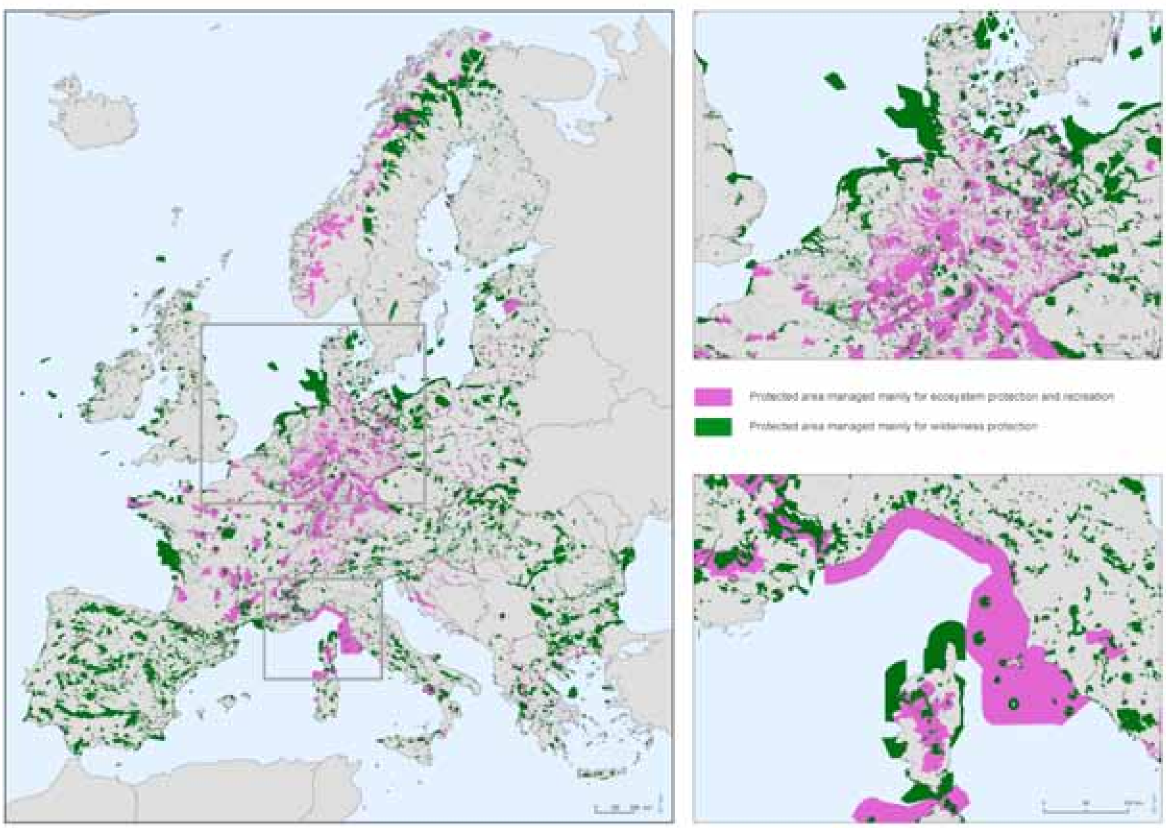


Figure 7.21. European inventory of nationally designated areas.

7.2.2.2 Recreation potential index

Different aggregation methods have been tested, assigning different set of weights to the different components. The results presented in this document refer to the scheme in Figure 7.22, where the three components concerning naturalness, protected areas and presence of water bodies hold the same weight.

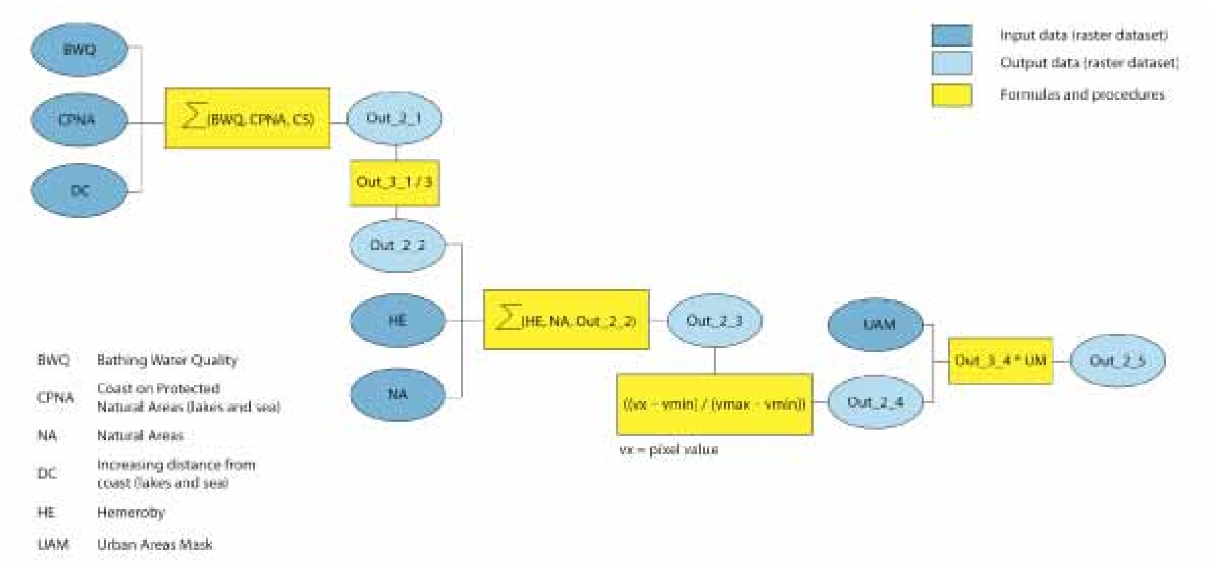


Figure 7.22. Recreation services: data aggregation scheme.

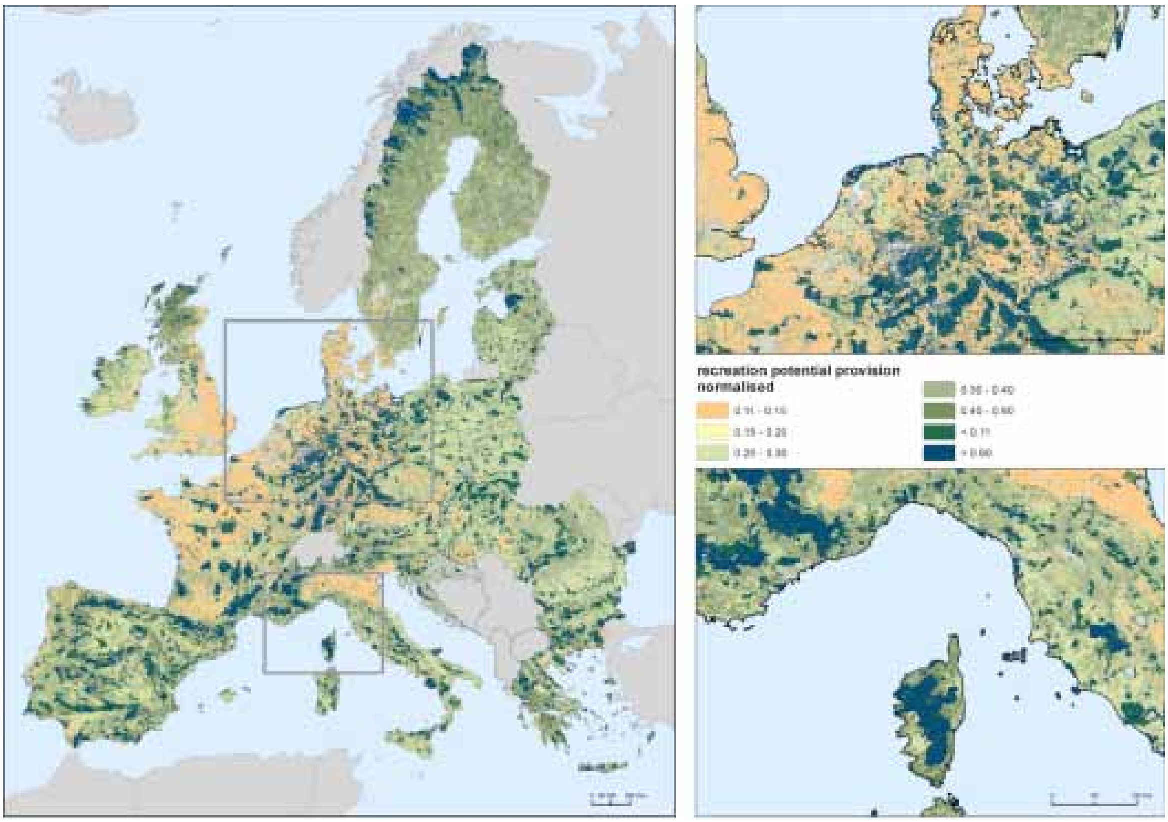


Figure 7.23. Recreation Potential Index

The final result, the RPI (Recreation Potential Index), expresses the capacity of ecosystems to provide recreational services, and is presented in Figure 7.23.

7.2.3. A recreation opportunity spectrum for the EU

The purpose of the Recreation Opportunity Spectrum (ROS) Inventory is to: identify, delineate, classify and record areas within a region/country into recreation opportunity classes based on their current state of remoteness, naturalness and expected social experience (Ministry of Forests, Forest Practices Branch for the Resources Inventory Committee, Canada, 1998). Examples of application of the ROS inventory to non-EU nations (US, Canada, Australia, New Zealand, Japan) are available in literature (Clark and Stankey 1979; Parkin et al. 2000; Joyce and Sutton 2009; Yamaki et al. 2003). In the current application a ROS has been established for Europe, adapting overseas experiences to the peculiarities of the European continent. This required the zoning of Europe in terms of proximity vs remoteness, to be defined on the basis of available data (TeleAtlas road network and the distribution of urban areas extracted from CORINE land cover), that was carried out through an expert survey. A panel of European experts was asked to fill out a table in which they had to define thresholds for distances from roads and urban, and assign each combination a label (neighbourhood, proximity, far, remote, very remote). Results were averaged and presented in Table 7.5. Once applied to the data, it provides the map shown in Figure 7.24.

Table 7.5. Scheme identifying the classes of a recreation opportunity spectrum for the EU.

			Distance from road classes (m)					ROS 1 categories	
			1	2	3	4			
			< 1000	1000-5000	5000-10000	> 10000			
Distance from urban areas classes (m)	1	0 - 5000	1	2	2	4		1	Neighborhood
	2	5000 - 10000	2	2	2	4		2	Proximity
	3	10000 - 25000	3	3	3	4		3	Far
	4	25000 - 50000	3	**4**	**4**	4		4	Remote
	5	> 50000	4	4	4	5		5	Very Remote

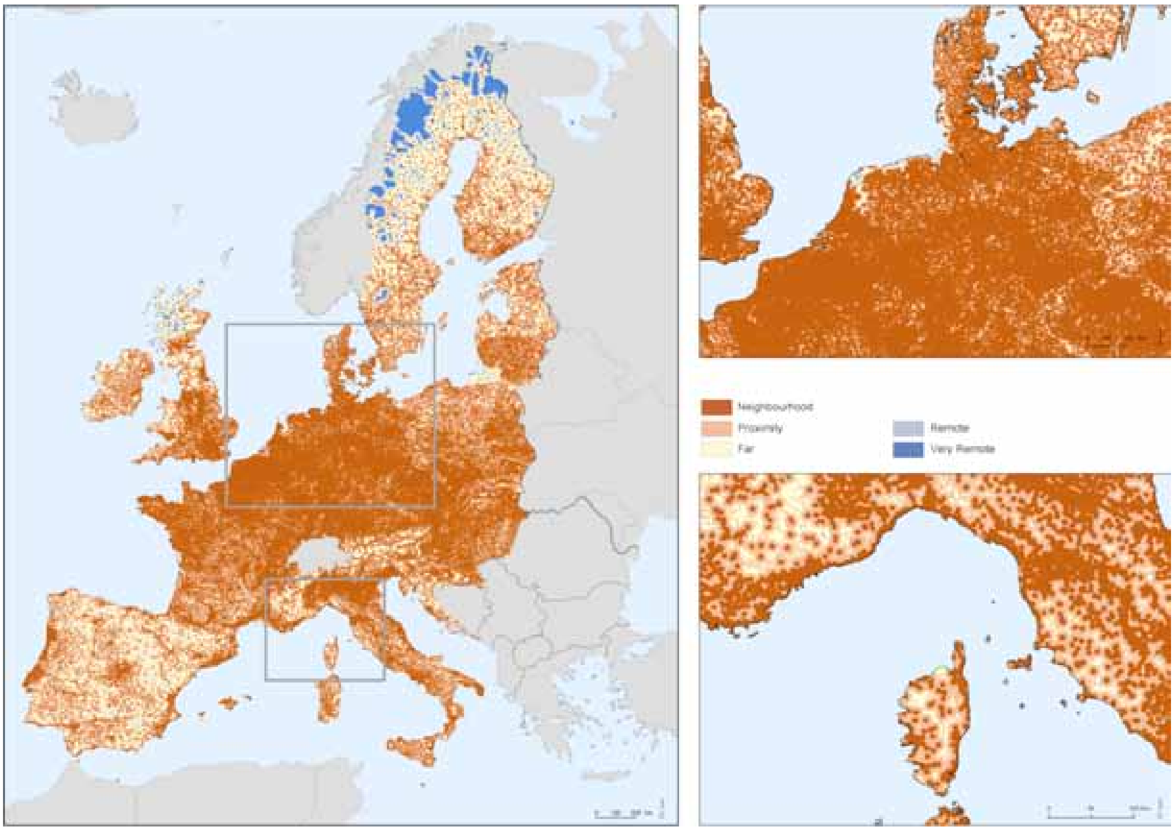


Figure 7.24. Map of Europe according to classes of proximity vs remoteness.

The final ROS (Figure 7.25) has been obtained by merging the indicator on recreation provision and the zoning for the EU on the basis of the table shown in Table 3.6. Thresholds for the potential recreation index were derived from a statistical analysis of data distribution, coupled with a cross-analysis of control sites. The result provides information both on the quality of recreation provision and its accessibility.

Table 7.6. Classes of the European ROS.

		Recreation Potential						
		1	2	3	4	5	6	7
		< 0.11	0.11 -0.15	0.15 - 0.20	0.20 - 0.30	0.30 - 0.40	0.40-0.60	> 0.60
ROS1	1-Neighborhood	1	1	4	4	7	7	7
	2-Proximity	1	1	4	4	7	7	7
	3-Far	2	2	5	5	8	8	8
	4-Remote	3	3	6	6	9	9	9
	5-Very Remote	3	3	6	6	9	9	9

1	low provision - easily accessible
2	low provision - accessible
3	low provision - not easily accessible
4	medium provision - easily accessible
5	medium provision - accessible
6	medium provision - not easily accessible
7	high provision - easily accessible
8	high provision - accessible
9	high provision - not easily accessible

Figure 7.25. Recreation Opportunity Spectrum for the EU (see map 3.4 page 25).

7.2.4. Recreation provision to the European citizen

An analysis on population data allows estimating the quality of recreation provision to the European citizens. The current exercise addresses recreation and not tourism, and in particular daily recreation, ranging from a short walk or a bicycle ride to a car displacement for a Sunday trip. This can be estimated at EU, national or regional level.

Population pressure is calculated on the basis of population density, assuming that in daily recreation the maximum travelled distance is 60 km. The resulting indicator is shown in Figure 7.26, and is used to calculate population access to ROS zones.

Population pressure is expressed using the cumulative population living in the surroundings of each 1 km^2 cell; therefore it is based on the number of visitors that each location in the EU can potentially receive. The shares are calculated referring to the total cumulative population, either of the EU or of a single Member State (therefore counting each citizen 11304 times, which is the number of cells he can reach in a 60 km radius). Final statistics have to be interpreted from the point of view of the share of the cumulative population (Figure 7.26) that has access to the different ROS zones. Based on these figures, as expected, final results show that the great majority of population has access to areas where accessibility is high, and quality of provision is low (23.7%), medium (44.6%) or high (26.8%) (Figure 7.27). It is as well interesting to see how the shares change in different countries. Compared to Sweden, for example, Italy has a higher share of population that can easily access areas with a medium degree of recreation provision (50.2% vs 45.7%), but 24.5% of population has access to areas with a low degree of recreation provision, compared to 1.3% in Sweden (Figure 7.28).

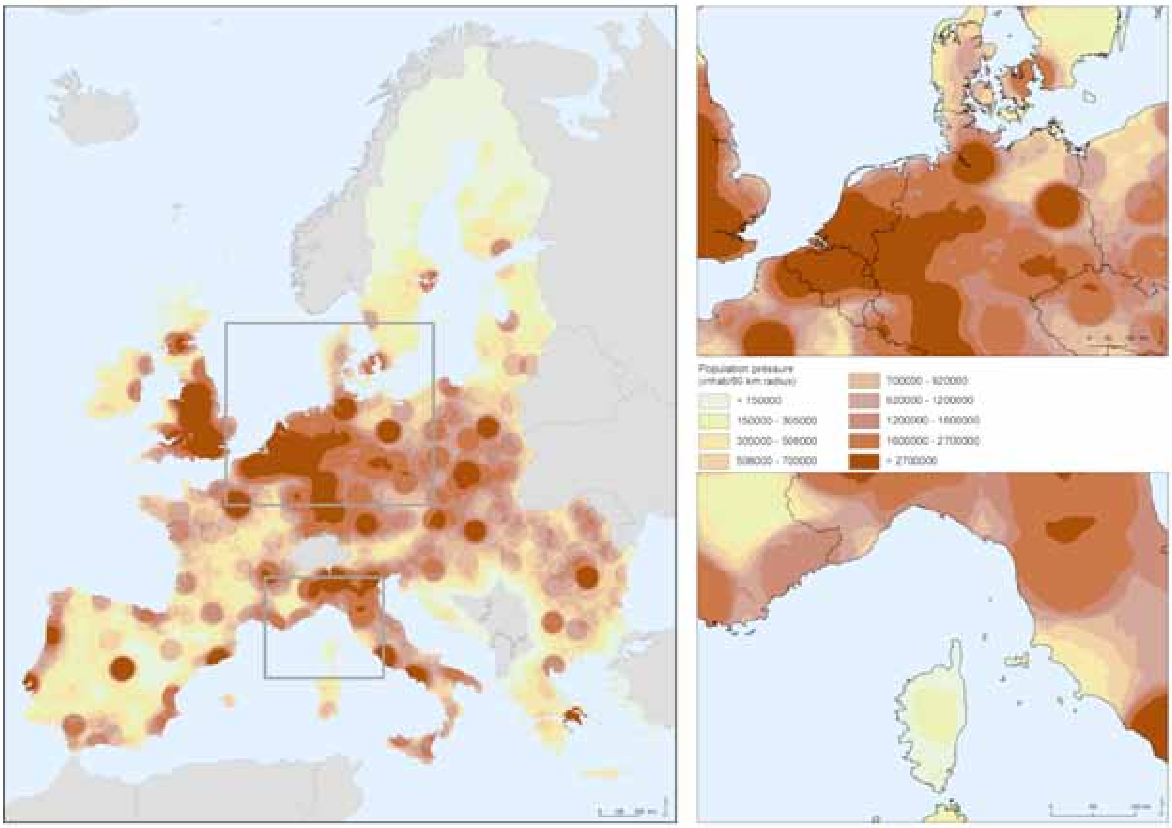


Figure 7.26. Population pressure in the EU, calculated on a 60 km radius.

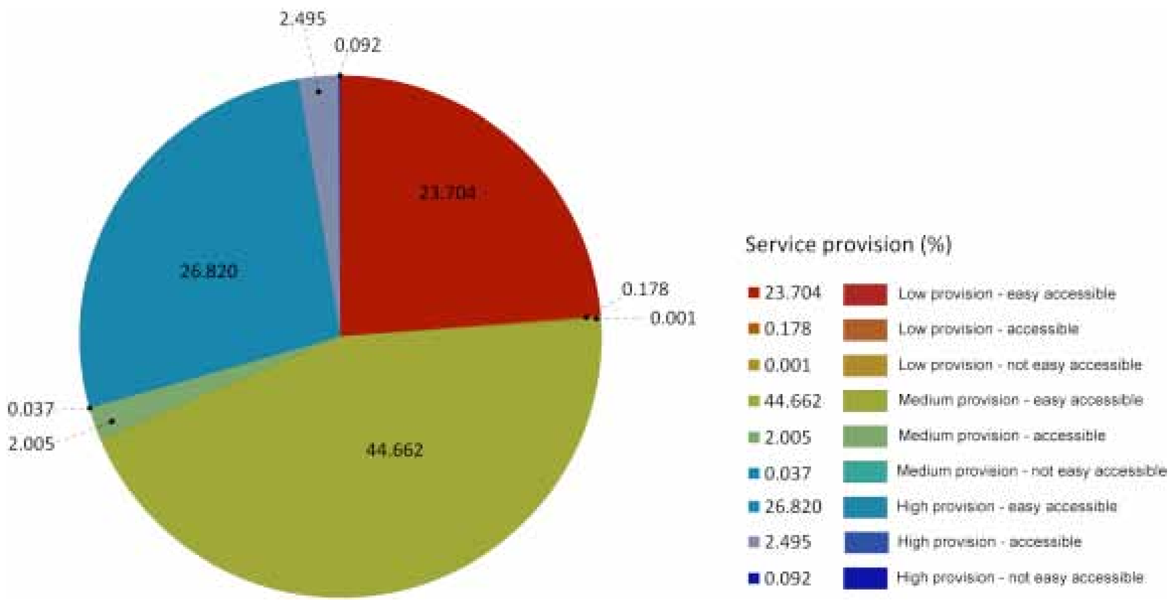


Figure 7.27. Population access to ROS areas.

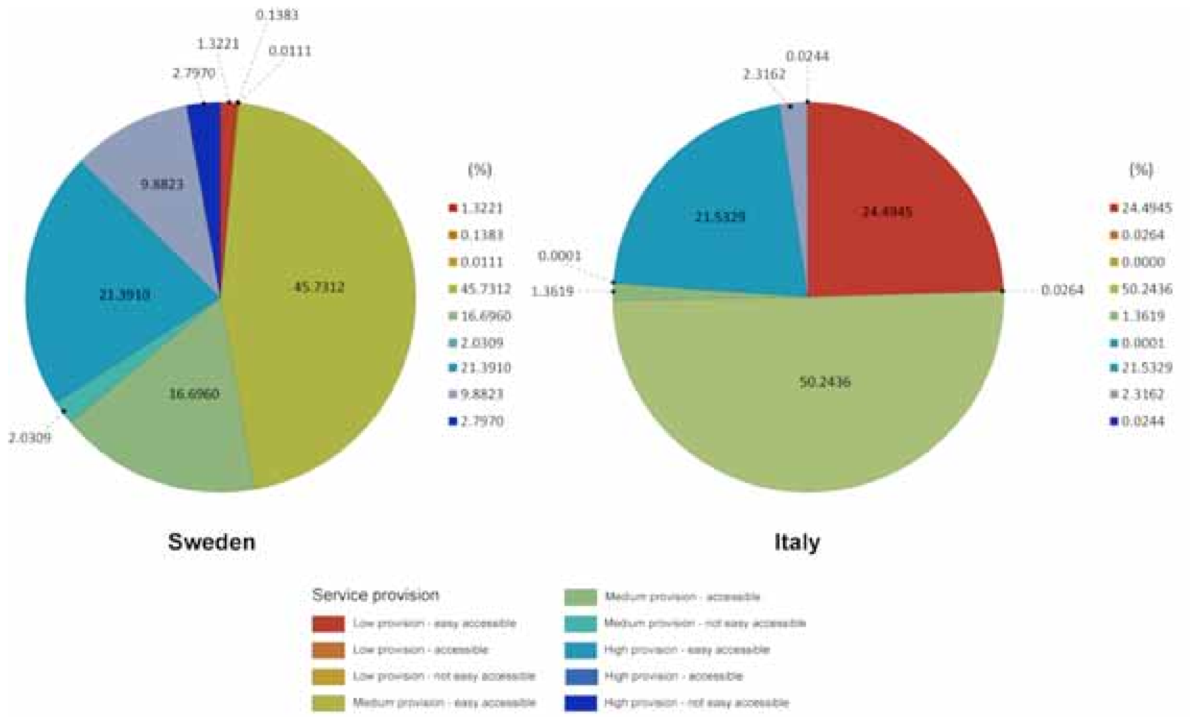


Figure 7.28. Population access to ROS areas in Sweden and Italy.

7.2.5. Mapping recreation services at country scale: the case of Finland

7.2.5.1 Recreation Potential Index

The methodology described above to map recreation services at continental level has been adapted and downscaled for an application at Country level, namely on Finland. The wide availability of data at a more detailed resolution has allowed some refinement in the procedure, but also required some changes.
Also in this case visitors' data were not available to calculate the actual fruition of recreation services, but data on summer cottages and recreation facilities allowed a more in-depth analysis of the links with recreation potential.
The components of recreation potential are the same of the EU wide exercise, modified as follows:
Hemeroby or degree of naturalness: the hemeroby layer was recalculated on the basis of the EU methodology on CORINE level 4 available at 25 m resolution. The hemeroby state for the more detailed land cover legend was obtained from literature and from discussion with experts. In particular for forests:

- Lapland forests are mainly unmanaged, and the degree of naturalness does not depend on their protection status. These forests can be given a higher naturalness value compared to the EU average even if not part of a protected area.
- In Lapland and in the Province of Kainuu forests in protected areas are mainly unmanaged and can be given a higher naturalness value (for Lapland the value is the same as in the bullet above).
- Forests in protected areas in other parts of Finland are managed forests and should be given a lower naturalness value compared to the EU average.
- All other forests are given a lower naturalness value than forests in protected areas.

The layer of protected areas was derived from the following sources: Natura 2000 data, UNESCO sites, Nationally designated areas (National - CDDA), Finnish National Parks and Local Protected Areas.
CORINE land cover map at 25 m resolution was used to extract the coastline of lakes and seas. The function used to calculate the distance from coast is the same as reported in Figure 7.20.
Data on bathing water quality were added as in the EU wide exercise.

The recreation potential index was calculated following the scheme presented in Figure 7.22, and also in this case expresses the capacity of ecosystems to provide recreational services (Figure 7.29).

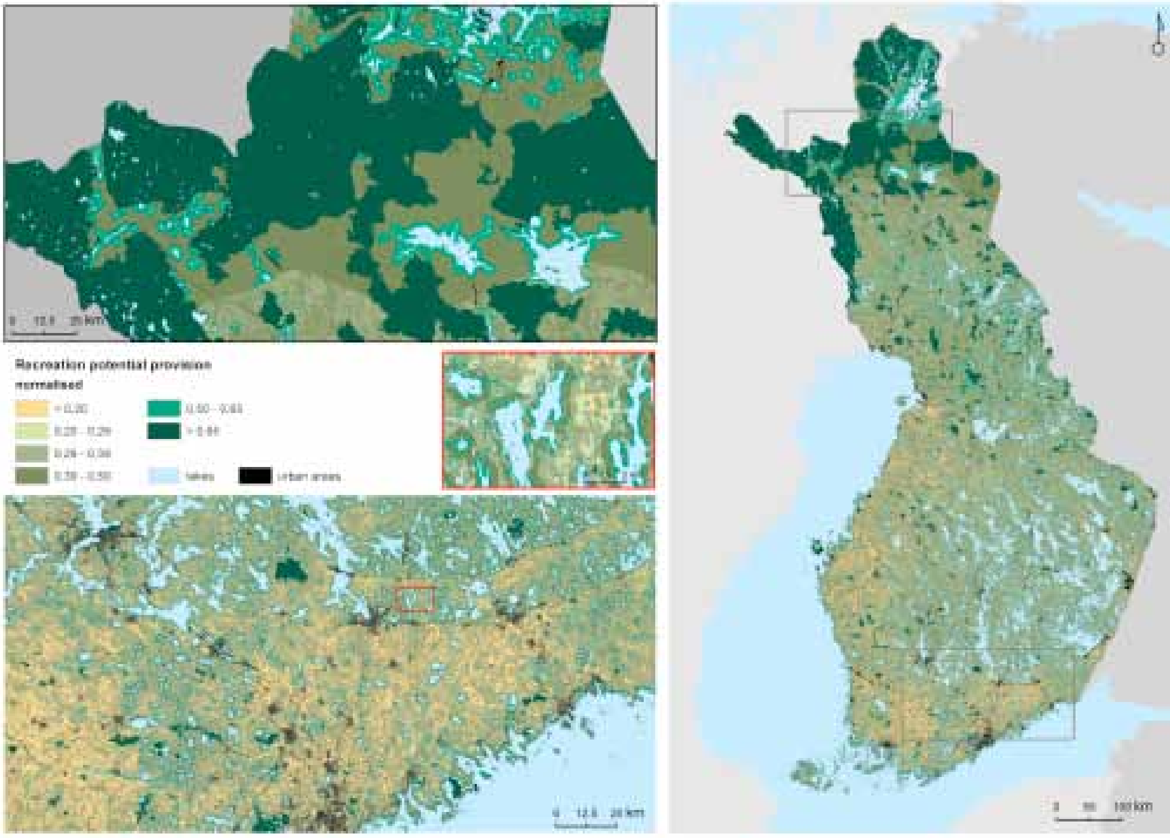


Figure 7.29. Recreation Potential Index for Finland (Corine Finland level 4/"CLC2006 - Maankäyttö/Maanpeite (25 m)" - copyright: SYKE, partially METLA, MMM, MML, VRK -boundaries of Forest Lapland/"Metsäkasvillisuusvyöhykerajat (ym)" - copyright: SYKE -boundaries of Kainuu province/"Inspire1.geo.hall100mkunta" - copyright: National Land Survey of Finland 7/MML/11 -Finnish National Parks and Local Protected Areas/"Luonnonsuojelualueet ja erämaat" - copyright: SYKE, Metsähallitus, Centres for Economic Development, Transport and the Environment)

7.2.5.2 A Recreation Opportunity Spectrum for Finland

The ROS zones have been redefined for the Finnish study case. In fact the table reported in Table 7.7 averages the perception of proximity/remoteness over the European continent, but analysed singularly in each Country such perception changes, depending on the environmental setting, the anthropic context, culture, education (Lupp et al. 2011). Therefore the scheme was recalculated on the basis of input from a panel of Finnish experts (Fig. 14). The resulting map is shown in Figure 7.30.

Table 7.7. Scheme identifying the classes of a Recreation Opportunity Spectrum for Finland

			Distance from road classes (m)					ROS 1 categories	
			1	2	3	4			
			< 300	300-1000	1000-3000	> 3000			
Distance from urban areas classes (m)	1	< 700	1	2	2	4		1	Neighborhood
	2	700 - 2000	2	2	2	4		2	Proximity
	3	2000 - 9000	3	3	3	4		3	Far
	4	9000 – 35000	3	4	4	4		4	Remotness
	5	> 35000	4	4	4	5		5	Very Remotness

Input data for the Finnish ROS are the TeleAtlas road network (levels 1-7) including gravel roads, and urban areas extracted from CORINE land cover map at 25 m resolution. Urban areas have been defined clustering pixels of urban residential classes closer than 200 m to each other. Isolated pixels have not been taken into consideration.

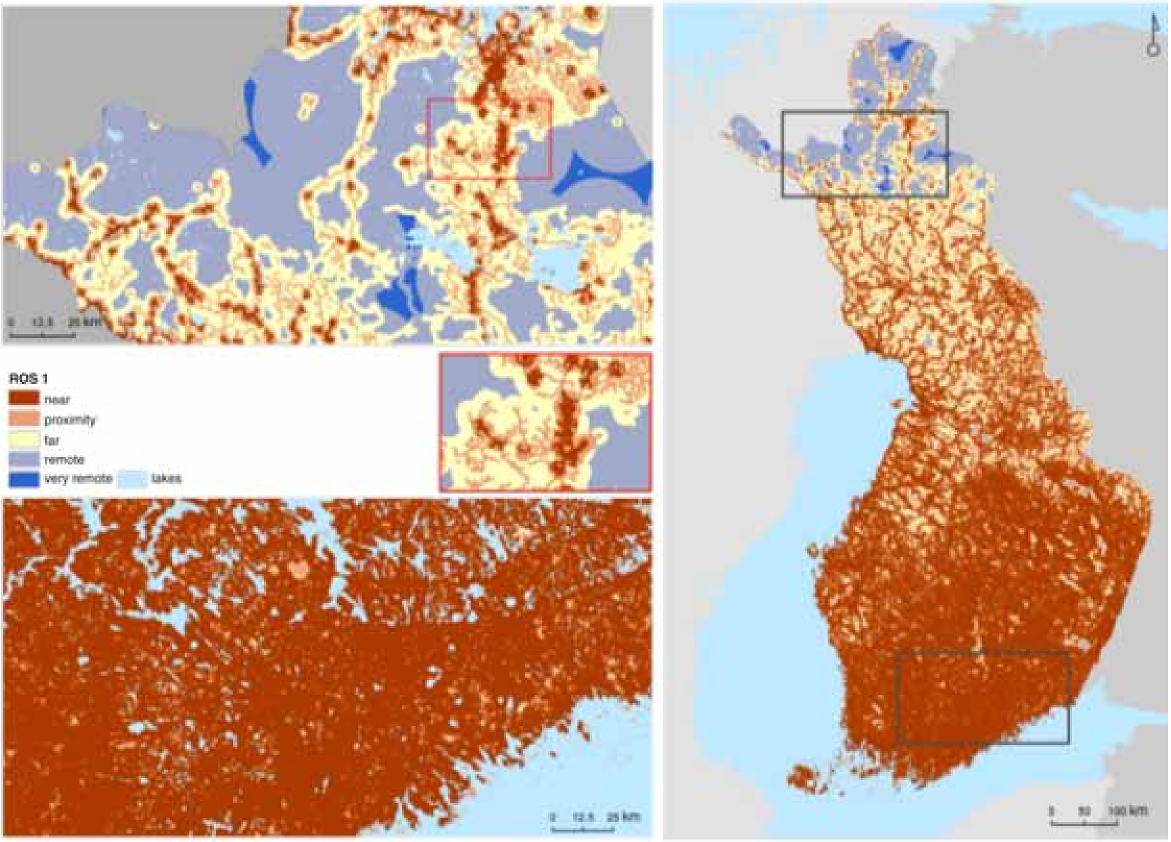


Figure 7.30. Map of Finland according to classes of proximity vs remoteness

The final ROS (Figure 7.31) has been obtained by merging the indicator on recreation provision (Figure 7.29) and the zoning of Finland on the basis of Table 7.8. As in the case of the EU, the result provides information both on the quality of recreation provision and its accessibility.

7.2.5.3 Recreation provision to the Finnish citizen

In the case of Finland, besides calculating population pressure through a focal analysis as in the EU case, a different approach was tested, based on the concept of "population active living potential" which refers to the conditions of areas that encourage the likelihood of integrating physical activity into daily routines (Riva et al. 2008).

Table 7.8. Classes of the ROS for Finland

		R					
		1	2	3	4	5	6
		< 0.20	0.2 -0.29	0.29 - 0.39	0.39 - 0.50	0.50 - 0.65	> 0.65
ROS1	1-Neighborhood	1	4	4	4	7	7
	2-Proximity	1	4	4	4	7	7
	3-Far	2	5	5	5	8	8
	4-Remote	3	6	6	6	9	9
	5-Very Remote	3	6	6	6	9	9

1	low provision - easily accessible
2	low provision - accessible
3	low provision - not easily accessible
4	medium provision - easily accessible
5	medium provision - accessible
6	medium provision - not easily accessible
7	high provision - easily accessible
8	high provision – accessible
9	high provision - not easily accessible

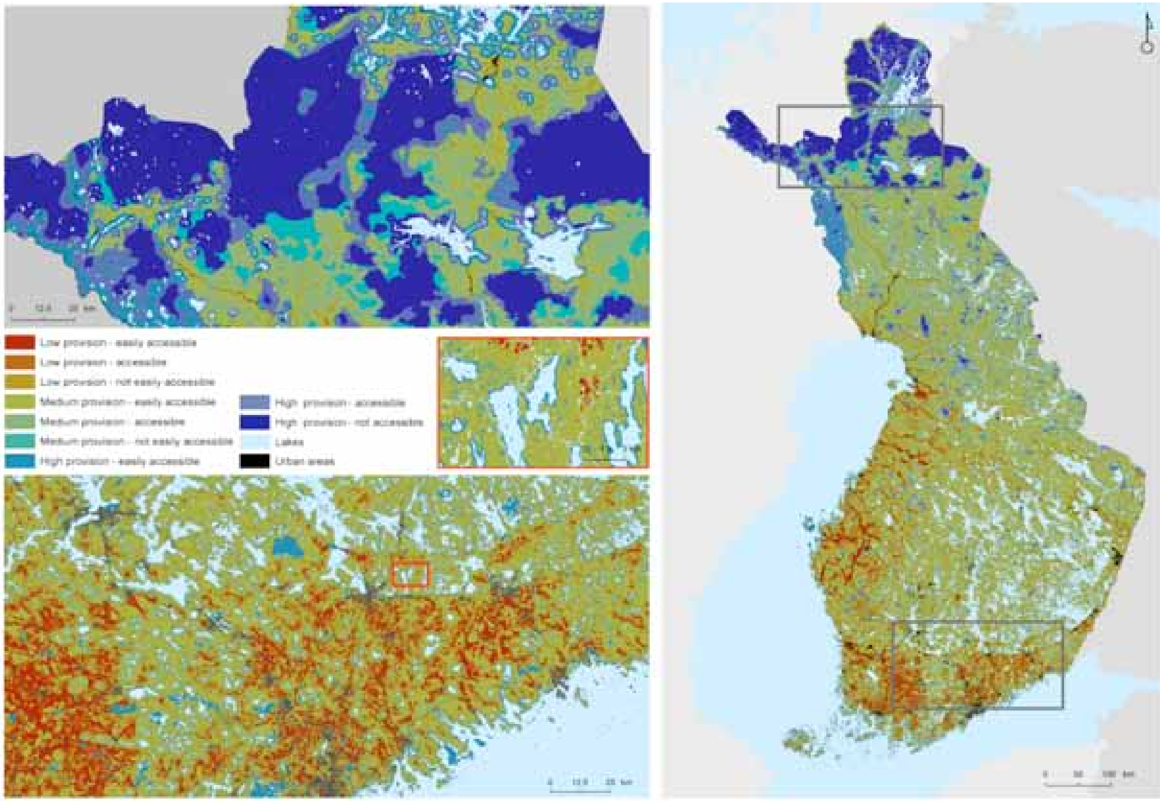


Figure 7.31. Recreation Opportunity Spectrum for Finland.

The *population living potential* is a particular type of smoothing technique which derives from a specific application of the potential accessibility concept (Kafadar 1996; Rezaeian et al. 2004).

Generally the potential accessibility measures depend on a cost sensitivity function of the distance, used as model of people's capacity to reach a destination (Kwan, 1998; Kwan, 1999; Talen and Anselin 1998). We estimate that the active living potentials of individuals decrease following an inverse logistic function of distance, so data are allocated as follow:

$$P_j = \sum_{i=1}^{n}\left(p_i \cdot \frac{f(d_{ij})}{\sum_{k=1}^{n} f(d_{ik})} \right)$$

Where P_j: smoothed data of cell j; p_i: cell i data; $f(d_{ij})$: inverse logistic function of the distance between cell centroid i and cell centroid j. Such model (shown in Figure 7.32) guarantees that the total amount of smoothed population is equal to the original data (total Finnish population).

Table 7.9 summarizes the share of population having access to the different ROS areas, as calculated per NUTS3 region on the basis of both the focal analysis and the active living potential. Results show similar trends, differences are due to different accounting of densely populated areas when they are located in the tail of the distribution function in Figure 7.32.

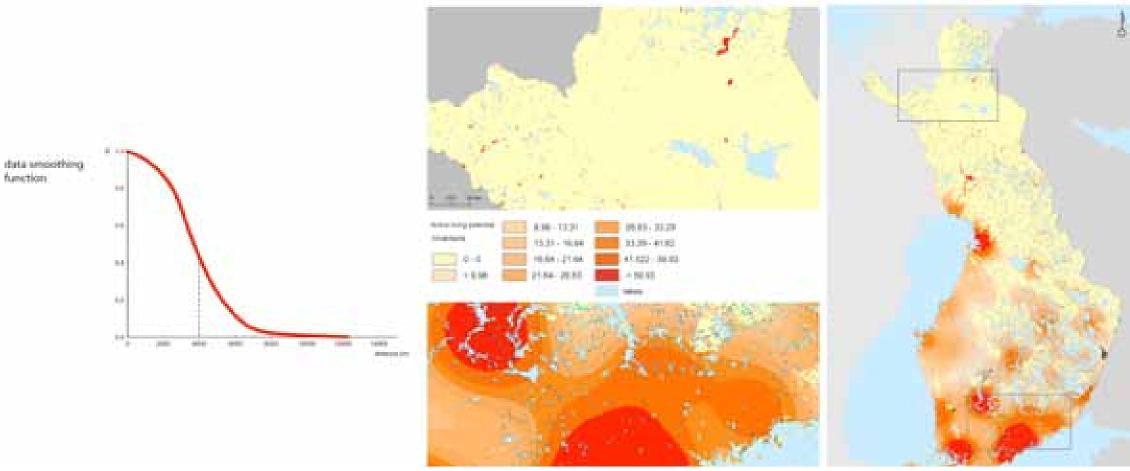


Figure 7.32. Left: function used to calculate population active living potential; Right: population active living potential.

Table 7.9. Percentage of population access in Finland for NUTS 3

	North Karelia (1)		Central Ostrobothnia (2)		Southern Ostrobothnia (3)		Southern Savonia (4)		Lapland (5)		Kainuu (6)	
	*	**	*	**	*	**	*	**	*	**	*	**
Low provision - easily accessible	3,32	5,62	12,15	15,22	20,61	21,83	0,98	1,71	1,21	5,65	1,17	2,86
Low provision - accessible	0,02	0,01	0,22	0,14	0,13	0,10	0,00	0,00	0,12	0,08	0,08	0,08
Low provision - not easily accessible	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
Medium provision - easily accessible	77,93	78,54	66,36	69,61	69,29	68,95	71,01	73,50	46,02	72,68	66,91	77,54
Medium provision - accessible	2,54	0,51	11,37	7,88	4,15	3,94	0,45	0,16	29,22	7,05	13,59	5,30
Medium provision - not easily accessible	0,02	0,00	0,00	0,00	0,00	0,00	0,02	0,00	0,73	0,00	0,00	0,00
High provision - easily accessible	15,06	15,14	5,39	5,03	4,77	4,47	26,85	24,43	10,13	13,52	13,91	13,25
High provision - accessible	1,00	0,18	4,50	2,13	1,05	0,71	0,62	0,16	8,96	1,01	4,29	0,96
High provision - not accessible	0,11	0,00	0,01	0,00	0,00	0,00	0,08	0,04	3,61	0,00	0,05	0,00
	Northern Ostrobothnia (7)		Northern Savonia (8)		Aland (9)		Uusimaa (10)		Finland Proper (11)		South Karelia (12)	
	*	**	*	**	*	**	*	**	*	**	*	**
Low provision - easily accessible	9,39	13,34	2,57	3,41	2,32	3,32	17,86	17,82	28,25	28,22	5,04	8,79
Low provision - accessible	0,33	0,17	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	74,09	0,00
Low provision - not easily accessible	0,01	0,01	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,43	0,00
Medium provision - easily accessible	64,49	70,15	75,03	75,63	66,79	74,86	68,94	67,95	62,07	62,66	0,02	74,79
Medium provision - accessible	15,30	8,59	1,54	0,57	0,11	0,13	0,03	0,03	0,34	0,27	20,14	0,14
Medium provision - not easily accessible	0,09	0,05	0,00	0,00	0,17	0,04	0,00	0,00	0,00	0,00	0,20	0,00
High provision - easily accessible	6,44	6,33	20,36	20,26	29,13	21,38	13,17	14,21	8,95	8,63	0,07	16,29
High provision - accessible	3,65	1,25	0,49	0,14	0,62	0,10	0,00	0,00	0,38	0,21	0,00	0,00
High provision - not accessible	0,31	0,11	0,00	0,00	0,87	0,18	0,00	0,00	0,00	0,00	0,00	0,00
	Kymenlaakso (13)		Pirkanmaa (14)		Central Finland (15)		Paijanne Tavastia (16)		Tavastia Proper (17)		Eastern Uusimaa (18)	
	*	**	*	**	*	**	*	**	*	**	*	**
Low provision - easily accessible	14,81	16,64	7,43	6,96	2,23	2,42	16,17	16,98	15,43	15,67	20,99	21,39
Low provision - accessible	0,04	0,05	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
Low provision - not easily accessible	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
Medium provision - easily accessible	72,80	72,65	79,66	80,02	77,84	79,30	72,85	73,10	72,74	72,82	70,41	69,77
Medium provision - accessible	0,33	0,37	0,36	0,27	1,44	0,38	0,12	0,09	0,17	0,18	0,09	0,06
Medium provision - not easily accessible	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
High provision - easily accessible	11,89	10,14	12,41	12,69	18,04	17,85	10,82	9,81	11,63	11,29	8,44	8,70
High provision - accessible	0,14	0,15	0,14	0,06	0,45	0,06	0,04	0,02	0,03	0,03	0,07	0,08
High provision - not accessible	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
	Satakunta (19)		Ostrobothnia (20)									
	*	**	*	**								
Low provision - easily accessible	18,46	20,15	19,60	19,38								
Low provision - accessible	0,01	71,89	0,06	0,06								
Low provision - not easily accessible	0,00	0,00	0,00	0,00								
Medium provision - easily accessible	72,28	0,00	66,98	66,49								
Medium provision - accessible	0,70	0,50	4,39	3,76								
Medium provision - not easily accessible	0,01	0,02	0,01	0,01								
High provision - easily accessible	8,14	7,26	7,93	9,62								
High provision - accessible	0,38	0,14	0,91	0,59								
High provision - not accessible	0,03	0,03	0,11	0,09								

(* = Focal Statistic analysis output, ** = Population Living Potential output)

7.2.5.4 Model to approximate preferences for outdoor recreation

Availability of data allowed the definition of a model to describe preferences of Finnish population for recreation in nature. The reference dataset for this exercise is the density and distribution of summer cottages (Figure 7.33).

The explanatory variables taken into consideration are the availability of recreation facilities in State owned land, expressed as attractivity exerted on neighbouring population in terms of distance, the accessibility expressed as friction exerted by the road network on the accessibility of every location in Finland, the distance from coast (lake and sea) and the recreation potential index.

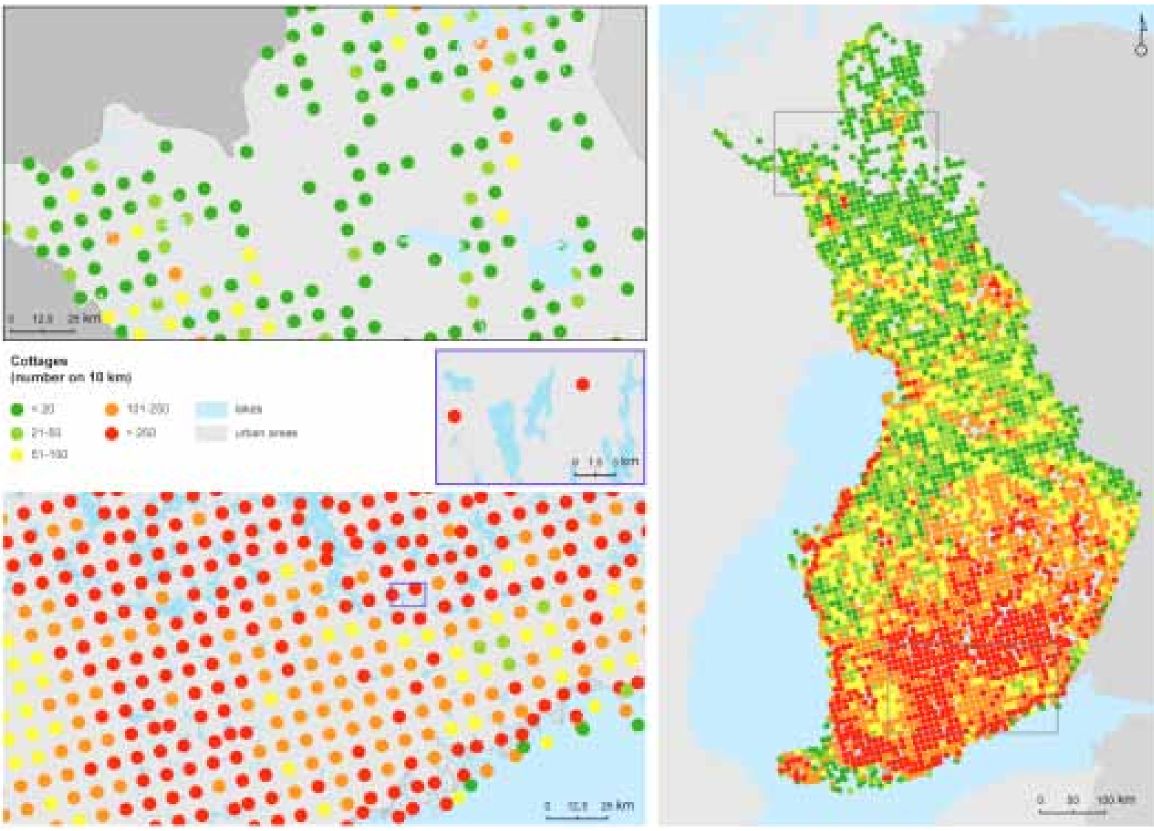


Figure 7.33. Distribution of summer cottages per 10 km cells ("RHR Rakennukset" - copyright: SYKE, VTJ/VRK 4/2010)

Attractivity exerted by recreation facilities on neighbouring population. Figure 7.34 shows available input data, these have been grouped according to the categories listed in Table 7.10. It has to be pointed out, though, that data in some categories are deficient (i.e. campings, harbours), nevertheless they have been used to show the applicability of the method.

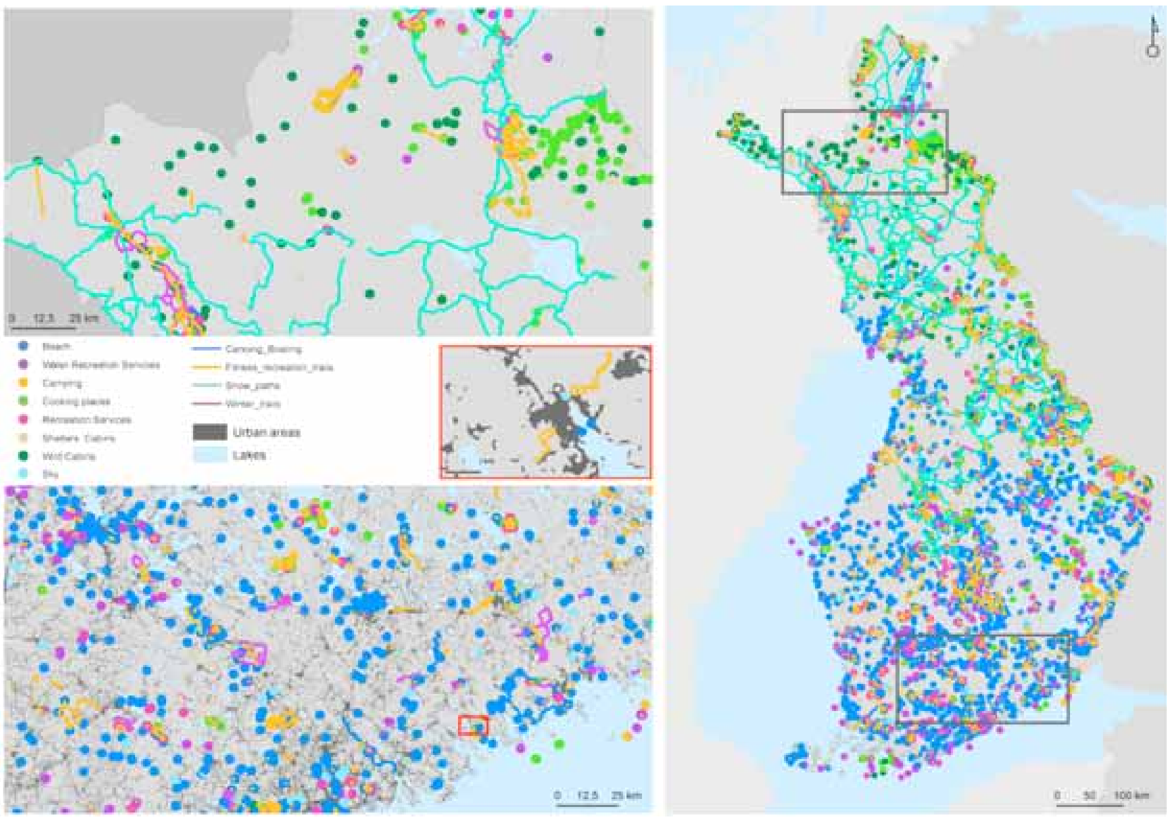


Figure 7.34. Type and location of recreational facilities ("Luonnon virkistyskäyttömahdollisuudet Suomessa (Virgis)" - copyright: SYKE).

Inverse logistic functions of distance have been defined for each category, to represent the attractivity of each facility on neighbouring population. The defined functions are based in equation (1) and are shown in Figure 7.35, and were applied separately to each group of facilities, results were added together and the final output rescaled in a 0-1 range. In the resulting map the higher the values, the closer and the more numerous the recreational facilities (Figure 7.36).

Table 7.10. Recreation facilities categories.

Recreational Services categories	Number of points	Classification
Beach	1968	Beach
Camping ground	2	Camping
Camping site	120	
Cooking / grill shelter	100	Cooking places
Fire place	1468	
Hiking centre	3	Recreation Services
Information building	75	
Observation tower	55	
Other outdoor recreation centre	5	
Parking area / site	387	
Lappish shelter / wind shelter / turf hut	951	Shelters_Cabins
Day cabin	83	
Cross-country skiing centre	173	Ski
Downhill skiing centre	77	
Guest harbour	6	Water Recreation Services
Harbour (with services)	6	
Hole in the ice for winter swimming	250	
Launching site for boats	59	
Nature harbour	54	
Pier	214	
Recreational fishing site	10	
Site for going ashore	26	
Wilderness cabin (for free use)	294	Wild Cabins
Wilderness cabin (has to be reserved)	62	
Bird watching tower	144	Not in the analyis
Rock climbing site	7	
Toilet	1869	
Boating	520	Canoing_Boating
Canoeing and rowing	490	
Hiking	7666	Fitness_recreation_trails
Cycling	1113	
Fitness	3783	
Horse riding	221	
Nature	1973	
Other	109	
Walking	420	
Wheelchair	50	
Dog sledge riding	7	Snow_paths
Reindeer sledge riding	1	
Snowmobiling official trail	818	
Snowmobiling track	3681	
Cross-country skiing, free style	4124	Winter_trails
Cross-country skiing, traditional style	5566	
Skating	23	
Snowshoeing	4	

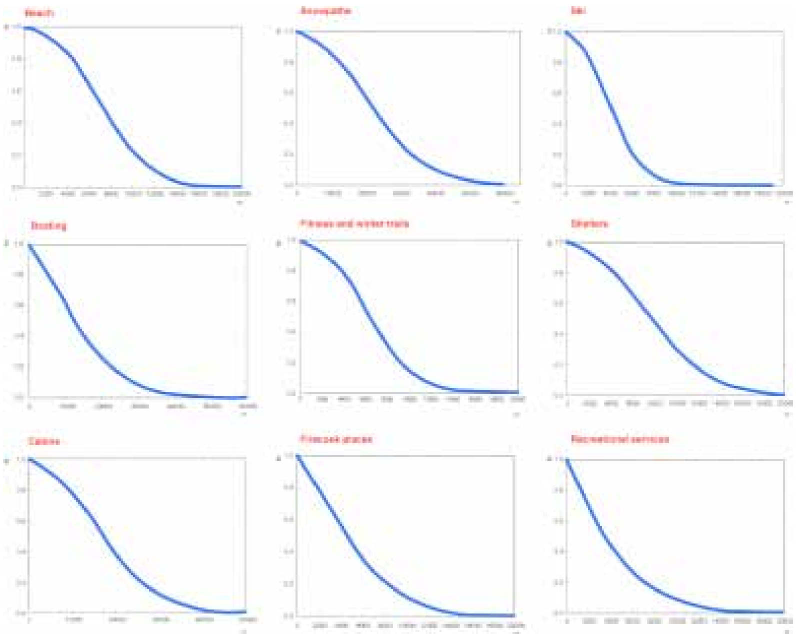


Figure 7.35. Distance functions for each recreation facility group.

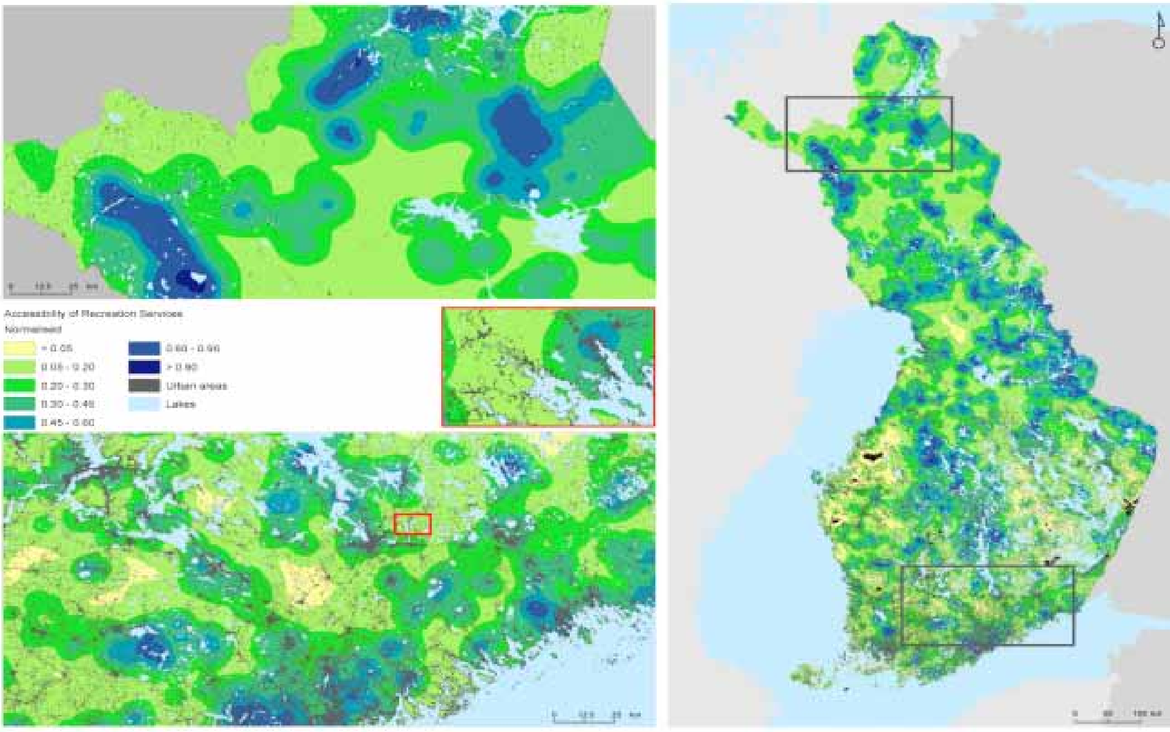


Figure 7.36. Aggregated attractivity of recreation facilities in state owned land.

Accessibility. Input data are provided by the TeleAtlas road network reclassified according to Table 7.11.

Table 7.11. Reclassification of teleatlas road network for Finland.

<table>
<tr><th>Tele Atlas Functional Road Class</th><th>Reclassification</th></tr>
<tr><td>0: Main Road: Motorways</td><td>A</td></tr>
<tr><td>1: Roads not belonging to ‘Main Road’ major importance</td><td rowspan="3">B</td></tr>
<tr><td>2: Other Major Roads</td></tr>
<tr><td>3: Secondary Roads</td></tr>
<tr><td>4: Local Connecting</td><td rowspan="3">C</td></tr>
<tr><td>5: Local Roads of High Importance</td></tr>
<tr><td>6: Local Roads</td></tr>
<tr><td>7: Local Roads of Minor Importance</td><td>D</td></tr>
</table>

An inverse logistic function has been defined for each category (Figure 7.37), to represent the degree of friction exerted by roads of that category when population in the neighbourhood is travelling to a certain location. The maximum distance taken into consideration is 100 km. The functions in figure 24 have been applied to respective road categories, results have been added and rescaled in the 0-1 range (Figure 7.38).

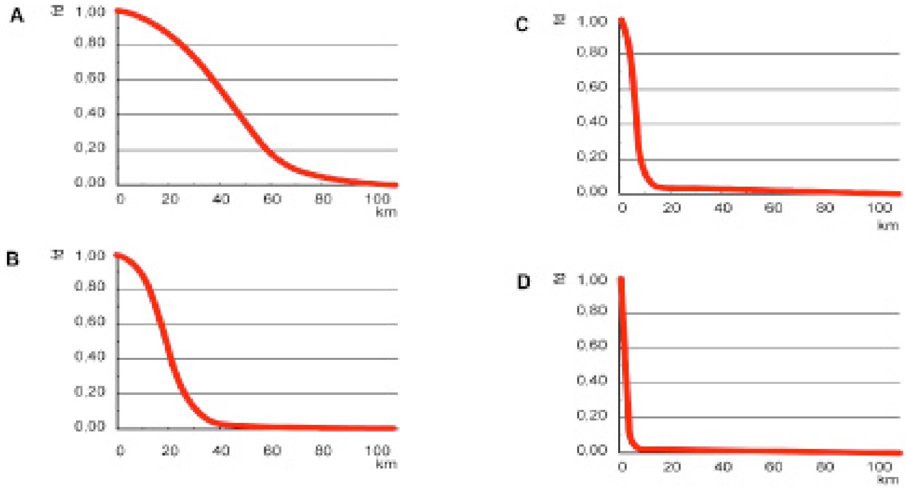


Figure 7.37. Functions assigned to road categories to represent accessibility of each location in Finland, expressed in terms of friction exerted by the road network

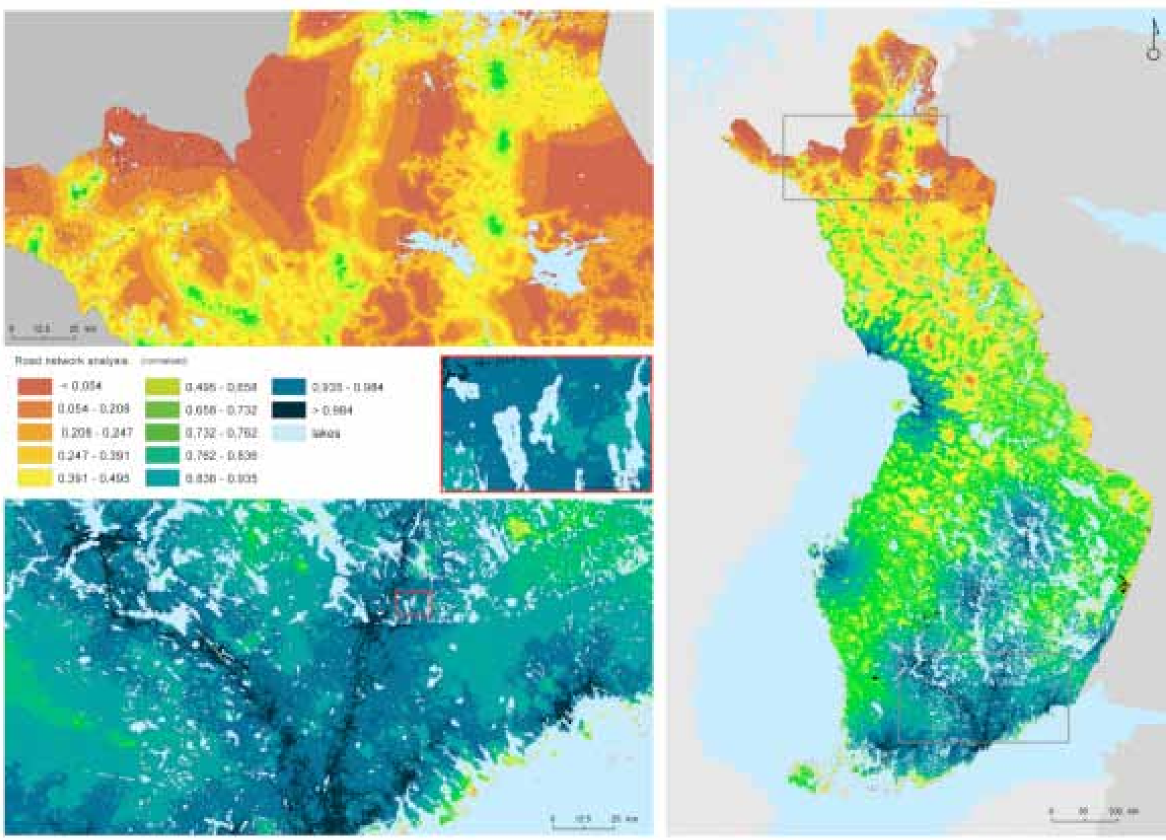


Figure 7.38. Accessibility of each location to population living in a 100 km radius.

Distance from coast (lake and sea). This is the Euclidean distance from the closest coastline.

Recreation potential index. This is the same as in Figure 7.29. All input data have been aggregated to the same resolution of the summer cottages layer (10 km x 10 km), and rescaled to the 0-1 range. The analysis has been carried out on the whole of Finland (Table 7.12), and separately only on the Southern part (Table 7.13), that is where the majority of summer cottages is present.

Table 7.12. Model to describe the distribution of summer cottages in Finland

Variable	Coefficient	StdError	t-Statistic	Probability	Robust_SE	VIF [1]
Intercept	-0,255198	0,02486	-10,265264	0,000000*	0,032629	--------
Accessibility from road network	**0,441913**	0,019687	22,446493	0,000000*	0,023833	1,446237
Recreation potential	**0,286322**	0,028905	9,905507	0,000000*	0,040131	1,777073
Attractivity of recreational services	**-0,112301**	0,017779	-6,316664	0,000000*	0,016027	1,11862
Distance from coast	**-0,3617**	0,031443	-11,503416	0,000000*	0,027947	1,294606
Number of Observations:	1936					
Multiple R-Squared [2]:	**0,288111**					
Adjusted R-Squared [2]:	0,286636					

Table 7.13. Model to describe the distribution of summer cottages in Southern Finland.

Variable	Coefficient	StdError	t-Statistic	Probability	Robust_SE	VIF [1]
Intercept	**-0,146238**	0,012937	-11,30396	0,000000*	0,014356	--------
Accessibility from road network	**0,342066**	0,010955	31,225481	0,000000*	0,012615	1,585048
Recreation potential	**0,151583**	0,013049	11,616819	0,000000*	0,015081	1,600978
Attractivity of recreational services	**-0,103708**	0,010536	-9,843225	0,000000*	0,010353	1,000576
Distance from coast	**-0,238187**	0,017393	-13,694136	0,000000*	0,019349	1,093768
Number of Observations:	3346					
Multiple R-Squared [2]:	**0,323728**					
Adjusted R-Squared [2]:	0,322919					

Results show that the selected variables explain 28 to 32% of the distribution of summer cottages and the model is driven mostly by accessibility and distance from coast. The recreation potential index is also positively correlated to cottages distribution. Availability of recreation facilities in State owned land does not seem to be in direct relation with locations of summer cottages.

This can probably be explained by the fact that there are so called "Everyman´s rights" in Finland which give the right to access any nature area independently from the ownership of the area (with some exceptions like strict nature reserves or military areas). That is the probable reason for not having correlation between state owned land and summer cottages. People use the surroundings of their summer cottages freely, with the only exception that they cannot enter other people´s yards. Common economically managed Finnish forests are suitable for many recreational activities like berry and mushroom picking, walking, hiking, bird watching etc. Comprehensive data on all available recreational services may improve the correlation.

7.2.6. Mapping recreation services at national scale: the case of The Netherlands

The methodology to map recreation services at EU level has been based on a service supply driven approach. Recreation potential is therefore mapped assuming that recreation is positively correlated to some territorial feature/indicators associated with the potential capacity of the ecosystem to attract visitors. The mapping of the Dutch case is based on a service demand driven approach because it considers the personal motives and preferences to enjoy and recreate, and therefore directly links to the definition of recreation services as the pleasure that people derive from natural or managed ecosystems. This approach is feasible because the partial availability of datasets on preferences at a high spatial resolution. The 'capacity' component of recreation fruition is therefore replaced by the actual 'preferences' resulting in the following algorithm: Fruition ~ Preferences × Accessibility.

7.2.6.1 Dutch preferences for recreation

Positive preferences have been identified in the Netherlands by Goossen and Langers (2000) for different types of land cover (Table 7.14) and for other indicators such as presence of cultural sights and differences in altitude (Table 7.14). The preferences do not only consider the 'attractive' (positive) factors but also those (negative) factors that may decrease the attractiveness of a certain ecosystem (Table 7.14), i.e. the amount of noise and recreation crowds (Vries et al. 2007). Ecosystems may be of extreme beauty but if the environment is very noisy, they will not provide a flow of cultural services.

Each indicator is spatially mapped at a resolution of 500x500 m^2. Thereafter, the percentage area of each indicator within a distance of 5 km around the central grid-cell is calculated. This buffer of 5 km, which makes an area of approximately 25 km^2, is assumed to be large enough to have a walk or a cycle tour (which is a very popular recreation activity in the Netherlands). Finally, a spatial database is made for each indicator for every 5-percentage (including 0%) of the amount of an area (Goossen et al. 2009). There is a current recompilation of recreational preferences for landscapes through the special website www.daarmoetikzijn.nl (www.myplacetobe.eu). The results are very promising and have the potential to be used in the future for a more detailed mapping of recreational services after linking the landscapes preferences to recreational ecosystem services. The website gives the internet-users the opportunity to compile their own preferred 'imaginary' landscapes. With the use of geo-referenced data the internet-user landscape preferences are compared with real landscapes. The result is a unique personalized map with a person's own appreciation of the Dutch landscapes. All preferences and personalized maps are saved in a database. From 2006 on, almost 250.000 Dutch users visited the website. The outcome from the first five years (2006-2010) of the website is used to map the average landscape preferences of the Dutch citizens, as shown in Figure 7.39.

Table 7.14. Average preferences (%) for type of land use, for increasing attractiveness indicators (1 is nothing and 100 is as much as possible) and for for decreasing attractiveness (1 is not annoying and 100 is very annoying)

Land use	Average preference (%)
Forest	30
Heath lands, sand drifts and dunes	18
Natural grassland, marsh and reeds	14
Agricultural area with panoramic view	12
Half-open agricultural area	17
Enclosed agricultural area	17
Sea and large lakes	19
Ditches, streams, rivers and ponds	13
Businesses and industry areas	4
Inhabited areas	12
Indicators	**Average preference (%)**
Sights to visit	48
Differences in altitude	37
Indicators	**Average preference (%)**
Visual disturbances	59
Noise	64
Recreation Crowds	38

Figure 7.39 shows the degree of landscape preference for recreation. Regions with a high score (green areas) are more preferred than regions with a lower score (dark orange areas). This map gives insight in the potential demand for recreation in a region. However it needs to be analysed when considering the ecosystems linked to those landscapes. In addition, the map does not give information about the recreational flows. Therefore the accessibility and recreational infrastructure (the supply according to the Recreational Opportunity Spectrum) have still to be included, and combined with data about potential use.

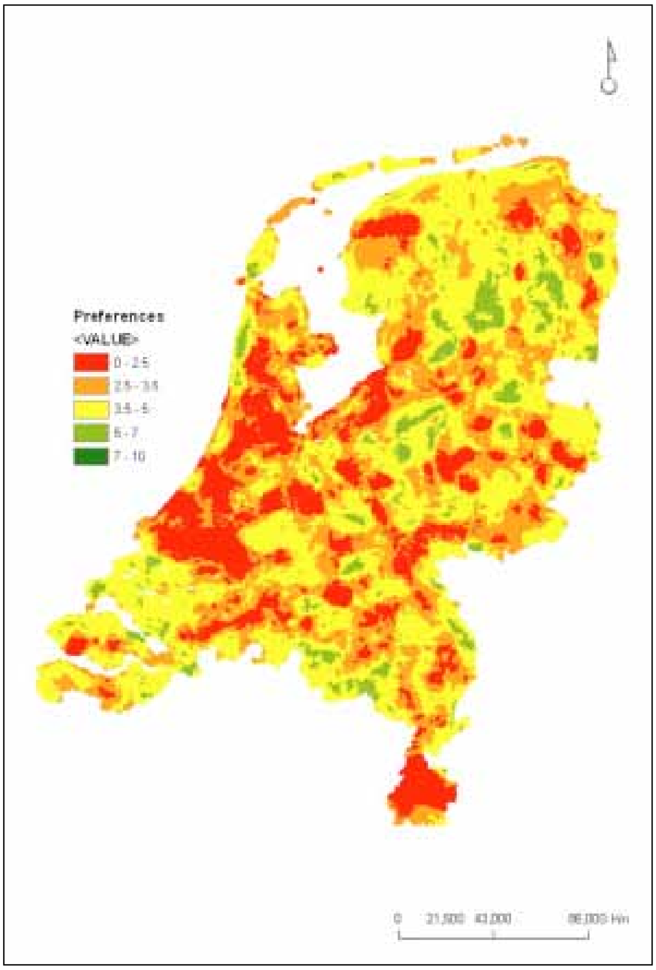


Figure 7.39. Landscape preferences for recreation of Dutch citizens.

7.2.6.2 Mapping the most popular dutch recreational activity: cycling

To map the potential use of ecosystems for recreational cycling, the following data are needed :

- Cycling infrastructure: From the Topographical Map with a scale of 1:10.000 a cycling network was developed, consisting of a combination of roads (with permission to cycle) and special cycle paths.
- Geographical distributions of Dutch citizens: the geographical distribution of citizens was mapped by using the distribution of number of inhabitants of the (geographical) smallest postal-code areas and number of beds in tourist accommodations (hotels, campsites, bungalows and etcetera) in that postal-code area. Data of Statistics Netherlands (2008) were used to count the overnight stays in tourist accommodations of Dutch and foreign tourists. These data are available for every type of accommodation at specific Dutch Tourist area level and they were spread evenly among the accommodations in the postal-code areas in that level.
- Cycling preferences of Dutch recreationists: to calculate the cycling use of the network, the participation rate and frequency for recreational cycling was used. Because The Netherlands is a multicultural society, differences in participation and frequency level occur between the autochthonous Dutch and the non-western allochtonous Dutch (Vries et al. 2004). With a basic network analyses, a calculation was made on how far the cyclists could travel through the network. Research shows (Goossen 2009) these preferences. The average recreationist cycle 1.5 hours (24 km), which is a maximum duration of 45 minutes (vice versa) per day.

An additional step was necessary to sum up the bicycle potential in those regions that can be visited from multiple points of departure (Figure 7.40, Figure 7.41). For the whole of The Netherlands the potential use of the network by recreation cyclists is calculated (Figure 7.42).

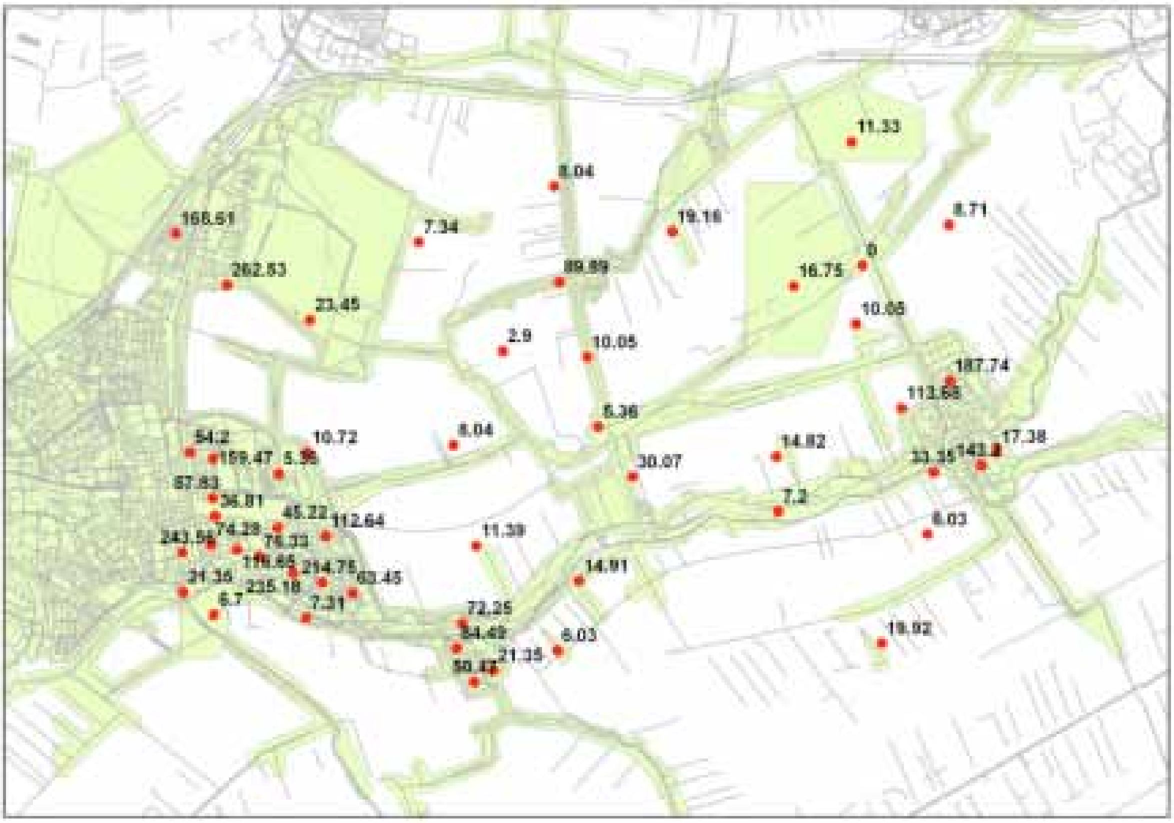


Figure 7.40.Areas that are accessible by bike. Red points show the points of departure, the numbers show the potential cyclists, and the lines show the bicycle network.

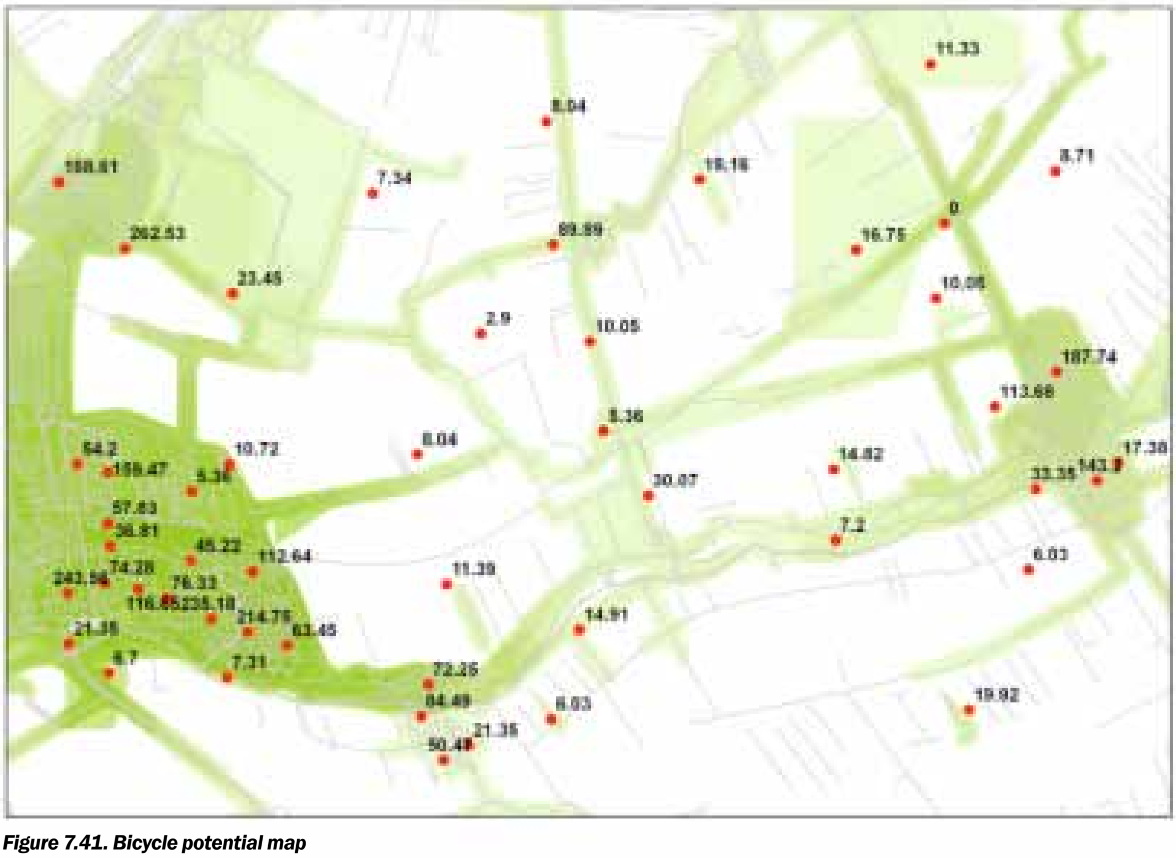


Figure 7.41. Bicycle potential map

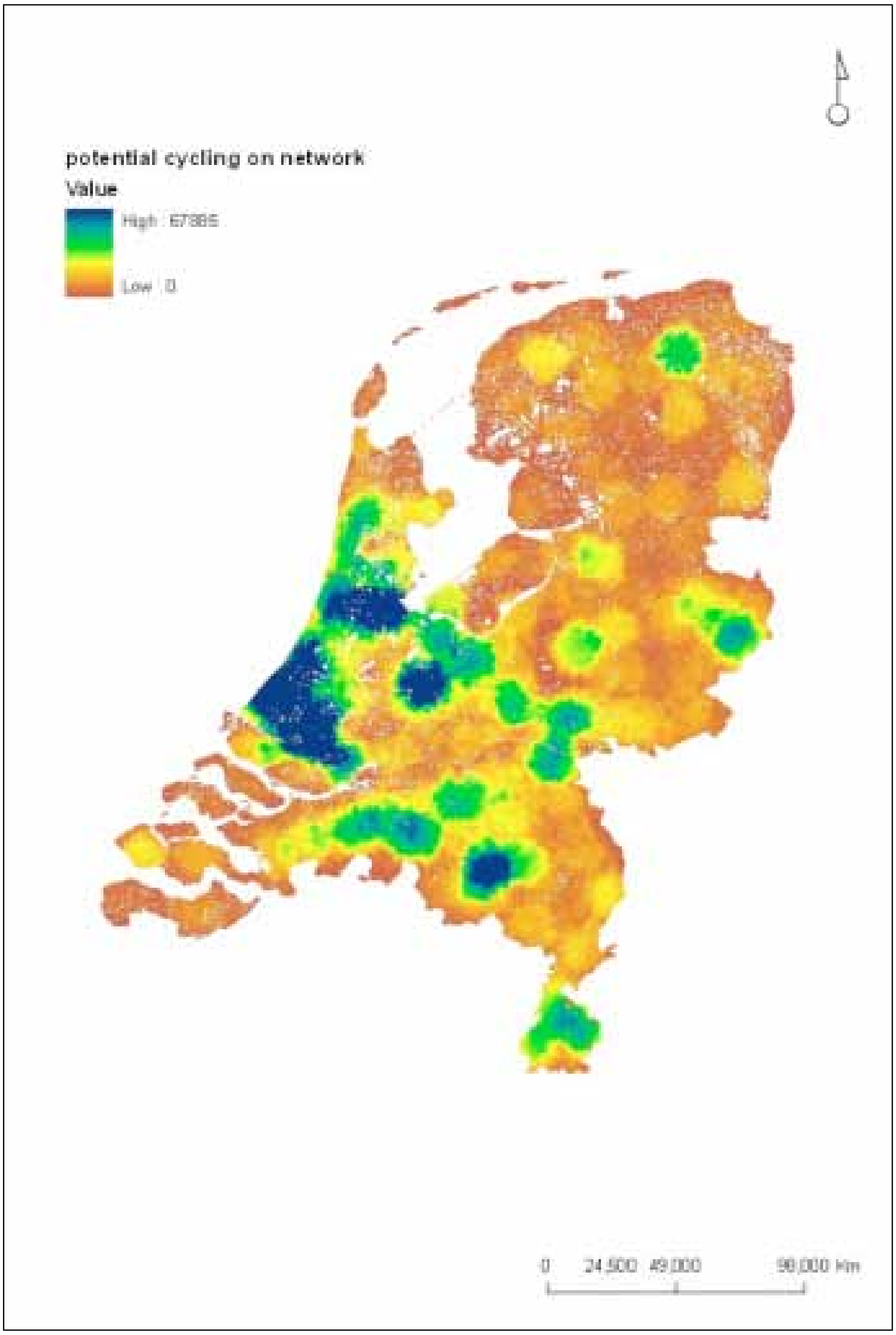


Figure 7.42.Amount of potential cycling pressure (use of recreation cyclists on roads and cycle paths).

7.2.6.3 Recreational opportunity index

The final ROS (Figure 7.43) was obtained by combining the indicator on preferences of recreation services (Figure 7.39) and the zoning of The Netherlands on the basis of the potential pressure of recreation cycling as shown in Figure 7.42. As in the case of the EU, the result provides information both on the quality of recreation provision and on its accessibility.

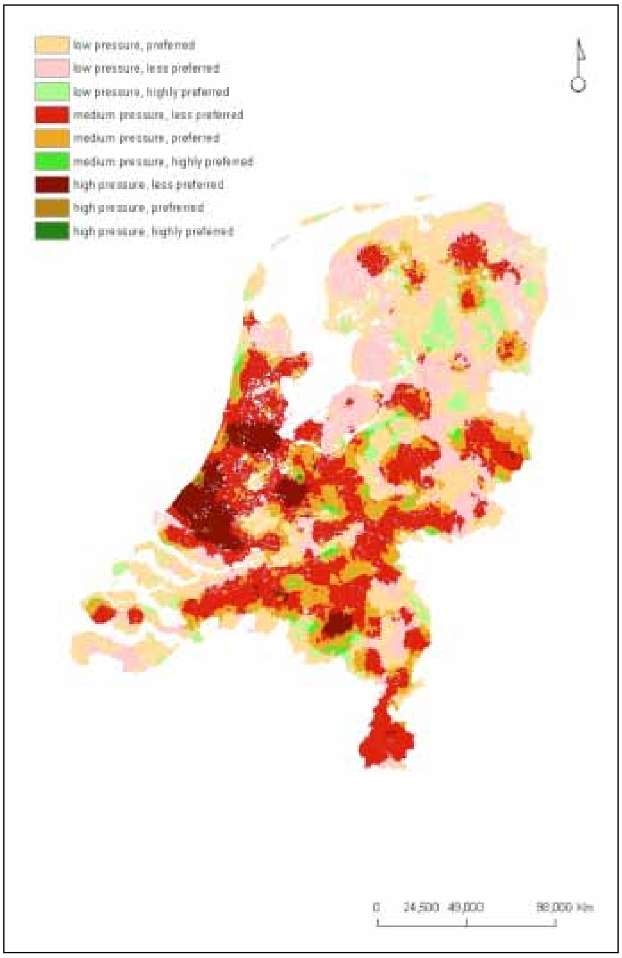


Figure 7.43. The Recreation Opportunity Spectrum for cycling for The Netherlands

7.3 Forest ecosystem services

7.3.1. Introduction

Ecosystems and the associated biodiversity offer a range of goods and services that humans need in order to earn an income and secure sustainable livelihoods. In addition, biodiversity contributes to ecosystem processes that maintain continuous production of ecosystem services. Recognising the links between biodiversity and ecosystem services would help stakeholders to avoid biodiversity losses that lead to unacceptable ecosystem services losses. For implementation of the new EU Biodiversity Strategy it will be important to take advantage of the results of the many existing studies of the valuation of biodiversity and ecosystem services in tandem with the Environmental Valuation Reference Inventory (www.evri.ca) and FNU database over demand for forest recreation and biodiversity value (Zandersen et al. 2007). The results of the studies can be brought together into a broader context in order to use them to model the demand for the biodiversity and ecosystem services within the policy context presented to map services and trade-offs.

In this PRESS-case study on forest ecosystem services we have explored different approaches to mapping forest features and explicit ecological functions and services to the economies of the EU, at the European, National and Regional scale. Each of the approaches combines different techniques of data collection (e.g. field survey, LIDAR) and data handling (GIS, dynamic computer models), and combines data from forest stand level with data from higher spatial scales. In some areas of the EU, forest is still quantitatively a major land use category; in other countries it covers less than 10% (Figure 7.44).

Figure 7.44. Proportion of total forest from total land area (% at 1km resolution).

The focus has been to collect maps of forest features and human activities in forest areas and evaluate them for potential to be translated into ecosystem service maps. A small selection of maps and background data found is shown and discussed here.

7.3.2. Mapping at the european scale

7.3.2.1 Objectives and management types in forestry

Figure 7.45 provides a European scale view of Forest Management Approaches (a new methodological framework and its applicability to European forestry (Duncker et al. in press).

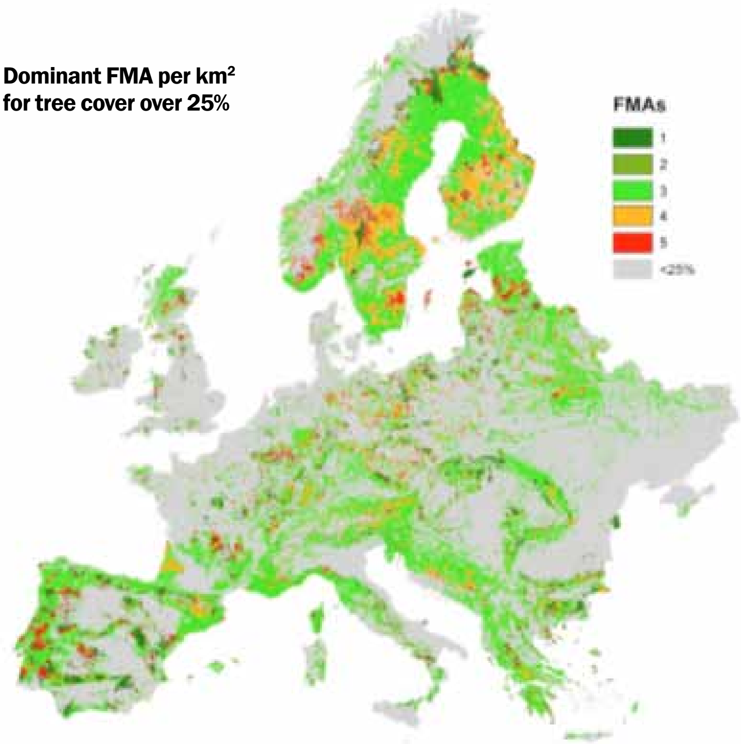


Figure 7.45. Dominant forest management approaches (1= nature reserves; 2=close to nature; 3= combined objectives; 4=intensive even aged; 5=short rotation forestry).

The choice between different forest management practices is a crucial step, in short- as well as long-term decision-making in forestry and when setting up measures to support a regional or national forest policy. Some conditions such as site, forest health status and social and economic demand are often pre-determined, whereas operational processes such as species selection, site preparation, planting, tending or thinning are factors that can be altered. In principle, a forest management approach (FMA) provides a structure for decision-making including a range of silvicultural operations throughout the stand development phases. Five forest management approaches (FMAs) representing a gradient of management intensity have been described using specific sets of basic principles which enable comparison across European forests. Despite being arranged along an intensity gradient, the forest management approaches are not considered to be mutually exclusive. In contrast, the range of options allows for an increasing degree of freedom in potential silvicultural operations. Thus, the management objective could emphasize the economic dimension at the expense of the environmental and social dimensions of sustainability. As forest ecosystem services are thus affected, the five forest management approaches have implications on all three dimensions of sustainability. The approach to forest management on a location can be classified according to management actions and decisions made during the management cycle. By placing the management goal and decisions along a gradient of intensity of intervention with the natural process, five classes of forest management approaches are identified:

1. Nature reserves: No interventions
2. Close-to-nature forestry: The natural process is mimicked whenever possible
3. Combined objective forestry: There are both economical and ecological concerns
4. Intensive even-aged forestry: Timber production
5. Short rotation forestry: Biomass production

The categories can be matched with respectively dominant performance of regulating (1, 2), cultural (2, 3) and provisioning (4, 5) ecosystem services.

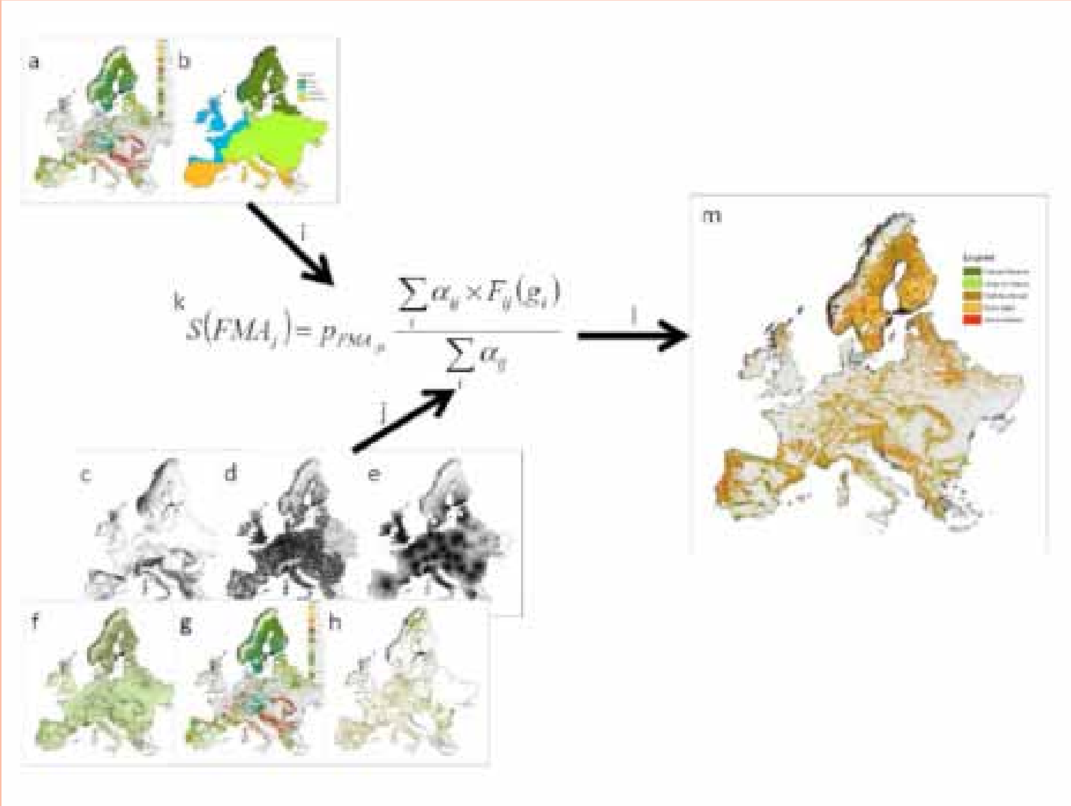
The various FMA's occur throughout Europe. There is no large scale main differentiation; it is quite well in line with the forest characterization of Europe. However, some regional characterization is apparent. Scandinavia is characterised by large areas of FMA IV, the intensive even aged forest, with relatively large patches of FMA V (the short rotation production) and FMA I (the unmanaged forests). The latter are generally restricted to the high altitude and high latitude forests. The western central European countries show a highly fragmented forest landscape with a mix of all FMAs. The regions of Aquitaine, France and the north of Catalonia, Spain are characterised by a relatively large area of FMA IV, the intensive even aged forest, while Portugal is characterised by short rotation forestry, in a landscape dominated by multi-functional forests. Spain shows some large reserves (FMA I) dominated areas). Towards the east in Baltic States and Belarus, the multi-functional management dominates, with scattered areas of short rotation production. Towards the Carpathians, the multi-functional management dominates as well, now with scattered areas of reserves. This is also the case in Bulgaria, Greece and Italy. The Western Balkans are also dominated by multi-functional forests, with some areas of intensive even-aged forests.

Table 7.15. A list of the 12 critical decisions and the basic principles used to distinguish between five forest management approaches (FMAS) as well as the main silvicultural systems associated with each FMA.

Decision	**Basic principle by FMA**				
	Intensity scale				
	Passive "Unmanaged forest nature reserve"	**Low "Close-to-nature forestry"**	**Medium "Combined objective forestry"**	**High "Intensive even-aged forestry"**	**Intensive "Short rotation forestry"**
Selection of tree species (Naturalness of tree species composition)	Only species characteristic of the potential natural vegetation (PNV)	Native or site adapted species	Tree species suitable for the site	Tree species suitable for the site	Any species (not invasive)
Tree improvement/ Genetic engineering	No	Not genetically modified or derived from breeding programmes	Trees from tree breeding but not genetically modified	Trees from tree breeding but not genetically modified	Genetically modified or breeding
Type of regeneration	Natural regeneration / natural succession	Natural regeneration (planting for enrichment or change in tree species composition)	Natural regeneration, planting and seeding	Natural regeneration, planting and seeding	Planting, seeding and coppice.
Successional elements	Yes	Yes	Temporarily	No	No
Machine operation	No	Extensive	Medium	Intensive	Most intensive
Soil preparation	No	No (only to introduce natural regeneration)	Possible (mainly to promote natural regeneration)	Possible	Yes
Fertilisation / Liming	No	No (only if devastated soil)	No (only if devastated soil)	Possible	Yes
Application of chemical-synthetic protective agents	No	No	Possible as a last resort	Possible	Possible
Integration of nature protection	High	High	High	Medium	Low
Tree removals	No	Stem (solid volume)	Stem and crown (solid volume)	Up to whole tree	Whole tree and residues
Final harvest (and main silvicultural) system	No	Mimics natural disturbances, Single Stem Selection, Group Selection, Irregular Shelterwood	All possible, Seed Tree , Strip Shelterwood, Group Shelterwood Uniform Shelterwood Coppice with standards	All possible, clear-cut (long rotation) preferably used	All possible, Coppice, Clear fell (shorter rotation)
Maturity	No intervention	Long rotation length ≥ age of max. MAI	Med. rotation length ≈ age of max. MAI	Short rotation length ≈ age of max. financial return (low interest rate)	Shortest rotation length ≤ age of max. MAI or ≈ age of max. financial return (high interest rate)

BOX 3: Method to develop a forest management map of European Forests (Hengeveld et al. in press)

Management decisions for a specific forest stand is influenced by many different factors. They have been divided in four different categories: biotic conditions, abiotic conditions, socio-economic conditions and political situation. The biotic component includes stand characteristics like stand area, tree species composition and stand structure. Abiotic conditions include site factors like climate, topography and soil. Socio-economic conditions include the wood market, extraction costs, transport opportunities, specific goals or interests by the forest owner, subsidies and recreation pressure. Political factors include policies, regulations and restrictions on forest operations issued at various levels of organisation. For each of these four categories we identified at least one European-wide spatially explicit dataset that corresponds to a factor that will influence the owner's decision. In total we have selected eight factors.



Mapping procedure

For the biotic conditions we used the dominant species (a) in each pseudostand for a given FMA (see table 1). To incorporate regional differences in species use, these applicabilities were assigned based on four biogeographical regions (b). For the abiotic conditions, we selected the slope (c) as important decision variable. Two types of proximity maps were used as a proxy for socio-economic conditions. Small scale proximity (d), defined as distance to cities of at least 25,000 inhabitants, represents recreation pressure. Large scale proximity (e), i.e., to cities of at least 750,000 inhabitants, is considered a proxy for distance to major wood-working industries. Additionally we used the percentage of the pixel covered by forest (f) and stand area (g) as proxy for the economic feasibility of intensive forestry. For the political framework we used a map with the Natura2000 sites (h), as indication where operations will be more restricted likely to be influenced by conservation policies.

The strategic management choices of where to conserve nature, and where to produce wood are often done locally at the management unit, or nationally at assigning reserve areas. Mapping these areas at the European scale provides the basis for discussion about strategic choices across Europe in view of possible future trade in ecosystem service targets between EU Member States. This policy option may arise from economic considerations of most efficient ecosystem service management across the EU.

7.3.2.2 Mapping provisioning services

Figure 7.46 and Figure 7.47 are base maps for the timber production service, stocks and utilization rate. Detailed statistics are available for many countries, as well as timber production models, at country and EU level.

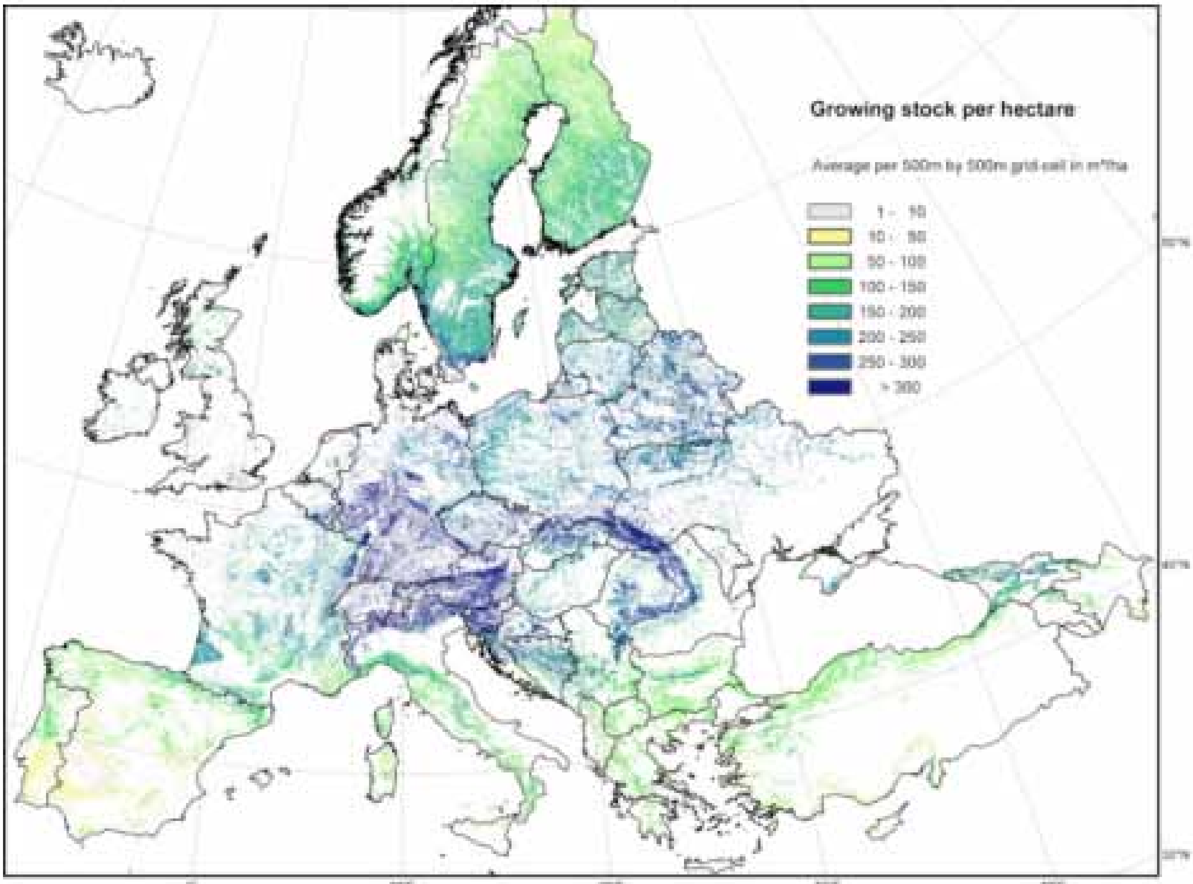


Figure 7.46. Growing stock in m³ per hectare per 500m x 500m grid cell.

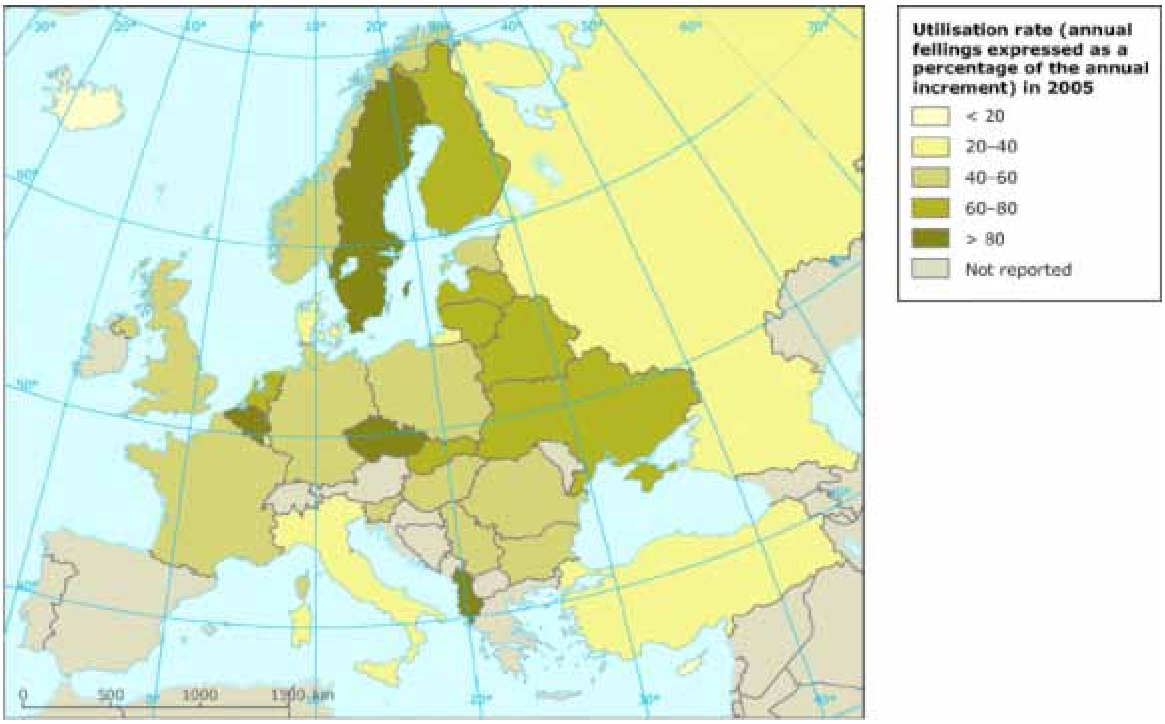


Figure 7.47. Utilisation rate (% of annual increment) (source: EEA, 2009).

7.3.2.3 Forest ecosystem diversity indicators in european forests

The biodiversity features of forests are available in various map formats. There are maps of Forest types (Figure 7.48), Dominant Species (Figure 7.49) and Age distribution (Figure 7.50).

Correlated with these forest features are biodiversity aspects, at various geographical scales. These features, and many other structural variety aspects which are documented in many of these forest types, and are related to the dominant species and age classes, can be the basis for ecosystems services scores. The data at site and regional scale behind these maps could be combined at some regional and national levels, and aggregated to EU level, to make forest (structural) biodiversity maps which could be used to link with some regulating and cultural service (based on ecosystem service case studies such as in the COPI and TEEB databases; Braat and Ten Brink, 2008; Ten Brink et al. 2009; TEEB 2010).

7.3.2.4 Mapping recreation in european forests

Current approaches to modelling the recreational value of forests are often based upon regression models which relate forest inventory data to public preferences for different forest stands. An alternative method is presented by Edwards et al. (2009). Figure 7.51 illustrates what a map based on this methodology could look like.

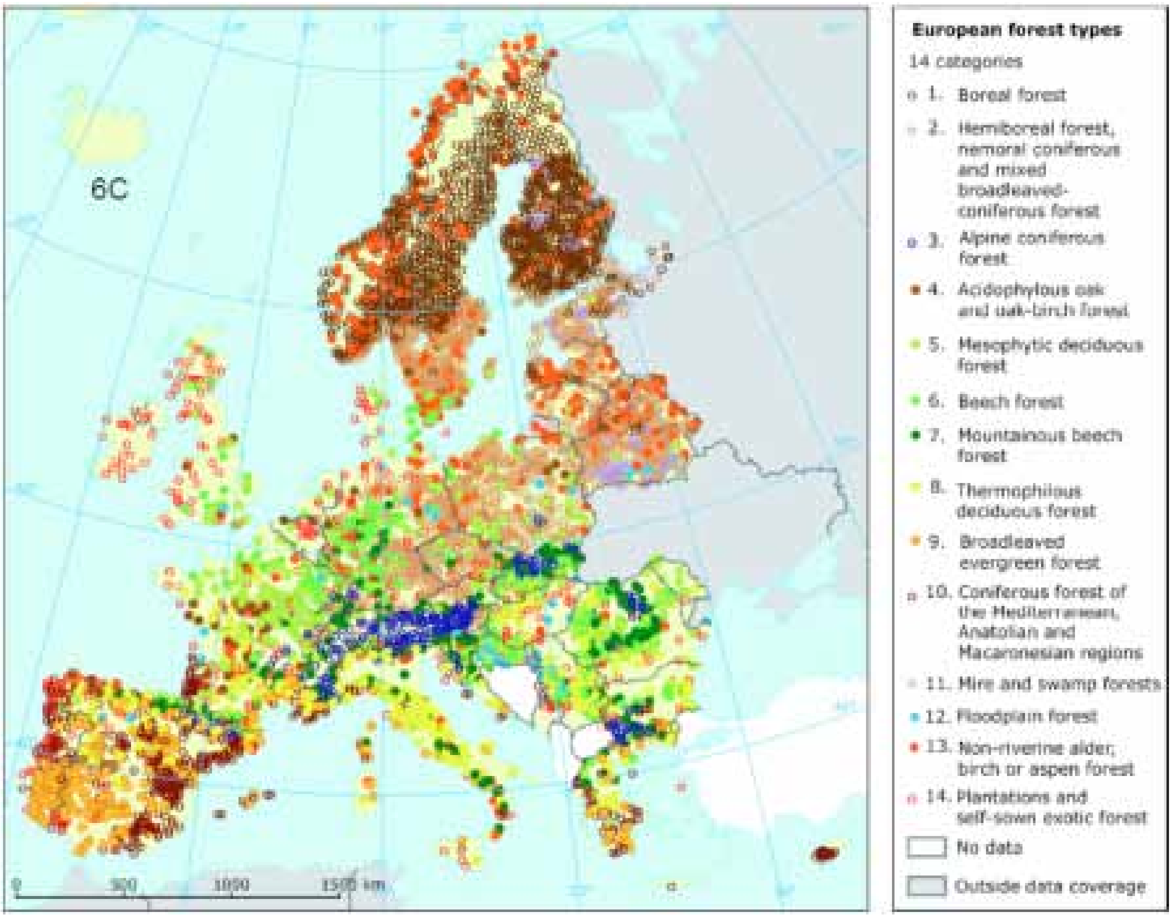


Figure 7.48. European forest types (source: EEA, 2009).

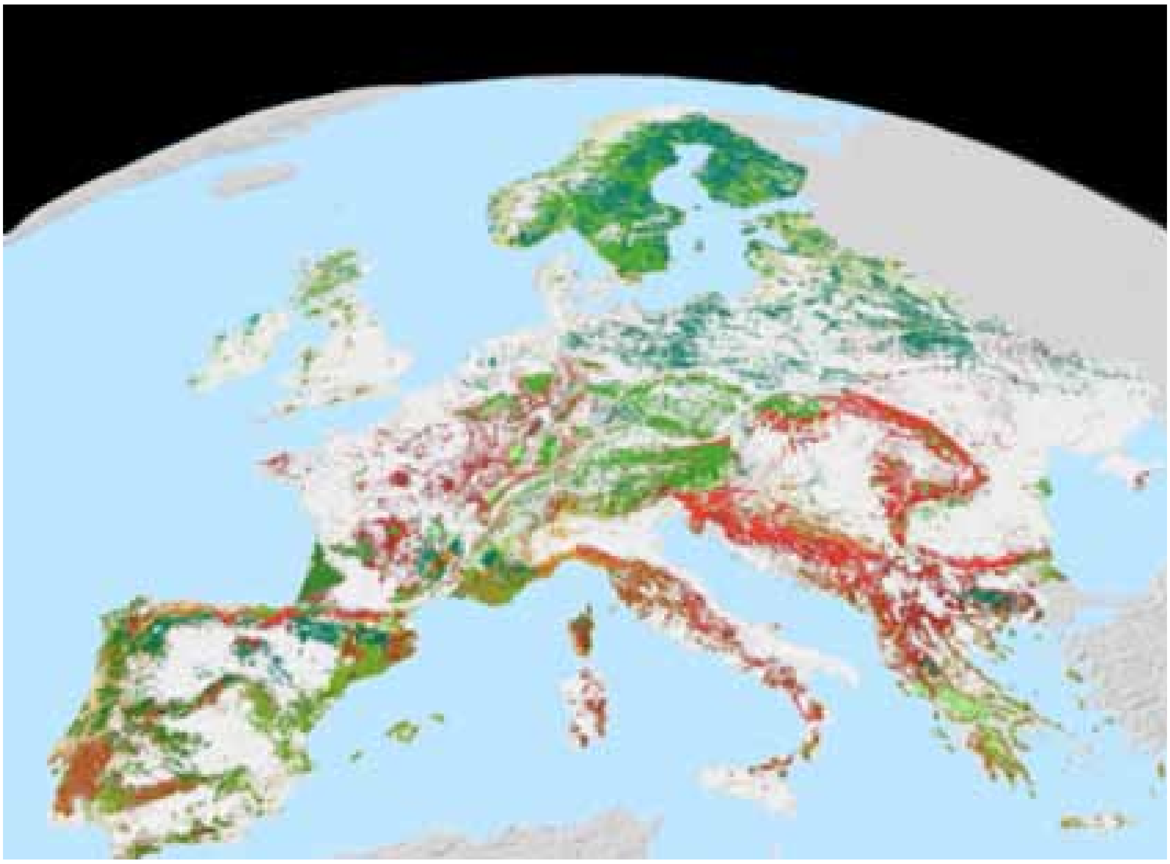


Figure 7.49. Dominant species in European forests (source: Brus et al, in prep).

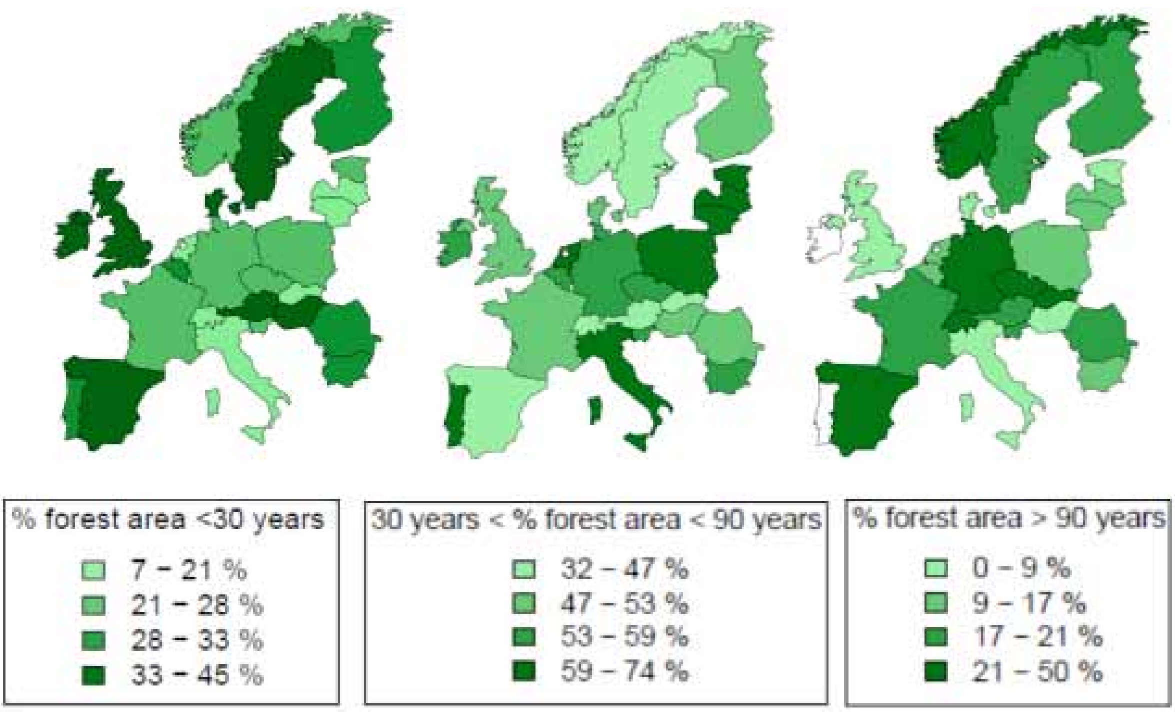


Figure 7.50. Distribution of age classes of European forests (source: EFI).

Figure 7.51. Recreational attractivity of forest in Europe (Schelhaas and Hengeveld; 2011)

The method is less resource intensive and with potential to be applied across large spatial scales, based upon a typology of forest management alternatives with common silvicultural characteristics across Europe. Recreational scores are derived for each of 20 forest stand types in a given European region through the use of Delphi surveys supported by a literature review of European forest preference research. Forest growth simulators are then used to forecast changes in the area of each forest stand type under a given scenario, and hence to the total recreational value of the forests in the region. A stepwise description of the approach is provided, and its application is illustrated using indicative recreational scores to assess the impacts of two contrasting levels of implementation of the Natura 2000 policy on the recreational value of conifer forests in the United Kingdom. The discussion considers the opportunities and risks associated with use of the approach in a European-wide context to guide policy decisions and planning. The data have been mapped at NUTS scale to generate a EU overview, with relative scores, dark green being very attractive to outdoor recreation activities, reflecting the most species diverse, old stand forests, and light green reflecting short rotation single species forests.

7.3.3. Forest ecosystem services at the national scale

In this section we briefly look at efforts to describe and map ecosystems services of forest in the Netherland as an example of the national scale, partly based on a review of Hoogstra and Willems (2005). Until 1900, the Dutch forests provided only a few ecosystem services which were relevant to society at large: wood production (provisioning service), stabilisation of sand dunes (regulating service) and soil improvement (supporting / regulating service). In addition, for a small group of wealthy estate owners, forests were a sign of prestige (cultural service) and important for hunting (cultural and provisioning service). Since the beginning of the 20th century forest ecosystem services have been expanded. Initially the "habitat / supporting" service became more and more important, as the nature conservation movement became a serious stakeholder, and after WWII also the recreational (cultural) and environmental services (mostly regulating services) of forests were acknowledged and stimulated. The Long-term Forestry Plan of 1984 officially recognized the functions (as ecosystem service were still called then) outdoor recreation, wood production, natural values and landscape quality. In the 1993 Forest Policy Plan environmental functions (regulating services) were added.

Provisioning services. Annually between 1.1 and 1.4 million m^3 of wood is harvested in the Netherlands. This is only 7-10% of the domestic wood consumption, imported from other European countries or from tropical countries. Sawn softwood imports come mainly from Europe, half of the sawn hardwood is imported from Malaysia. The Netherlands is nearly self-sufficient in paper production. Non-timber forest products only play a minor role; only Christmas tree production and horticultural greenery are of commercial interest. The collection of most non-timber forest products such as fruits or mushrooms are mainly part of recreational activities. Hunting provides on average only 7% of the income of forest owners; most Dutch people are not in favour of hunting.

Cultural services. Recreation is the most important active use of forest and nature in the Netherlands. The results of a national survey showed that the Dutch place a high value on the recreational function of forests (Figure 7.52). As a nation, at this moment around 200 million trips are made to the forest each year; an average of half a million a day. Three-quarters of the population go for a walk in the forests now and again, on average about twice a month. Older people and those who live close to the forest visit forests more frequently. At this moment, about 82% of the forests are open to the public (Figure 7.53). About 40% of the Dutch are of the opinion that there are not enough forests in their living environment. In the south-western part, in the western part and in the northern part of the Netherlands even 60% of the population feels that there should be more forests. Forests are also of increasing importance is the improvement of the living environment of housing areas. In some areas, the vicinity of forests adds up to 10% to the value of real estate property, amounting to billions of euro in total.

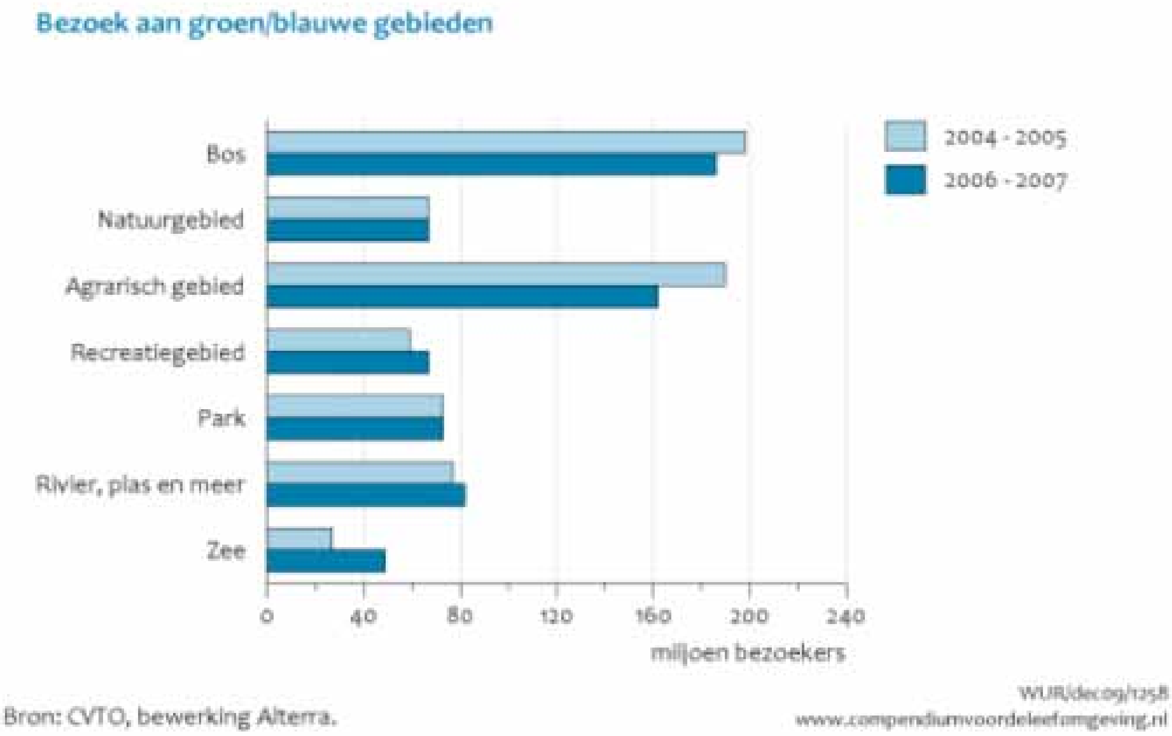


Figure 7.52. Visits to green – blue area in The Netherlands (bos=forest)

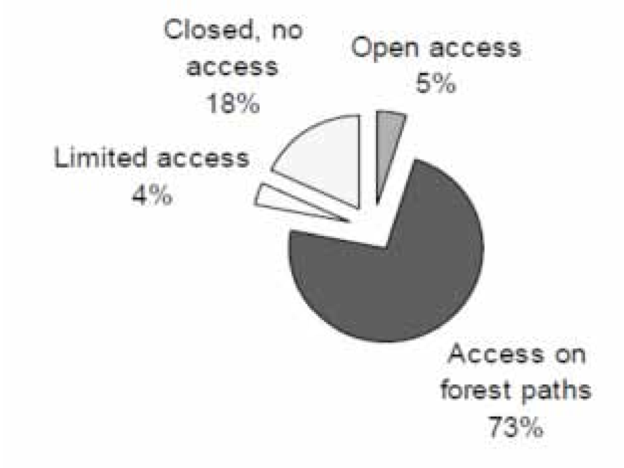


Figure 7.53. Accessibility of forest in The Netherlands

Habitat (supporting) services. Nature (habitat for species) functions of forests are highly valued. This is reflected by the fact that 25% of the total forest cover has a protected nature status and 14% of the non-protected forests are owned by private nature conservation organisations. The commitment of the Dutch population to nature (including forest) is determined annually on the basis of (1) interviews and (2) statistics on the support of people to nature conservation, e.g. through membership of nature conservation organisations and voluntary work in nature conservation.

Regulating services (Environmental functions). Forests provide different regulating services (environmental functions), e.g. purification of water and air, shelter against wind and rain, provision of shadow and coolness. As regards carbon dioxide absorption, in 2000 the Dutch forests absorbed in total 68 million tonnes of carbon. Per ha, the net sink is 2.2 tonnes of CO_2 per year.

7.3.4. Forest ecosystem services at the regional scale

7.3.4.1 Protection against natural hazards

The protection of people, houses and infrastructure against natural hazards by forests has been mapped in different alpine countries. The methods are being harmonized through European research and development projects (PCRD, Interreg). The rationale is to integrate at least 3 maps: hazard extension (from geomorphological models), human stakes (with a rating of the vulnerability), and forests having a mitigation role (evaluated also through models). In the case of protection against rock falls, we get the following kind of map (Figure 7.54).

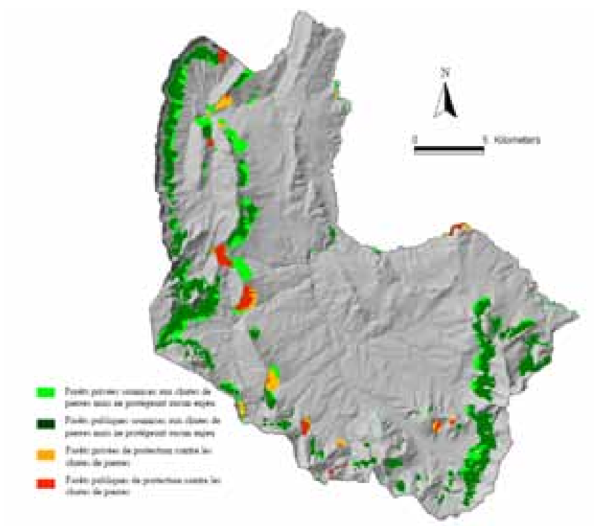


Figure 7.54. Natural hazard protection map in south eastern France; 1 private forests having an interception role, but without stake protected; 2 public forests having an interception role, but without stake protected; 3 private protection forests (interception and stakes protected); 4 public protection forests (interception and stakes protected).

This example is typical of socio-ecological service (SES) mapping, which will generally combine an ecosystem map (here an area and the forests mitigating it) with a sociosystem map (human stakes). This represents the meeting of a ecosystem "offer", as a potential service, with a "user" from the socio-system. Between the offer and the user, may be several physical, economic or social filters/obstacles (e.g. accessibility) to be taken in account for the quantification and the valuation of the service.

7.3.4.2 Aerial lidar and multispectral satellite scenes

In this example of protection forests, we see the necessity of a fine description of the ecosystems and their geomorphological support. This ecosystem map is a basis for any Socio-Ecological Service mapping. At a national scale, the habitat maps may be sufficient or, for volume production purposes, the types of sites and stands, derived from national forest inventories. But at the local scale, these general descriptions are not sufficient, and do not allow an economical valuation of the protection or provision services.

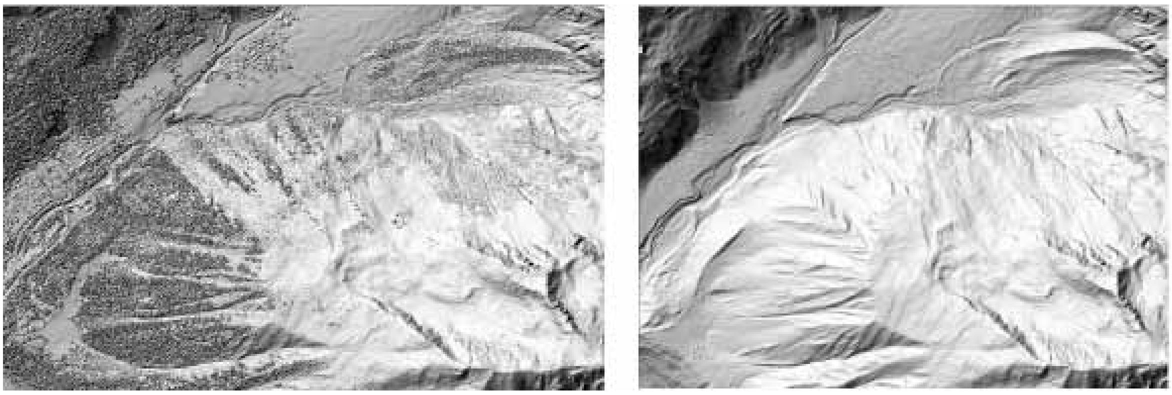


Figure 7.55. The two layers derived from lidar data : DSM (digital surface model) and DEM (digital earth model).

This is the reason for the fine description of geo-ecosystems through improved remote-sensing tools: air-born LIDAR, and satellite multispectral satellite scenes, which can be acquired at a large scale. The LIDAR is a laser scanning method, providing up to 20 (x, y, z) points m^{-2}., from which at least two layers may be extracted : a fine Digital Earth Model (DEM), with precision < 1m, and a digital Surface Model (DSM), giving the envelope of the tree crowns with the same fine definition. Hence, with tree recognition software, a very precise description of the geo-ecosystem (trees, ground) is derived. The access costs may also be assessed finely through road-networks recognition and characterization (Figure 7.55).

7.3.4.3 Timber

For production purposes, at least 2 levels of "service" have to be distinguished, from potential to actual: the potential sustainable harvest, and the quantities actually harvested (in m^3 and €) under technical, economical and social constraints. In the mountainous context, these constraints are enhanced and foster research, but the problem of taking theses constraints into account is general (Figure 7.56).

A first assessment of the sustainable level may be the potential increment, deduced from the site, as a long term potential. This is rather easy to process and to map at large scale level, from site models derived from geomorphology. A finer assessment could be the present increment, drawn from national forest inventories. A still better assessment will take into account the level of maturity and stocking of the stand, compared with usual silvicultural norms. This has been done in France, but not mapped, the results being given only by regions (21 for France). Local mapping would be problematic, but perhaps possible (if blurred at 1 km x 1km), but also in cooperation with NFI.

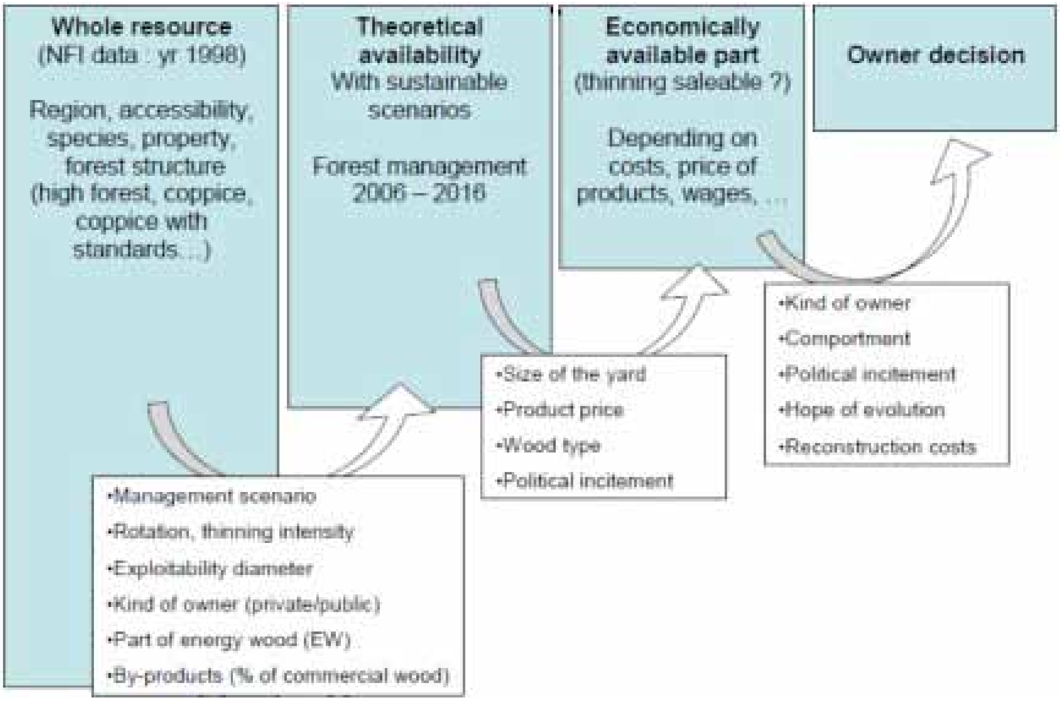
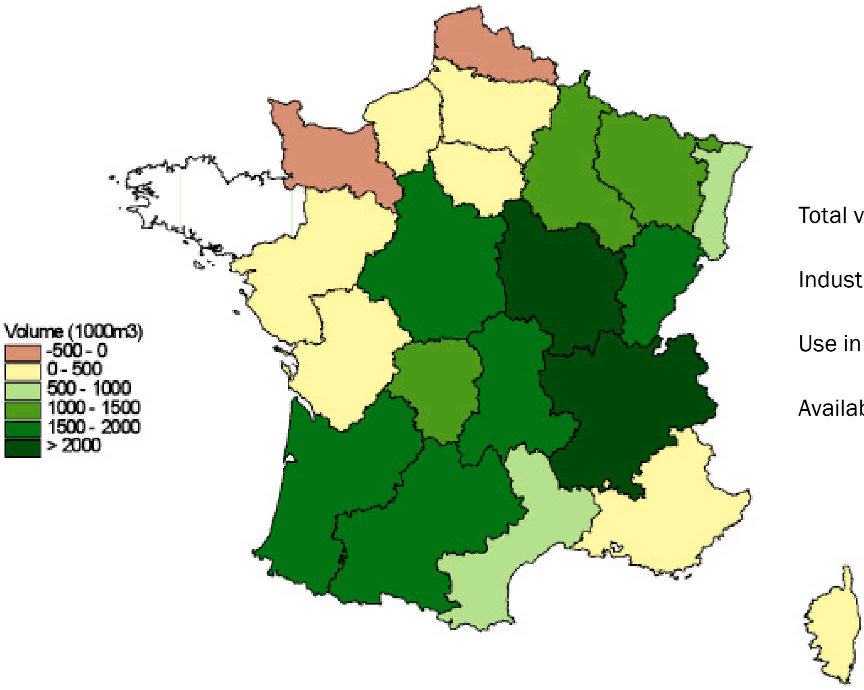


Figure 7.56. Type of volume estimations.

Combining the statistics on wood harvesting at global levels, and local models of the balance between harvesting costs and timber value, the actual harvest could probably be spatialised and mapped with an adapted blurring. That should also be done with NFI cooperation. These harvest models may be used for simulations, by modification of the parameters of constraints. But it is then necessary to work at the fine scale priori to derive a map at the coarse scale. This is because an average is not meaningful to calculate the costs, only the accessible stands will be harvested and the others will remain. This point is important to avoid global overestimations in scenarios, leading to inappropriate industrial investments and excessive pressures on forests.

In Figure 7.57 the volumes presently used in each region have been derived from potential volumes, to provide a potential availability.



Total volume	50.4
Industrial use	12.4
Use in households	17.4 to 25.3
Availability	12.7 to 20.6

Figure 7.57. Wood available for new purposes by administrative area (10^3 m^3 $year^{-1}$)

7.3.4.4 Carbon sequestration

Forests provide different regulating services e.g. purification of water and air, shelter against wind and rain, provision of shadow and coolness. As regards carbon dioxide absorption, in 2000 the Dutch forests absorbed in total 68 million tonnes of carbon. Per ha, the net sink is 2.2 tonnes of CO_2 per year. Figure 7.58 illustrates the range of Carbon values across this Natura2000 area in the centre of the Netherlands.

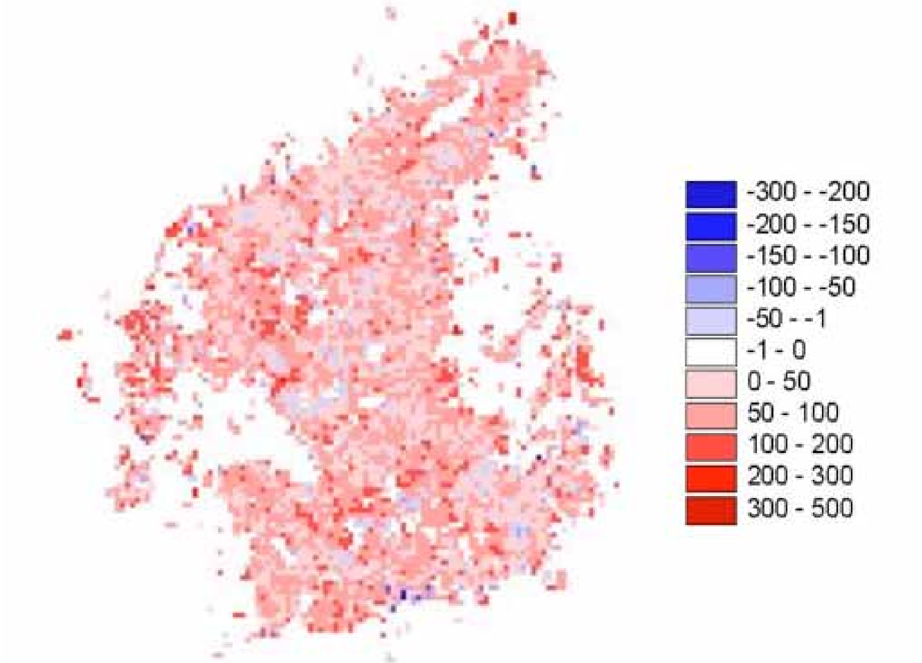


Figure 7.58. Carbon fluxes at the stand level (negative is source, positive is sink, in mg c ha^{-1}) for the Veluwe as assessed with the co2fix model (Schelhaas and Nabuurs, 2001).

7.3.4.5 Forest biodiversity at habitat scale

Forests provide timber material through well-established markets, but the associated habitat values of forests are also gained through unmarketed recreational activities, forest carbon sequestration, maintenance of biodiversity, microclimate, protection against natural hazards and water quality. Decisions on the use of forest resources should be based on a comparison between the expected monetary value of the harvested timber and the costs associated with the ecosystem goods and services that are lost as a result of timber loggings (Kallio et al. 2008).

However, ecosystem goods and services that do not have monetary value are generally not accounted for in the decision making process. Here we describe work which is an attempt to develop quantitative measures of biodiversity and ecosystem goods and services, in order to achieve sustainable use of forest resources. Traditionally, commercial forests are managed to maximize timber output. A methodology for integrating economic efficiency and biodiversity value is the base for mapping forest related biodiversity services (Kallio et al. 2008; Juutinen et al. 2008).

The trade-off between biodiversity and timber harvest value can be derived by the production frontier method, like e.g. in Pukkala et al. (1997), who developed biodiversity indices and employed them at the forest level for harvest planning. Calkin et al. (2002) explored trade-offs between the likelihood of persistence of a wildlife species and timber production by applying a model integrating spatial wildlife population, timber harvest and growth models. Nalle et al. (2004) evaluated land-use decisions and looked for cost-effective land-use alternatives. They combined a wildlife population simulation model with the economic model. The aim of the model was to calculate the present value of the sum of consumers' and producers' surpluses from timber harvest. Polasky et al. (2005) analyzed the consequences of alternative land-use patterns on the persistence of species and the economic returns. This type of models account for habitat preferences, habitat area requirements, and dispersal ability for different species to predict the probability of species persistence, as well as the land unit characteristics and location to predict the value of commodity production in a given land-use pattern to search efficient land-use patterns (Luque and Vainikainen 2008; Luque et al 2004; Romero-Calcerrada and Luque 2006; Figure 7.59). Spatial oriented habitat quality models are the primary source to evaluate habitat quality and relate it with its value.

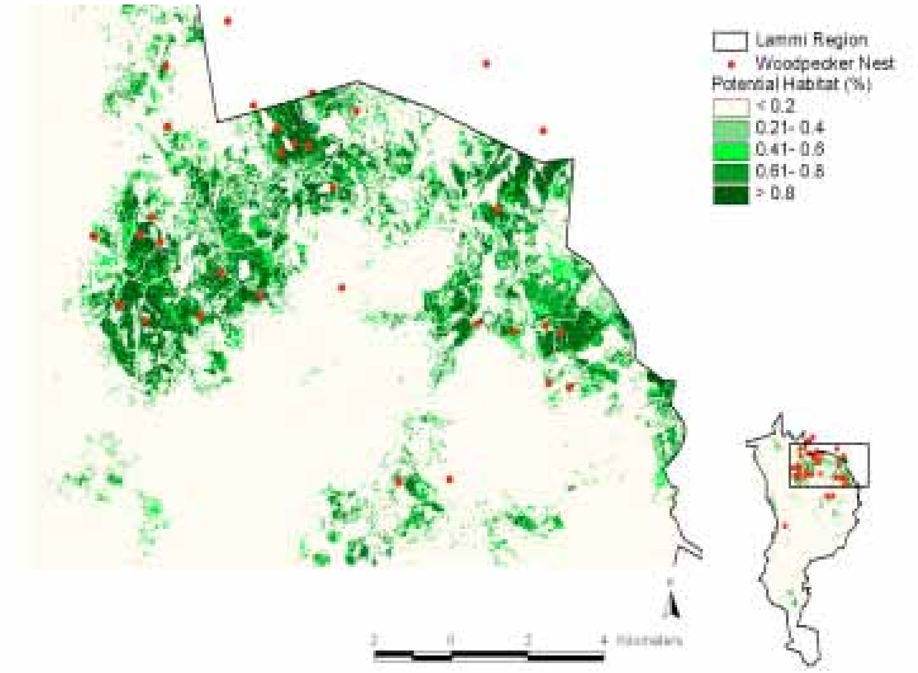


Figure 7.59. Three-toed woodpecker nesting habitat modelled for the years 1989 and 2000 using a spatial association approach among variables and nesting sites for each year (Romero-Calcerrada and Luque 2006).

New efforts to develop habitat quality models that can serve to calculate trade-offs using the production frontier method to map biodiversity services in the Alps, are under way, but still much work is needed to pursue the required methodological steps to reach the maps of related services. An approach for the spatial potential distribution of Pygmy-Owl is show bellow (Figure 7.60). In the same way the habitat of *Bonasa bonasia* in the Chartreuse mountain range within the Alps also is shown in Figure 7.61.

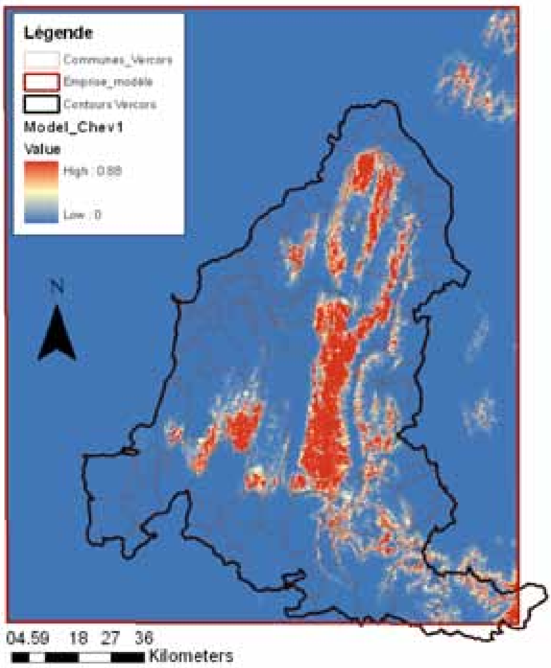


Figure 7.60. Potential presence distribution Glaussidium passerinum Eurasian pygmy-owl within the vercors range, France

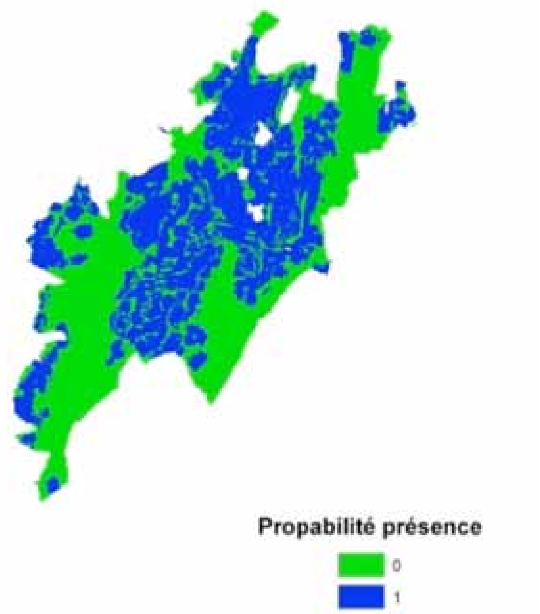


Figure 7.61. Potential presence distribution of Bonasa bonasia in the Chartreuse moutnin range within the alps, France

7.3.5. Conclusions and recommendations

The overall impression of the case study results is that in the forestry sector data are plentiful but ecosystem services has not yet been much of an organizing principle in collecting or presenting the data. Maps of forest features are available at different geographical scales but few of them can readily be used to develop operational ecosystem services maps.

Furthermore, data are widely collected for and about timber production purposes, both on the ecological and on the economic side, but seldom systematically for other uses of forests. There are case studies on recreational use and increasingly on climate issues (carbon sequestration and storage). Nevertheless synthesis and appropriate models are needed in order to produce derived information that can be coupled to ecosystem services algorithms that needs to be developed to be translated into usable outputs maps The ability of forest models to realistically simulate ecosystem services depends both on the ability of the model to accurately reproduce forest state and on the ability to translate the output of the forest models into suitable ecosystem service metrics (Schroter et al. 2005). For some ecosystem services, such as timber production, carbon sequestration, water quality regulation, and tree diversity, the output of forest and landscape models often correspond exactly to the relevant units for assessing the ecosystem service. However, for other ecosystem services, such as the moderation of extreme events (e.g. avalanche protection, flood protection) and recreational and aesthetic value, the quantitative output of the forest models regarding forest state must be transformed into a metric that appropriately reflects the ecosystem service that can be mapped.

This study illustrates that it is possible to read from the EU level maps “pattern variety” across Europe, e.g. of forest types, dominant species and age classes. The patterns are more informative when “point” information (obtained at site level in actual forest areas) is mapped than when regional or national averages are mapped. The latter approach however can be informative in a comparative sense across Europe (see e.g. the age classes maps). It remains to be seen which mapping approach present the better basis for evaluating changes in the forests over time. Obviously, local level degradation of forest and associated services will be averaged out and become invisible in the regional aggregates maps.

The national level case shows that in the past few years the attention for ecosystem services in the Netherlands has steadily increased, but so far very few official mapping attempts have been done. There are a few explorative studies, for instance of for ecosystem services of soils (Faber et al. 2009), but not specifically for Forest ecosystems. Data availability is not so much the problem, as forest statistics are readily available, both for the State Forests and for the private forests. The Dutch government has recently decided to conduct a TEEB analysis of the Dutch ecosystems (both natural, semi-natural, agricultural and urban) for which ecosystem service maps shall be developed.

At the regional level, the case in France illustrates that i) SES mapping supposes to combine 3 types of information (maps) : (Geo)Ecosytem characteristics, defining a potential service (offer), Sociosystem characteristics, characterizing the demand (users), and constraints map, playing a role of links/filters, structuring the relation between offer and demand (access, economical balance, socio-ecological limitations). ii) National aggregations have to be done from local studies at a fine scale. This bottom up approach is more data consumer but will provide a more accurate analysis. Modelling in tandem with high and medium resolution remote sensing data will allow scaling up the spatial dimension of the information to produce regional level maps. Close collaboration with National botanical institutions, NGO’s, naturalist’s organizations and NFIs, will be needed within the process. iii) The existing local data of NFIs, oriented towards statistical results at large scale, are often insufficient for an appropriate cartography even blurred at 1km resolution. A lot of progress may be done, for all SES assessment, by the improvement of these local data on ecosystems, through remote-sensing methods boosted by the LIDAR development.

At the habitat level, despite increasing attention to the human dimension of conservation projects, a rigorous, systematic methodology for planning for ecosystem services has not been developed. This is in part because flows of ecosystem services remain poorly characterized at local-to-regional scales, and their protection has not generally been made a priority. Spatially explicit conservation planning framework can be developed on the base of trade-offs and opportunities for aligning conservation goals for biodiversity with ecosystem services as an example for Europe.

In future monitoring programs and conservation practice it will be important to compared the degree to which contrasting conservation network designs protect biodiversity and the flow of services. Targeting ecosystem services directly can meet the multiple ecosystem services and biodiversity goals more efficiently but cannot substitute for targeted biodiversity protection (Chan et al. 2006).

In all, for future research it will be important to consider potential trade-offs between conservation for biodiversity and for ecosystem services, a systematic planning framework offers scope for identifying valuable synergies.

ANNEX 1. Overview of EU level maps and data sources, and relevance to forest ecosystem services

description	resolution	source	method	Timber	Carbon sequestration	Biodiversity	Recreation	Erosion
available datasets for mapping ecosystem services from forests in Europe								
standing volume	Nuts0-2	Schelhaas et al. 2003	official statistics different methods per country	X	X			
increment	Nuts0-2	Schelhaas et al. 2003	official statistics different methods per country	X	X			
forest area	Nuts0-2	Schelhaas et al. 2003	official statistics different methods per country	x				
forest structure	Nuts0-2	Schelhaas et al. 2003	official statistics different methods per country	X	X	X	x	
recreational value of forests	Nuts0-2	Schelaas et al. in prep	Interpretation of ageclass distribution of forests				X	
future forest volume & structure	Nuts0-2	EFISCEN	projection model	X	X	x	x	
forest area split to tree type	1km	Schuck et al. 2002	remotesensing & regional stastics	x				
forest volume split to tree type	500 m	Gallaun et al 2010	remotesensing & NFI plotdata & regional stastics	X	X			
tree species map (18 species)	1km	Brus et al in prep.	interpolation of NFI plotdata & ICP plotdata & regional statistics	x		x	x	
tree species map (6 species)	1km	Troeltzsch et al 2009	interpolation of ICP data & regional statistics	x		x	x	
future forest area	1km	CLUE, Verburg	landuse projection model	x				
Under development								
Forest management	1km	ALTERRA	development of new methodology		x		x	
Forest volume by 18 species	1km	ALTERRA	integrating existing sources	X	X	x		
Future forest volume & structure	1km	ALTERRA	EFISCEN-SPACE	X	X	X	X	
Potentials								
Erosion prevention	1km		combining elevation & slope maps with infrastructure & soil properties					X
Risk of windthrow & fire	Nuts0-2	Schelhaas et al. 2003	combining forest structure with fire weather index & wind forecasts	x	x			

8. Methods and indicators for mapping ecosystem services

8.1 Introduction

This chapter concentrates on the methodological aspects of mapping ecosystem services. It deals with selection of indicators, mapping approaches and data, use of models to create maps and information management techniques.

A first part concentrates on indicators for ecosystem services and evaluates methodological aspects of mapping these indicators. Working from lists and tables of indicators proposed in TEEB reports (TEEB 2010a; 2009) and by Layke (2009) (based on MA indicators), types of indicators are reviewed and evaluated (structure, quantity) as well as sources of data (Remote Sensing (RS), Photos vs measurements in databases). An essential problem is that ecosystem services are essentially flows of matter, energy and information, while most spatially explicit measurements in ecosystems are on stocks, structure and pattern. Special attention is given to usability of biodiversity data for inferring ecosystem services, as biodiversity is measures widely across Europe.

Scaling issues are considered in a second part. Indicators and their dimensions are not necessarily consistent across spatial scales, but need to be translatable.

A third objective is to formulate indicators in such a way that the (economic) Valuation step can be taken (in the near future). In addition, when thinking about the "actual" services (flows from ecosystems to Man and Economy, what are the challenges to map (spatial) dimensions of Ecosystem Services Use (be it resource extraction, recreational use, carbon sequestration). This would involve something like overlay maps of sources-located (= the ecosystems) and users-located (people in cities and rural landscapes).

Finally, a brief exploration considers the ideas to develop a digital Atlas of Ecosystem Services, and the organisational and infrastructural aspects of such an Atlas.

8.2 Indicators

8.2.1. Functions of indicators

The functions of indicators in the process of mapping ecosystems services are considered to be, in principle, no different than those in, for example, environmental studies or biodiversity studies. Much has been published about this. Here, a brief summary of the major features and steps is given, based on a.o. TEEB D0, chapter 3.

- **Communication:** indicators are selected which represent elements of (eco)systems which reflect the condition and trends of the systems which are considered relevant to society (for no specific purpose)

- **Early warning:** The objective is to communicate to society the state of their environment and early detect changes which may affect the well-being of humans and society.
- **Impact assessment:** indicators are selected specifically to show the consequences of action or inaction for human well-being by measuring the efficiency of measures we take
- **Target achievement:** indicators are derived from policy targets, in this case the halting of the degradation of ecosystem services, benchmarking and monitoring performance in relation to (SMART-ly) defined target levels.

It is considered to be useful in communication about the concept of ecosystem services to distinguish between measures, indicators and indices (TEEB, 2010a).
The term measure is used to refer to the actual measurement of a state, quantity or process derived from observations or monitoring. For example, bird counts are a measure derived from an observation.
The term indicator is both used

- in a generic sense, meaning a feature of a system which indicates some aspect of that system (stock, flow, structure, diversity, distance to target), and
- a specific sense, meaning a well defined feature, with a well defined function, with associated measures for quantification and qualification, and ideally with a specific set of methods to obtain the relevant data to qualify and quantify.

An index is comprised of a number of measures combined in a particular way to increase their sensitivity, reliability or ease of communication. These are useful in the context of biodiversity assessment where multiple attributes and measurements, related to a wide variety of policies, have resulted in long lists of measures and indicators. To communicate these trends in a small number of simple and meaningful indices is sensible (Balmford et al. 2005).
Ecosystem services are considered as a "conceptual bridge" between ecological and economic systems (see diagram in Figure 8.1; From TEEB 2010a). Ecosystem service indicators therefore ideally reflect both aspects of the "ecosystem" which produces the service, and of the economic system for which it is a benefit. Therefore ecosystem services indicators and measures used must be convertible into economic terms and suitable for economic analyses. An additional requirement, still widely under discussion, is that ecosystem services indicators should address the sustainability of the use that society makes of the ecosystems. This would imply some baseline in the presentation of indicator values (see discussion later in this section).

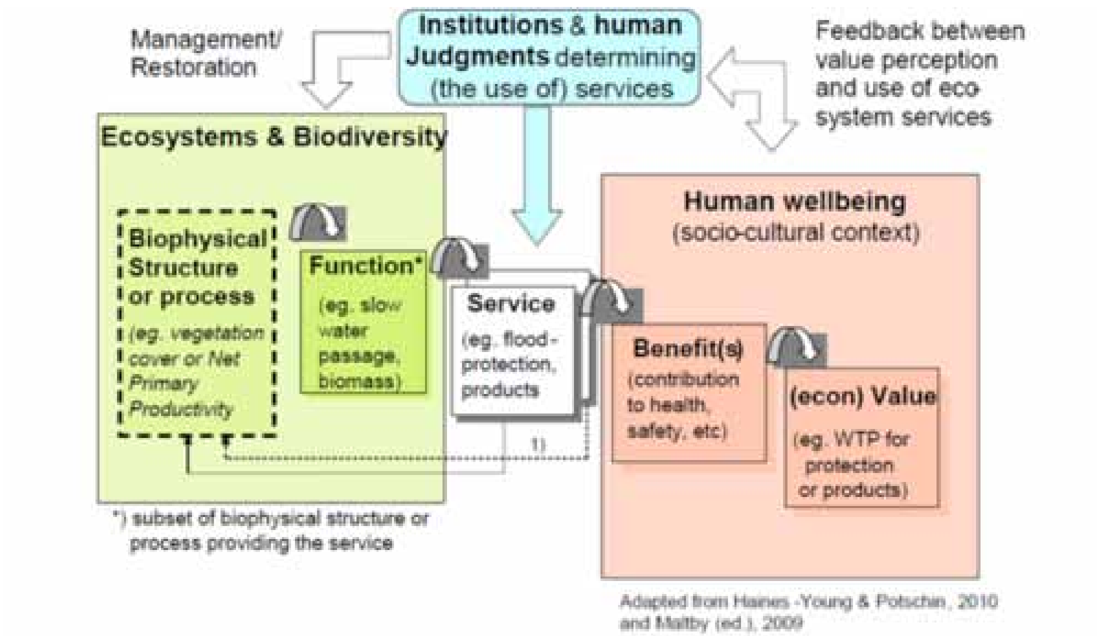


Figure 8.1. Ecosystem services as bridge between ecosystems (supply) and economic systems (demand) (De Groot et al. 2010).

It is often necessary, given the different functions of indicators, to identify a set of indicators which together reflect both the state and dynamics of the ecosystems, the flow of services, the benefits and associated values of the services to society and their spatial and time distributions. The structure of the SEBI2010 set of 26 biodiversity indicators is an illustration of this for biodiversity (Figure 8.2).

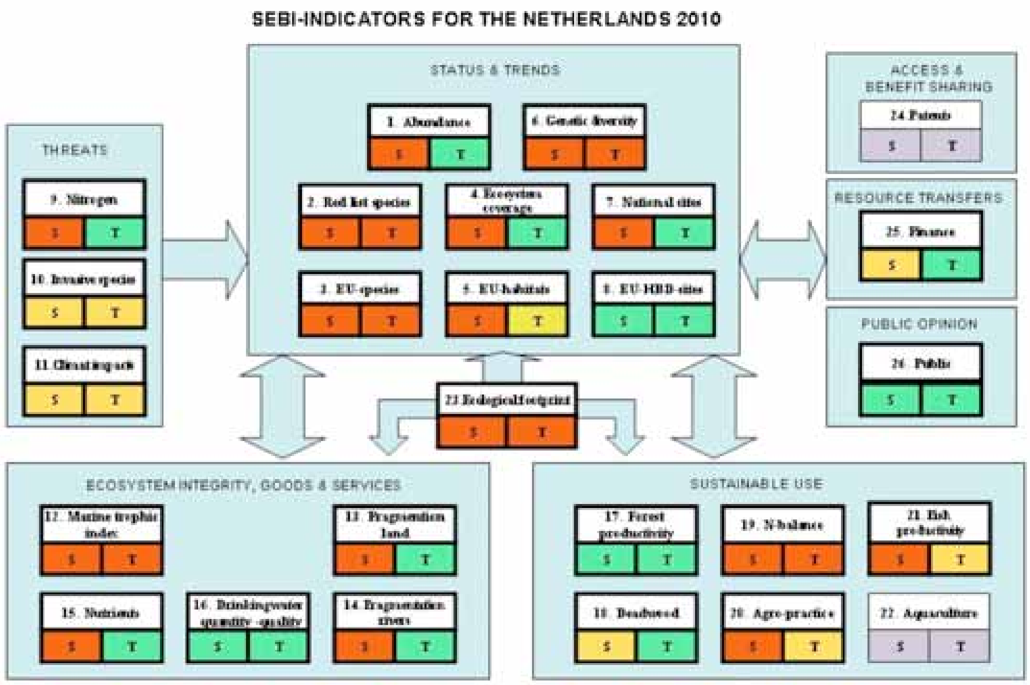


Figure 8.2. Biodiversity en ecosystem service inidcators of SEBI 2010; the case of The Netherlands; s= state; t= trend; red = below target; orange = risky ; green = on target; grey = no data yet.

Finally, as the nature of both ecological and economic systems is non-linear, with "surprising" changes in rates of production and use, a complete set of ecosystem service indicators needs to include features such as 'tipping points' or 'critical thresholds' (Secretariat of the Convention on Biological Diversity, 2010).

8.2.2. Ecosystem services indicators

The publication of the Millennium Ecosystem Assessment (MA) in 2005 has brought the concept of ecosystem services, which had been around in the science community since the late 1970s, finally to the political arena. In the COM(2006) 216 the European Commission used the phrase biodiversity and ecosystem services in many places in the text where before only biodiversity was mentioned. This also led to increased attention for indicators for ecosystem services, although the process of SEBI 2010 (Streamlining European Biodiversity Indicators 2010) had only just started to "streamline" the great number and variety of biodiversity related indicators.
As the concept of ecosystem services was introduced in many different sectors of European (and other) societies, the first ideas about the concepts crystalised often around traditional indicators in the production sectors for provisioning services, in environmental fields like climate, nutrients, water purification for regulating services and in the recreational and tourist sector as most familiar example of cultural services.

Purpose of ESS indicators. In line with the general purposes and functions of indicators (see above) the purposes of ESS indicators are thus:

- communicating the state and dynamics of ecosystem services, potential and actual, as a consequence of natural and human induced phenomena to public and policymakers (general communication; early warning)

- identifying the impacts of biodiversity and ecosystem loss and degradation on livelihoods and the economy (social and economic impact indicators)
- information for policy makers (from several points in the policy life cycle) for integrated decision-making that responds to environmental, social and economic needs (policy indicators).

Depending on the formulations used in the Post 2010 EU Biodiversity Strategy (in prep.) , the purposes of ESS indicators can be specified better. At this moment the adopted EU policy target for ecosystem services is phrased as "stop degradation" (European Council Decision, March 2010).

General properties and potential of ESS indicators. Ecosystem service indicators should make it possible to describe the flow of benefits provided by ecosystems and the associated biodiversity. The combination of measurements of biophysical capacities with measurements of benefit flows and economic values of ecosystem services can provide an effective tool that takes the whole value of our natural capital to human society into account. Even before the start of the TEEB project, and in fact during the Millennium Ecosystem Assessment project, it was clearly expressed that more attention was required to development of indicators of ecosystem services. Two lines of exploration were followed, one starting in the ecological domain with ecosystem features and functional biodiversity notions, and one starting in the economic domain, reasoning from the recognized benefits of society from ecological systems. Compared to 'traditional' biodiversity indicators on status and trends in species diversity and richness, ecosystem services indicators are a relatively new tool.

On the ecological side the "challenges" are:

- the complexity of functional relationships between ecosystem components
- how these affect the flow of services, and
- the multi-dimensional character of these services.

On the social, economic and institutional side the "challenges" are:

- recognition of commonly experienced benefits as being derived from and depended on the extent and quality of natural ecosystems (including biodiversity aspects)
- methods of valuation of the services in the human domain of social, economic and institutional rules and regulations
- lack of legal and institutional frameworks

In TEEB 2010a, Chapter 3, it is concluded that "we urgently need to better understand what is happening to biodiversity in order to conserve and manage ecosystem services effectively. All ecosystem services are underpinned by biodiversity and there is good evidence that biodiversity losses can have substantial impacts on such services". This PRESS Work Package 1 report is not the place to review the large body of literature on biodiversity indicators for their adequacy and effectiveness in representing the dynamics of all the aspects of biodiversity. Again, the literature is available to inform the reader, e.g. TEEB 2010a, Chapter 3; EASAC, 2005. The ambition in the PRESS project is to identify indicators which reflect ecosystem services adequately (scientific criterion) and effectively (policy criterion), and can be put on maps. Therefore, a brief review of the biodiversity indicators is included, which is taken from TEEB (2010a, Ch3). They indicators have been checked for usability as indicator for ecosystem services, building on what is already in the Table and providing initial comments about map-ability (= possibilities and problems in mapping the indicator).

In reviewing the literature, the following requirements for ecosystem services indicators emerged:

- because Ecosystem Services are in fact (1) flows of biomass, energy and information from ecosystems to humans and/or (3) represent actual work (energy transformations) in ecosystems, affecting environmental conditions for humans, and
- because flows are hard to observe and measure, but can be inferred from observations and measurements of changes over time in stocks, structure and spatial patterns, these are the types of ecosystem service indicators which seem most likely to be useful for mapping.

These observable c.q. measurable features can be mapped directly from aerial photos and Remote Sensing data, or indirectly from databases with ground data put in Geographical Information Systems. I therefore propose to use the term **DIRECT MAPPING** for the process of developing maps from remote sensing (RS) images and aerial photography, and to use the term **INDIRECT MAPPING** for the process of transforming (field) data into maps (Figure 8.3)

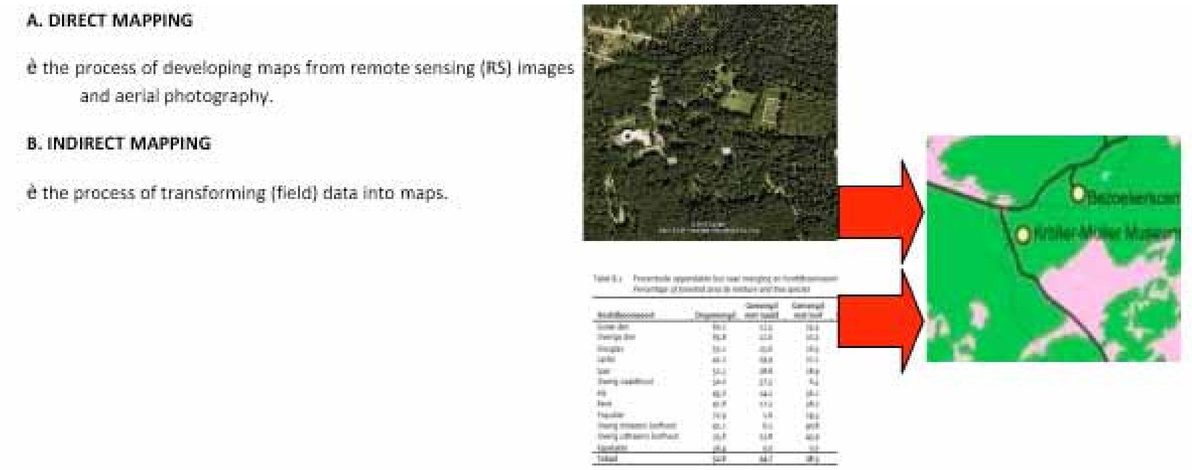


Figure 8.3. Direct and indirect mapping approaches

8.2.3. Indicators and maps

Maps are useful tools – not only for communication information about the world to the public but also to identify problems and solutions to policy makers. They provide a powerful instrument to communicate information spatially and can thus form the basis for targeting policy measures. For example, information in maps can identify who creates benefits from ecosystems and should therefore be eligible to receive a Payment for Ecosystem Services and who benefits from these ecosystem services and should therefore contribute to payments, if only to secure the future provision of such services

There are already several measures of ecosystem services in existence (see e.g. Layke 2009 for review). These measures are used to model and map the production of particular ecosystem services based on abiotic, biotic and anthropogenic factors, as well as knowledge of relationships between these factors. There is ample evidence, highlighted in Balmford et al. (2008) and in recent studies (e.g. Troy and Wilson 2006; Wendland et al. 2009), that the spatial mapping of ecosystem services at regional and global scales is a rapidly growing research area.

- Projects (The Heinz Center 2008; ATEAM) have made good progress in the development of indicators and the mapping of ecosystem services, even to the point of including scenarios of future change (Metzger et al. 2006).
- Wendland et al. (2009) examined trade-offs in the use of payments for ecosystem services (PES) mechanisms to finance biodiversity conservation in Madagascar.
- Troy and Wilson (2006) identified five core steps in the decision framework for mapping ecosystem services and conducting value transfer studies.
- Naidoo et al. (2008) observed that there is limited evidence of the spatial estimation of ecosystem services and the flow of benefits to near and distant human populations beyond a few local case studies.

Most of the existing quantitative analyses still tend to provide aggregated values for large regions, and data availability and disaggregation of spatial data are still a limitation to the mapping of ecosystem services. Tables 4.1 to 4.4 provides a survey and evaluation is made of the potential and map-ability of biodiversity indicators (from TEEB D), CH 3.

Table 8.1. Biodiversity indicators and their relevance for mapping ecosystem services (diversity)

Measures of diversity Species diversity, richness and endemism Beta-diversity (turnover of species) Phylogenetic diversity Genetic diversity Functional diversity Potential as indicators of ecosystem services: Not easily linked to specific provisioning or regulating ecosystem services, with the exception of proposed measures of functional diversity. Some support for congruence between diversity and service levels. Some importance of species and genetic diversity in promoting ecosystem resilience across ecosystem services. Genetic diversity linked to options for bio-prospecting and food security. Cultural values of diversity, especially education, research and aesthetic values, are a link to cultural ecosystem services. Map-ability: Species richness and pattern variety can be directly mapped (i.e. from RS, aerial photos), at least at the level of large plants (trees, scrubs) and vegetation units (habitats). The other indicators can be represented on maps (indirectly mapped) if spatially explicit data are available.

Table 8.2. Biodiversity indicators and their relevance for mapping ecosystem services (quantity)

2. Measures of quantity Extent and geographic distribution of species and ecosystems Abundance / population size Biomass Net Primary Production (NPP) Potential as indicators of ecosystem services: Clear links to provisioning services. Measures of stocks (1,2,3,) and flows (4) of ecosystem services. Useful for ecosystems and species with social and cultural values => cultural services. Some use in measuring regulating services, which rely on biomass or a particular habitat / vegetation cover (e.g. carbon sequestration, erosion control, water flow regulation). Map-ability: Extent and geographic distribution of species and ecosystems; this can typically be mapped directly for large species (trees, scrubs, some animal populations e.g. elephants), and ecosystems (habitats, vegetation units); much data is available to do indirect maps Abundance / population size: Only possible to directly map this indicator in large species; indirectly mapping has high potential, as many plant and animal populations have been studied in geographical context. Biomass and Net Primary Production (NPP): Not possible to map directly, but proxies can be registered, e.g. by ground cover %. They are measured and modeled widely, so indirect maps can be developed for many ecosystems.

Table 8.3. Biodiversity indicators and their relevance for mapping ecosystem services (condition)

Measures of condition Threatened species (Red List Index (RLI) Threatened habitats / ecosystems (degradation) Ecosystem connectivity / fragmentation Trophic integrity Changes in disturbance regimes Population integrity / abundance measures cf reference values Potential as indicators of ecosystem services: these indicators are not often linked to quantified changes in ecosystem service levels. however useful indicators of sustainability, (if thresholds are known) policy relevant as point of action to halt degradation may be indicated Map-ability: Threatened species (Red List Index (RLI) ➛ well possible indirectly Threatened habitats / ecosystems (degradation) ➛ well possible indirectly; loss of area can be mapped directly Ecosystem connectivity / fragmentation ; well possible directly and indirectly Trophic integrity : only indirectly Changes in disturbance regimes : mostly indirectly (nitrogen pollution), but also directly (fires, floods) Population integrity / abundance measures cf reference values (only indirectly).

Table 8.4. Biodiversity indicators and their relevance for mapping ecosystem services (pressure)

Measures of pressures Land cover change Climate change Pollution and eutrophication (Nutrient level assessment) Human footprint indicators (e.g. HANPP, Living Planet Index – LPI, ecological debt) Levels of use (harvesting, abstraction) Alien invasive species Potential as indicators of ecosystem services: When linked to particular species (e.g. fish) or ecosystems (e.g. wetlands) which provide or support ecosystem services, then these measures are useful (predictive/correlative) indicators of ecosystem service levels and declines. Useful to indicate the sustainability of ecosystem service use and supply. Map ability: Land cover change: direct + indirect Climate change: indirect (based on temperature, rainfall etc.); direct by changes in vegetation occurrence Pollution and eutrophication: indirect Human footprint indicators : indirect Levels of use : direct e.g. logged forest; indirect Alien invasive species: indirect

Table 3.4 in TEEB 2009 offers a useful first set of ecosystem services indicators, based on the MA framework, that are already in use or are being developed. Some of the few existing and commonly agreed indicators on regulating services have been drawn up from the environment sector (e.g. climate change and carbon sequestration/storage rates, natural flood protection). So far, there are more indicators for provisioning services than for regulating and cultural services, due to our clear and immediate dependency for basic needs fulfilment on provisioning services which are mostly incorporated into marketed commodities (e.g. wood for timber, fuel and food). The flow of benefits from regulating and cultural services is not as visible or easily measurable: many non-market services are therefore enjoyed for free. Proxy indicators can help us estimate benefits associated with these services by referring to the

capacity of an ecosystem to provide them – but these are only a short-term solution. More widespread use of ecosystem services in political decisions will require us to improve regulating and cultural service indicators (Layke 2009) and incorporate them into EIA and SEA type of legislation. Promising ideas such as the trait concept (Layke 2009), which seeks the clear definition of characteristics required for the provision of services, are available but need further elaboration. Figure 4 shows an expanded version of Table 3.4 of TEEB 2009, with a 3rd column added in which the Map-ability is indicated as based on the type of indicator.

Table 8.5. Map-ability of provisioning ecosystem service indicators (after TEEB, 2009).

PROVISIONING SERVICES		MAP-ABILITY
Food Sustainably produced/ harvested crops, fruit, wild berries, fungi, nuts, livestock, semi-domestic animals, game, fish and other aquatic resources etc.	• **Crop production** from sustainable [organic] sources in tonnes and/or hectares • **Livestock** from sustainable [organic] sources in tonnes and/or hectares • **Fish production** from sustainable [organic] sources in tonnes live weight (e.g., proportion of fish stocks caught within safe biological limits) • **Wild animal/plant production** from sustainable sources in tones	Crops directly; all indirectly; Data for crops and livestock at FAO; Fish data not as reliable; other data scarce;
Water quantity	• **Total freshwater resources** in million m^3	Direct (surface water) and indirect (surface and groundwater) ; Data widely available.
Raw materials Sustainably produced/ harvested wool, skins, leather, plant fibre (cotton, straw etc.), timber, cork etc; sustainably produced/ harvested firewood, biomass etc.	• **Forest growing stock**, increment and fallings • **Industrial roundwood** in million m^3 from natural and/or sustainable managed forests • **Pulp and paper production** in million tonnes from natural and/or sustainable managed forests • **Cotton production** from sustainable [organic] resources in tonnes and/or hectares • **Forest biomass for bioenergy** in million tonnes of oil equivalent (Mtoe) from different resources (e.g. wood, residues) from natural and/or sustainable managed forests	Forests direct; forests and others indirect; Data widely available
Genetic resources Protection of local and endemic breeds and varieties, maintenance of game species gene pool etc.	• **Number of crop varieties for production** • **Livestock breed variety** • **Number of fish varieties for production**	No direct mapping; Data for varieties available in many EU countries for crops and livestock; fish data not reliable; aquaculture ??
Medicinal resources Sustainably produced/ harvested medical natural products (flowers, roots, leaves, seeds, sap, animal products etc.); ingredients / components of biochemical or pharmaceutical products	• **Number of species from which natural medicines have been derived** • **Number of drugs using natural compounds**	No direct mapping; only indirect Some statistics available;
Ornamental resources Sustainably produced/ harvested ornamental wild plants, wood for handcraft, seashells etc.	• **Number of species used for handcraft work** • **Amount of ornamental plant species** used for gardening from sustainable sources	No direct mapping . Data availability ?

Table 8.6. Map-ability of regulating ecosystem service indicators (after TEEB, 2009)

REGULATING SERVICES		
Air purification Regulation of air quality through exchange of air pollutants with vegetation	• **Atmospheric cleansing capacity** in tonnes of pollutants removed per hectare	No direct mapping; Data from cases; models
Climate/climate change regulation Carbon sequestration, maintaining and controlling temperature and precipitation	• **Total amount of carbon sequestered** / stored = sequestration / storage capacity per hectare x total area (Gt CO2)	No direct mapping; Data from models and case studies

Moderation of extreme events Avalanche control, storm damage control, fire regulation (i.e. preventing fires and regulating fire intensity)	• **Trends in number of damaging natural disasters** • Probability of incident	Direct mapping only fires (long lasting) and floods; + traces (avalanches) Statistics available for many types (floods, fires; avalanches)
Regulation of water flows Regulating surface water runoff, aquifer recharge etc.	• **Infiltration capacity/rate** (e.g. amount of water/ surface area) - volume through unit area/per time • **Soil water storage capacity** in mm/m • **Floodplain water storage** capacity in mm/m	Maps based on models and soil maps. Storage capacity from geomorpho-logical maps
Waste treatment and water purification Decomposition/capture of nutrients and contaminants, prevention of eutrophication of water bodies etc.	• **Removal of nutrients** by wetlands (tonnes or percentage) • **Water quality in aquatic ecosystems** (sediment, turbidity, phosphorous, nutrients etc)	Maps based on models and field data.
Erosion control / prevention Maintenance of nutrients and soil cover and preventing negative effects of erosion (e.g. impoverishing of soil, increased sedimentation of water bodies)	• **Soil erosion rate** by land use type	Maps based on models and field data.
Pollination Maintenance of natural pollinators and seed dispersal agents (e.g. birds and mammals)	• **Abundance and species richness** of wild pollinators • **Range of wild pollinators** (in km, regular/ aggregated/ random, per species)	Maps based on cases / field work
Biological control Seed dispersal, maintenance of natural enemies of plant and animal pests, regulating the populations of plant and animal disease vectors	• **Abundance and species richness of biological control agents** (e.g. predators, insects etc) • **Range of biological control agents** (e.g. in km, regular/ aggregated/random, per species) • **Changes in disease burden** as a result of changing ecosystems	Maps based on cases / fields data and models

Table 8.7. Map-ability of cultural ecosystem service indicators (after TEEB, 2009) (continued)

CULTURAL SERVICES		
Aesthetic information Amenities provided by the ecosystem or its components	• **Abundance and score of objects; landscape types**	Maps based on landscape features (direct and indirect maps) + survey scores (photo-based)
Recreation and ecotourism Hiking, camping, nature walks, jogging, skiing, canoeing, rafting, diving, recreational fishing, animal watching etc.	• **Abundance / area of recreation sites**	Direct maps, indirect on survey + land use maps
Cultural values and inspirational services, e.g. education, art and research	• **Abundance and score of objects / areas; landscape types**	Maps based on classes of objects; land use; archaeological, natural monuments etc

8.2.4. Relevant indicators at local scales

Chan et al. (2006), Nelson et al. (2009) and Reyers et al. (2009) use data from a variety of sources on ecosystems and biodiversity (especially functional types), land cover, population, access, hydrology and economic value to model and map multiple ecosystem services at a local scale in the USA and South Africa. These maps were used to investigate trade-offs and planning options by Chan et al. (2006), to quantify the consequences of land use change on ecosystem services by Reyers et al. (2009) and to investigate the consequences of future scenarios on ecosystem services by Nelson et al. (2009).

Many of these indicators have been used for investigating the consequences of ecosystem and biodiversity change on ecosystem services. While some of the indicators are expressed in biophysical quantities, these quantities (litres of water, tons of carbon) are convertible into economic terms. This conversion is clearly demonstrated in another local scale study by Naidoo and Ricketts (2006) in Paraguay where the value of ecosystem services was modelled and made spatially explicit to assess the costs of benefits of biodiversity conservation in the region.

8.2.5. Ecosystem services and the sustainability criterion

There are some authors that think that ESS indicators need to take account of the sustainability of provisioning and other services over time, to ensure that the long-term benefit flow of services is measured. High economic (monetary) values may arise from overexploitation of ecosystems (harvesting the stock instead of the annual production!), which than may lead to erroneous conclusions about land use and beneficial investments.

These phenomena may occur as well with provisioning services (e.g. overexploitation of fish stocks) as well as cultural services (e.g. degradation of nature areas due to high tourist densities) and regulating services (e.g. palm oil plantations instead of natural tropical forests). Indicators referring to those services therefore need to take sustainable production rates into account. This calls for a clear definition of what sustainability actually means with regard to those services.

The service delivery capacity depends on ecological measures such as ecosystem robustness, integrity and resilience, not on the time and place dependent economic asset value. Economic benefits from ecosystem services exploitation must be compared to the additional costs required to maintain ecosystem capital in the broadest sense (i.e. to mitigate overall degradation), rather than to the narrower measurement of the losses of benefits resulting from natural resource depletion. It is crucial to develop *a baseline* in order to determine where critical thresholds (e.g. population of fish stock within safe biological limits, soil critical loads) and alternative future pathways under different policy scenarios (e.g. fisheries subsidies reform, subsidies in the agriculture sector) may lie. However, setting critical thresholds raises substantial problems linked to ignorance, uncertainties and risk associated with ecological systems. The precautionary principle and safe minimum standards may offer ways to overcome these challenges.

Not all ecosystem service indicators can easily be quantified. This leads to a risk that decisions are based on those for which quantifiable information is available. As stated in TEEB D0 Chapter 3, “reliance on existing indicators will in all likelihood capture the value of a few species and ecosystems relevant to food and fibre production, and will miss out the role of biodiversity and ecosystems in supporting the full range of ecosystem services, as well as their resilience into the future.” To avoid risks of creating a policy bias by focusing on a subset of indicators high on the political agenda or the agenda of vested interests, complementary (not-yet-quantified) indicators must be developed. In parallel, ESS valuations that focus on a single service should be systematically cross-checked to assess the capacity of ecosystems to continue delivering the full variety of other services potentially of interest.

Using generic maps of levels of service production, and changes in these levels, as a proxy of value and value change, may miss out on 2 crucial facets related to ecosystem management thresholds: sustainability and vulnerability. This reflects the challenge highlighted in the MA that change in ecosystems and their services are seldom linear or independent and can often be accelerating, abrupt and potentially irreversible (MA 2005b). The loss of biodiversity and increasing pressures from drivers of ecosystem change increase the likelihood of these non-linear changes. While 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.

8.2.6. Applications of ecosystem service indicators

In environmental and resource policy. The development of ecosystem services indicators will inevitably have to be accompanied by a clear definition of relevant policy goals to ensure the effectiveness of such indicators as an integration tool. A widely recognised set of indicators on the quality of ecosystems and their capacity to provide ecosystem services will be necessary to effectively measure progress towards those targets and the efficiency of approaches taken.

A streamlined set of headline indicators would be sufficient for high level target setting and communication by policy makers, politicians, the press and business, but must be supported by wider sets for measurement and monitoring. A small set of headline indicators may be enough for communication and high-level target setting but there is also social and economic value in having detailed ecosystem service indicators for certain policy instruments. Initiatives such as Streamlining European 2010 Biodiversity Indicators (SEBI 2010) and the CBD global headline indicators have started taking into account a limited number of indicators relating to ecosystem capacity to provide services and goods (e.g. water quality of freshwater ecosystems) and to sustainable use of provisioning services (e.g. ecological footprint; area of forest, agricultural and aquaculture ecosystems under sustainable management).

In business. Ecosystem services indicators can also be included in corporate reporting standards to communicate the impacts of lost services on company performance and the impacts of companies on provision of these services (e.g. Global Reporting Initiative). These include e.g. policy assessments, Environmental Impact Assessments (EIA) and national accounting as well as procedures to analyse companies' economic dependency and impacts on ecosystem services through materiality or Life Cycle Assessments (LCA). In policy and environmental impact assessments, such indicators help us to answer questions on the economic, social and environmental consequences of different policy or planning options affecting biodiversity. With regard to national accounting, indicators can be integrated into Systems of National Accounts (SNA) through the development of satellite accounts.

8.3 Indicators

Ecosystem services, as identified and described by the Millennium Ecosystem Assessment may be defined and measured at different spatial scales. The biological processes underlying the services to a large extent determine whether the essential service providing unit is primarily local – regional (e.g. pollination, due to the physical limitations of the pollinators), others are in essence without physical boundaries (climate regulation, as defined by sequestration of free CO2). This means that mapping ecosystem services requires a clear definition of the spatial scale of the measurements (primary data), to be able to trace the data manipulations such as aggregation and disaggregation. In the selection of ecosystem service indicators these aspects of data and data handling need to be made explicit. Some of the indicators may easily be "upscaled" (or vice versa down-scaled), because their physical dimensions are expressed in the weight (volume) / area / time units (e.g. kg/ha/year for biomass production). This would suggest looking for indicators with such dimensions, but we realise that for some ecosystem services that may not be so easy or appropriate.

8.3.1. Hierarchical structure of maps

Building on the physical dimension of the core indicator for a service, associated dimensions and features may be linked, and represented in the maps at the different spatial scales. This would lead to a "Hierarchy of indicator features". For example: Forest timber production (m3/ha/yr) (Figure 8.4).

1. EU level map of forest timber production (m^3/ha/yr ; e.g. a few classes = colour)
2. MS level map of timber production per forest type combined with feature of different typologies e.g. single use, multiple uses.
3. Regional level map of timber production , again expanded with information about species, owner/ management agency, which can be represented on the map
4. Site level map of timber production per cohort per species (idem)

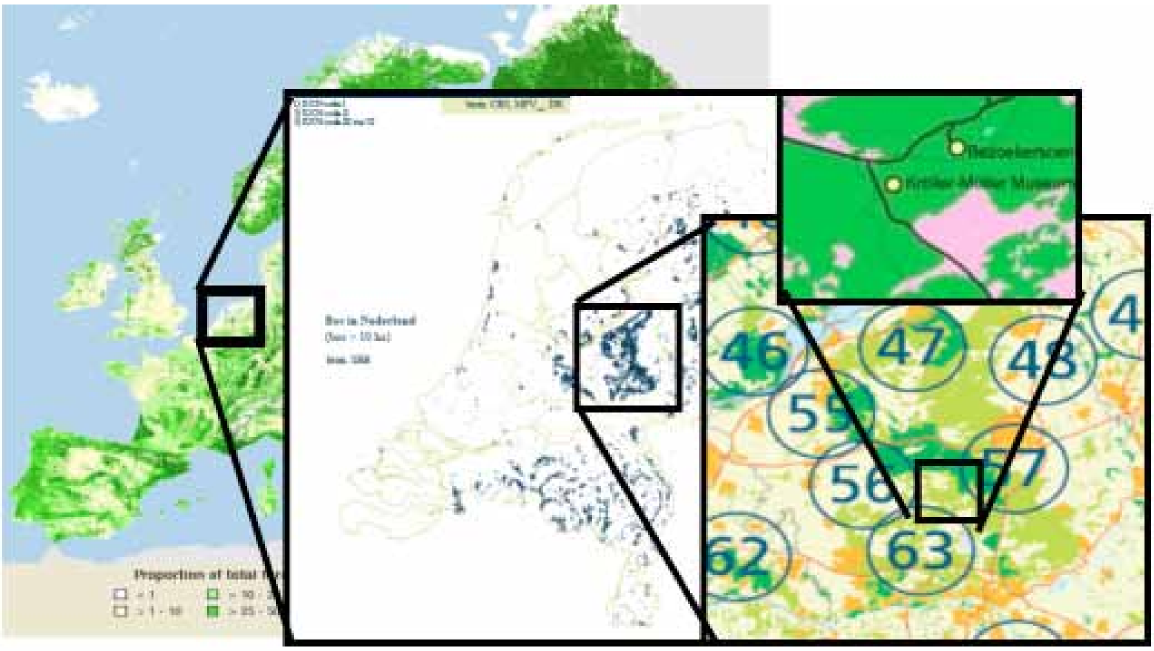


Figure 8.4. Hierarchy of maps (the maps are not reflecting the examples in the text)

If based on statistics, these data should be additive, upwards (and fall out in consistent sets downwards). If the map is based on RS or photo-data, identity (= colour = type) at each zoom-level should offer access to statistics (type = production data).

8.3.2. Multiple services maps

Ideally, a hierarchical set of maps becomes available for each specific/ defined ecosystem service, at predefined spatial scales (see illustration in Figure 8.5).

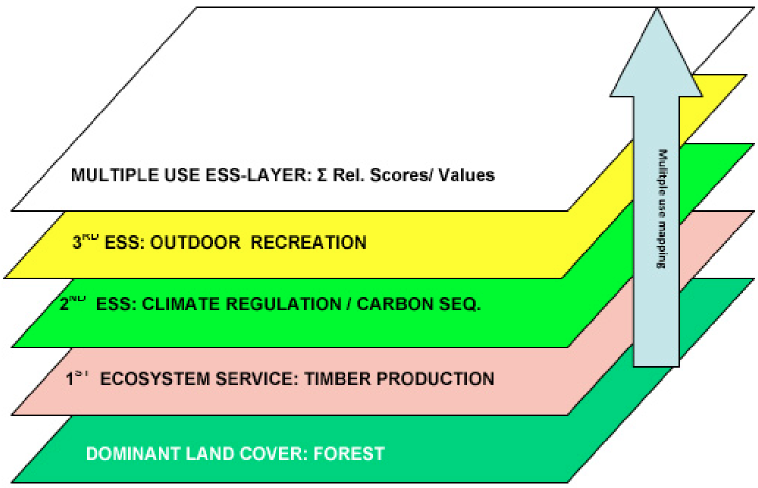


Figure 8.5. Multiple use ecosystem services mapping

The working assumption is that in due time, monetary values will be assigned, according to agreed methodologies to all selected ecosystem services. Assuming this to be realised, maps can be developed which have multiple use information, for example:

- Qualitative numerical addition: colour coded ecosystem services at each "cell" on the map (cells may be sites, regions; higher aggregation probably not useful)
- Qualitative specific combinations: colour coded combinations of ecosystem services at each cell of the map.
- Quantitative additions and combination, where the quantities of services in physical units are coded in a way that can be added and mapped.

- Multiple Use Values: total economic value per cell, either by added single use values, after establishing non-competitiveness, or by market based total land (sea) cell values.

8.3.3. Maps of synergies and conflicts

With the different sets of current (2000-2010 baseline) maps, potential threats and conflict due to (urban, agricultural, conservation) development plans as well as options for policy and management synergies can be identified and tested (Figure 8.6). Spatially explicit dynamic models, such as CLUE (Verburg et al. 2010), may help to explore these conflicts and options. The 2020 situation can be simulated under different scenarios and management strategies designed to achieve EU ESS objectives.

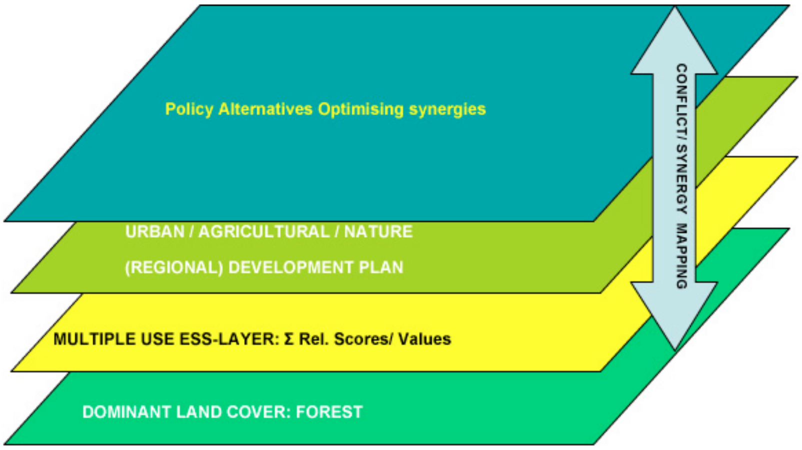


Figure 8.6. Mapping potential conflicts and synergies

8.4 From ecosystem services to ecosystem values

There are many different ways of defining well-being and the linkages between well-being and ecosystems. In this section, only concepts used in economics are considered. While this is by no means a fully comprehensive approach it provides a theoretically consistent underpinning for assessing values. Furthermore, it provides an operational approach to evaluate the consequences of changes in ecosystems and resulting changes in the flows of services, and is therefore amenable to evaluation of alternative policy directions.

Economic assessment of values builds on an extensive literature developed over the past 30 years on environmental valuation, attempting to identify and quantify values of goods and services when market transactions and associated prices do not exist to infer values. This highlights the importance of distinguishing between “value” and “price”. Often ecosystem goods and services are priced at zero, even though their value in terms of the impact on human well-being is non-negligible. Outdoor recreation is often used to illustrate this point. While the access to recreational opportunities is often free, and the price therefore zero, people choose to spend their time and expenditure on travel in order to benefit from the service. This fact is a simple manifestation of the discrepancy between price and value, and the environmental valuation research has developed from the need to demonstrate and quantify the values from ecosystems in order to account for such value in environmental policy development and evaluation. Initially this literature focused on the development of robust approaches to value individual services such as for example recreational opportunities from individual recreational sites. The research focus is now to an increasing extent concerned with the more complex task of valuing multiple and interacting sets of ecosystem services from entire landscapes.

In the section an overview of some of the challenges emerging from this change in focus is presented. Firstly, an overview of the types of values arising from the human interactions with ecosystems is given, as well as a brief overview of approaches to value ecosystem services. Secondly, we list additional methodological issues when attempting to map ecosystem services values. This includes a discussion of the extent to which ecosystem valuation takes into account spatial context and the challenges involved in landscape scale evaluation of the consequences of alternative policy direction. Finally, we conclude the section with a discussion on the future challenges involved in spatial ecosystem service valuation.

8.4.1. Typology of values

It has become customary to distinguish between use and non-use values as a first differentiation between the different sorts of economic values arising from ecosystems. Use value is a measure of the relative satisfaction, happiness, pleasure or satisfaction of preferences derived from, or desirability of, the consumption of a good or service. Table 8.8 provides definitions for the different types of use values. Non-use values (also known as passive values) are values that are not associated with actual use of a good or service (Brouwer et al. 1999) (Table 8.9).

Table 8.8. Different aspects of use value.

Direct use value (consumptive)	The value derived from the actual consumptive use of a good or service. For example, the harvesting of timber is a direct use consumptive value of a forest.
Direct use value (non-consumptive)	The value derived from the actual non consumptive use of a good or service, e.g. recreation is a direct non consumptive use value of a forest.
Indirect use value	The indirect (non consumptive) value that is derived from ecosystems, such as the role of ecosystems in maintaining clean water supplies.
Insurance value or quasi option value	The value derived from the reduction of risk to which an individual or society is exposed. E.g. Biodiversity may add to resilience of the provision of ecosystem services, which would be an insurance value of biodiversity.
Option value	The potential future value that may be derived by future individuals (descendant and future generations).

Table 8.9. Different aspects of non-use value.

Bequest values	The value derived from to the satisfaction gained from preserving a natural environment for future generations.
Existence value	The value conferred by the existence of an organism, or organisms, independent of their utility to humans. Is the value that people place on simply knowing that something exists, even if they will never see it or use it.
Intrinsic value	This is based on the object being valued for itself rather than because it serves a valued purpose. (Note that this definition implies that intrinsic value cannot be valued using an economic framework)

In the ecosystem service valuation literature the focus has so far mainly been on the first three types of use values; consumptive and non-consumptive use value and indirect use value and existence value. While it is often argued that insurance value is an essential ecosystem service, the challenges involved in estimating such values have so far proved too complex for most case studies. Table presents the types of values and methodologies applicable to the majority of ecosystem services valuations.

8.4.2. Mapping ecosystem values

8.4.2.1 Spatially explicit valuation

There are many ways in which space enters in valuation research. To give an overview of some of the key issues relevant in the context of ecosystem value mapping we describe two fundamentally different ways in which spatial context and spatial relations are important for ecosystem service valuation.

Direct relation between spatial variables and values. Spatially defined characteristics such as size, distance and proximity are often key determinants of the value of ecosystem service flow. As an example this has been demonstrated in a spatial valuation study of recreational activities (Termansen, 2008). In this study it is shown that large forest sites are more attractive to recreationists as they were preferred over smaller sites in a revealed preference study. The study also shows that easy access is a key determinant of recreational demand, however the study also showed that access is not sufficient for determination of recreational choice of site. Site characteristics giving information on the quality of the recreational opportunities are also important for the prediction of the flow of recreational activities. For example, existence of open semi-natural habitats in proximity to the forest sites, topography, nature trails, composition of tree species, share of old forest and access to the coast were shown as important attributes of recreational demand.

Table 8.10. Valuation methods applied to ecosystem services.

VALUATION METHOD	USE TYPE	APPLICATIONS	ES VALUED
Adjusted market prices: Market prices adjusted for distortions such as taxes, subsidies and non-competitive practices.	Direct use value (consumptive)	Provisioning services	Crops, livestock, woodland, etc.
Production function approach: Estimation of production functions to isolate the effect of ecosystem services as inputs to the production process.	Indirect use values	Regulating services	Maintenance of beneficial species, productive ecosystems and biodiversity; storm protection; flood mitigation; air quality, peace and quiet, workplace risk.
Damage cost avoided: Calculates the costs which are avoided by not allowing ecosystem services to degrade.	Indirect use values	Regulating services	Drainage and natural irrigation; storm protection; flood mitigation
Averting behaviour: Examination of expenditures to avoid damage	Indirect use values	Regulating services	Pollution control and detoxification
Revealed preference methods: Examine the expenditure made on goods related to ecosystem (e.g. travel costs for recreation; hedonic (typically property) prices in low noise areas).	Direct use value	Provision of space often classified as cultural services.	Maintenance of beneficial species, productive ecosystems and biodiversity; storm protection; flood mitigation; air quality, peace and quiet, workplace risk.
Stated preference methods: Uses surveys to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay for those goods	Use and non-use value	Applications to most types of ecosystem services	Water quality, species conservation, flood prevention, air quality, peace and quiet.

Source Bateman et al. 2011.

Indirect relation between spatial variables and values. The distribution of human populations is clearly spatially clustered. As the amount of people affected by a policy development will be of key importance for the costs and benefits associated with a policy implementation it is clearly important to have information on people distributions. Furthermore, socio-demographic characteristics are also often clustered in space. As variations in socio-demographics are often associated with preferences and attitudes towards the environment, this spatial component is likely to be significant for valuation mapping values and priorities between different service deliveries. Finally, the different ways in which ecosystem goods and services impact human well-being are also likely to be an important factor in value mapping. As use value rely on direct use of the environment, valuations are therefore likely to be closely related to the proximity of people to the resource. Non-use values are likely to have less of a spatial signature, as benefits from the ecosystem service do not rely on direct use.

8.4.2.2 Aggregation of values across space:

In economic policy evaluation it is essential to be able to value alternative policy directions, i.e. the change in ecosystem service provision. This means that if a policy proposal is predicted to result in a change in land use we need to be able to measure the change in ecosystem services flows at different locations to groups in society with potentially different preferences for different services. As argued above, the value of an ecosystem service flow is often dependent on the spatial context and the relative scarcity of the service. As alternative policy directions are likely to change both the spatial context and the magnitude of different ecosystem flows, the values of ecosystem services are also likely to change. This implies that valuation will not be a simple "add-on" step of associating fixed values to new quantities. Both the ecosystem service quantities delivered and the values of the change in delivery need to be assessed. Examples may help to illustrate this point. The value of a development of new forest sites for recreational opportunities will depend both on the characteristics of the sites, access from urban clusters and the availability of alternative sites. A policy proposal to develop new sites is therefore likely to reveal higher recreational values in close proximity to residential areas (all other characteristics being equal) and where few alternatives sites exist. The new sites will however also decrease the value of existing substitute sites and a landscape scale evaluation of changes in recreational service provision and recreational service demand is needed for economic policy evaluation.

8.4.3. Future challenges in ecosystem service value mapping

Improving the understanding of the spatial context of valuation and the ability to map implications of alternative policy delivery requires sufficient valuation data to determine and correct for spatial context. Many case studies do exist and for some resources sufficient research has been done to undertake meta-analyses allowing comparisons across studies (Zandersen and Tol, 2009). This has led to development of benefit transfer methods, developing best practice in the use of previous valuation studies to allow policy evaluation with cost effective use of additional survey effort. Progress has been made in the development of such methods to test spatial transfers of values. However, there is still insufficient systematically collected information to validate such approaches as the individual studies most often have attempted to improve on the methodology of the previous studies or address different aspects of the ecosystem service of interest.

8.5 Biodiversity of ecosystem services

Recent biodiversity policies introduce the concept of ecosystem services as a means of mainstreaming biodiversity into other policies, notably agriculture, fisheries and forestry. The argument is that these policies are dependent on biodiversity resources and are therefore partly responsible for some of the declines that are observed in biodiversity. The assumption is that the provision of ecosystem services is underpinned by and hence, correlated to biodiversity. As a consequence, maintaining ecosystem services is assumed to contribute to conservation of habitats and species.

Although it is evident the biodiversity underpins ecosystem services, the exact mechanisms remains poorly understood. Studies based on experiments, maps overlaying indicators for biodiversity with indicators for ecosystem services, field observations or meta-analysis of published data often report weak correlations between biodiversity and ecosystem services. The dominance of few species in ecological communities which are consuming and transferring the bulk of the energy and material flows in ecosystems may result in weak correlations between ecosystem services and biodiversity, often taking the form of an asymptotic relation whereby increasing biodiversity does not result in increasing ecosystem functioning once a plateau is reached. As a result, ecosystem service and nature conservation priorities may not always overlap.

In November 2010, Alter-Net organized a workshop which aimed to review the state of the art of present knowledge on the link between ecosystem services and biodiversity. Key to this debate is how to define biodiversity. A narrow definition puts biodiversity equal to species richness or relative species abundance. A broader definition of biodiversity including also structural and functional traits of species as well as landscape and ecosystem diversity may therefore result in much better relations between biodiversity and ESS.

A second argument is that ecosystem service indicators are often based on models which do not include biodiversity as a variable in the model. An example is nitrogen retention that is mapped in this study as an indicator for water quality regulation by rivers and streams. None of the equations used accounted for biodiversity but only physical variables related to climate and geomorphology of the landscape were considered. These variables are used in some way as proxies to the functioning of aquatic ecosystems rather than including in the models the processes performed by organisms. Aquatic biodiversity, in particular river bed bacteria, macrophytes and plankton, are indeed the main consumers of in stream nitrogen and recent evidence shows that river retention is positively influenced by aquatic biodiversity. But it remains challenging to include all these functions in models, in particular in terms of calibration.

The TEEB Ecological and economical foundation study reports extensively on the relation between biodiversity and ecosystem services describing for each service its sensitivity to variations in biodiversity.

8.6 Towards an atlas of ecosystem services

The development of a digital Atlas of Ecosystem Services has been proposed in the project, which have led to some ideas of structuring the maps of the digital Atlas across the European landscape of institutes and government agencies.

As European maps may be developed (a) from EU data sources (statistics, RS info) and (b) from Member State data sources (statistics, RS info, completed ESS maps) an organizational framework must be developed which encompasses the institutional aspects such as ownership of data, public versus private financing, access and payment for use, management and update of quality etc.

The data infrastructure may look as follows. Each country or institution maintains its own data stored or made available on a local server. The institutional servers should expose some of their data to a standard format (view) to the central server by means of web services. These data are queried on regular basis by a central validation and aggregation service to perform data check and produce a harmonized European dataset at different spatial resolution e.g. 1 deg, 5 deg, 1 km, 10 km, 100 km. This service also takes care of updates and ensures a sustainable storage of the data, which can be indexed for a fast access. XML RESTfull or optionally SOAP (if necessary) services should be used to transport the data between the services. Users in the outside world can then perform queries of aggregated data for a particular output, e.g. generation of maps and reports accompanied by INSPIRE compliant metadata.

8.7 Conclusions and the way forward

In order to make a comprehensive and compelling economic case for the conservation of ecosystems and biodiversity it is essential that we are able to understand, quantify and map the benefits we receive from ecosystems and biodiversity, and assign values to those benefits. This all must be done in a fashion that makes it possible to assess the contribution made by ecosystems (including the abiotic processes) and their biodiversity features to this value, as well as the consequences of changes in ecosystems and biodiversity for these values.

In view of the scientific evidence about loss of biodiversity and some ecosystem services (MA, 2005; EEA, 2010) and in view of the ambitions regarding the future state of biodiversity and ecosystems in Europe (EU, 2010) there is an obvious need:

- To be spatially explicit about location, extent, quality and threats to ecosystem services at a European scale while not producing large regional aggregations; this call for nested data sets (local => regional => national=> EU)
- For investment in spatially explicit data, local and regional scales are a first necessary step in improving ecosystem service mapping and in turn economic valuation
- To improve the alignment between available maps of ecosystem services and existing models or scenarios of future change.

The flow of ecosystem services from point of production to point of use is influenced by both biophysical (e.g. currents, migration) and anthropogenic (e.g. trade, access) processes which influence the scale of service flow from locally produced and used services (e.g. soil production) to globally distributed benefits (e.g. carbon sequestration for climate regulation). To move from mapping ecosystem service capacity (potential services) to mapping ecosystem service flows (actual services), we need to:

- Map how the flow of benefits and scale of flows influences the value of the service due to changes in demand and supply which vary spatially and temporally.
- Analyse currently available socioeconomic data for their usability for such multilevel value assessments
- Include information about the distribution of users, the socio-economic circumstances of users, governance systems, human pressure on ecosystems and other social measures like willingness and perceptions.

For some ecosystem services and some audiences, economic valuation is seen as essential. When considering potential trade-offs between provisioning services (usually captured by market prices) and regulating services (often non-marketed services), the absence of monetary values for regulating services can create a bias towards provisioning services. It is therefore considered important to develop a broadly supported set of valuation techniques for the various ecosystem services, in line with their ecological and their economic features. For the sake of biodiversity and for ecosystem resilience, it is important to keep a balance of services, instead of focussing on optimising just one (preserving ecosystem service provision is not the same as conserving ecosystems, let alone their biodiversity). Elements of ecosystems and biodiversity for which no role in service provision has been identified, and which are thus considered to create no economic value, are not "worthless", but beyond the (limited) scope of economic valuation. Their inherent value demands conservation beyond the economic calculus. Economic considerations and measurements can be helpful to support biodiversity and ecosystem service conservation, but they can become devastative if they are used as the main criterion. Basing conservation decisions on cost benefit analysis (CBA) means deciding based upon category errors (Skourtos et al 2010).

Until such an agreed set is available, clearly each type of information is important. Although qualitative indicators do not quantify and monetize benefits arising from ecosystem services, they are an important tool to underpin quantitative and monetary information and help to close gaps where no such information exists.

9. Glossary of abbreviations

AFOLU	Agriculture, Forestry and Land Use
ATEAM	Aquatic and Terrestrial Ecosystems Assessment and Monitoring
BOD	Biochemical Oxygen Demand
BOD5	5-Day Biochemical Oxygen Demand
CAP	Common Agricultural Policy
CAPRI	Common Agricultural Policy Regionalised Impact
CBA	Cost Benefit Analysis
CBD	Convention on Biological Diversity
CDDA	Common Database on Designated Areas
CEH	Centre for Ecology & Hydrology
CICES	Towards a Common International Classification of Ecosystem Services
CLC	Corine Land Cover
CLUE	Conversion of Land Use and its Effects
CO	Carbon Monoxide
CO2	Carbon Dioxide
COPI	Cost of Policy Inaction
CORINE	Coordination of Information on the Environment
DEM	Digial Earth Model
DG	Directorate General
DSM	Digital Surface Model
EASAC	European Academies Science Advisory Council
EEA	European Environment Agency
EIA	Environmental Impact Assessments
EPIC	Erosion-Productivity Impact Calculator
ESS	Ecosystem Services
EU	European Union
EVALUWET	European Valuation and Assessment Tools: Supporting Wetland Ecosystem Legislation
FAO	Food and Agriculture Organization of the United Nations
FFH	Fauna-Flora-Habitat
FMAs	Forest Management Approaches
FSS	Farm Structure Survey
GAK	Joint Task for the Improvement of Agricultural Structures and Coastal Protection (D)
GIS	Global Information System
GLEAMS	Groundwater Loading Effects of Agricultural Management Systems

GREAT-ER	Geography-Referenced Regional Exposure Assessment Tool for European Rivers
GREEN	Geospatial Regression Equation for European Nutrient losses
GWAVA	Global Water AVailability Assessment model
GWAVA-WQ	Global Water AVailability Assessment model - Water Quality
ha	hectare
HANPP	Human Appropriation of Net Primary Production
HGMUs	Homogeneous Spatial Mapping Units
HOST	Hydrology of Soil Types
HRU	Hydrological Response Unit
ILE	Integrierte Ländliche Entwicklung (Integrated Rural Development Concept)
INSPIRE	Infrastructure for Spatial Information in Europe
JRC	Joint Research Centre
LCA	Life Cycle Analysis
LCM	Land Cover Map
LEADER	EU's Rural Development Programme
LIDAR	Laser Imaging Detection and Ranging
LPI	Living Planet Index
MA	Millennium Ecosystem Assessment
MAB	Man and the Biosphere Programme
METSO	Forest Biodiversity Programme for Southern Finland
Mio	Million
MONERIS	Modelling Nutrient Emissions in RIver Systems
MORECS	The Met Office Rainfall and Evaporation Calculation System
N2O	Nitrous Oxide
NCYCLE	Nitrogen Cycle Model
N_EXRET	Model for simulating total N export and retention in large scales
NFI	National Forest Inventories
NGO	Non-Governmental Organisation
NPP	Net Primary Production
NUTS	Nomenclature of Territorial Units for Statistics
PCRD	Programme Cadre de Recherche et Développement
PES	Payments for Ecosystem Services
PRESS	PEER Research on Ecosystem Services
PROPWET	Proportion of time when soil moisture deficits are less than 6mm
Q10	River nitrate retention at high flow
Q50	River nitrate retention at medium flow
Q95	River nitrate retention at low flow
RLI	Red List Index
ROS	Recreation Opportunity Spectrum
RPI	Recreation Potential Index
RS	Remote Sensing
SACs	Special Areas of Conservation

SCIs	Sites of Community Importance
SEA	Strategic Environmental Assessment
SEBI2010	Streamlining European Biodiversity Indicators 2010
SNA	Systems of National Accounts
SOAP	Simple Object Access Protocol
SPAs	Special Protection Areas
SQ	Status Quo
SYKE	Finnish Environment Institute
TEEB	The Economics of Ecosystems and Biodiversity
UNESCO	United Nations Educational, Scientific and Cultural Organization
WEDSS	Wetland Evaluation Decision Support System
WFD	Water Framework Directive
WP3	Work Programme 3
XML RESTfull	Web services using HTTP and the principles of REST - Respresentational state transfer

One ton refers to 1000kg

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Ecosystems are critically important to our well-being and prosperity as they provide us with food, clean air or fresh water and they maintain a livable biosphere. Consequently, ecosystem services are increasingly considered as crucial argument to support decision making in policies that affect the use or the state of natural resources. In particular, new biodiversity policies, which are now adopted at global and EU scales, have set targets to safeguard biodiversity as well as to maintain the supply of ecosystems services.

The inclusion of ecosystem services into biodiversity policies has increased the demand for demonstrating the value of natural capital in order to justify investments in biodiversity protection. Hence, in order to make a comprehensive and compelling economic case for the conservation of ecosystems and biodiversity it is essential that we are able to understand, quantify and map the benefits received from ecosystems and biodiversity, and assign values to those benefits.

The PRESS initiative addresses some of the knowledge gaps which stand in the way of performing such a spatially-explicit, biophysical, monetary and policy assessment of ecosystem services in Europe. This report presents the first results focused on a selection of cases at different spatial scales to test and further develop methodologies for mapping indicators and policy analysis.

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