International Journal of Biodiversity Science, Ecosystem Services & Management
i*First*, 2012, 1–15

Mapping water quality-related ecosystem services: concepts and applications for nitrogen retention and pesticide risk reduction

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One of the challenges of using the ecosystem service (ES) framework in the context of planning and decision support is the question of how to map these services in an appropriate way. For water quality-related ESs, this implies a movement from the display of classical water quality indicators towards the mapping of the service itself. We explore the potential of mapping such water quality-related ESs based on three case studies focusing on different aspects of these services: (1) a European case study on pesticides, (2) a multi-scale German case study on nitrogen retention and (3) a more local case study on nitrogen retention in the Elbe floodplain (Lödderitzer Forst). All these studies show a high spatial variation of the results that can be depicted in maps of ES supply. This allows an identification of areas in which nitrogen retention is highest or which areas face the highest ecological risk due to pesticides. The multi-scale case study shows how the level of detail of the results varies with model resolution – a hierarchical approach to environmental and river basin management seems useful, because it allows the planners to determine scale-specific environmental problems and implement specific measures for the different planning levels.

Keywords: ecosystem services; mapping; water quality; nitrogen; pesticides; water framework directive

Introduction

Ecosystem services (ESs) are the benefits that humans obtain from ecosystems (Millennium Ecosystem Assessment 2005). The Millennium Ecosystem Assessment and follow-up projects such as ‘The Economics of Ecosystems and Biodiversity’ (The Economics of Ecosystems & Biodiversity 2010) have raised the awareness of ESs in the scientific community as well as in stakeholder and decision-maker circles (Fisher et al. 2008; Seppelt et al. 2011). However, the ES concept still faces multiple challenges regarding research needs and its application in policy support (Carpenter et al. 2006; Daily et al. 2009; de Groot et al. 2010; Seppelt et al. 2011). One of the challenges of using the ES framework in the context of planning and decision support is the geographical mapping of services (de Groot et al. 2010). Such maps are assumed to be valuable tools for the interaction with stakeholders and decision-makers.

Water-related ESs are among the most frequently studied ESs. Seppelt et al. (2011) found that from a sample of 153 ES studies, 105 considered as water-related services (60 fresh water provisioning, 52 water quantity regulation and 40 water quality regulation, with several studies focussing on more than one water-related service). Most case studies that investigate water quality regulation services reported water quality as a service in itself. However, water quality itself is not an ES (Brauman et al. 2007), but an indicator of the result of intermediate services such as water purification and erosion control, or it is a driving factor for other services such as recreation or fish nursery habitat. From our point of view, ESs need to be mapped directly.

Water quality regulating services act by absorbing or filtering pollutants or by preventing erosion. The processes related to these services may take place during overland flow, during infiltration and leaching, during ground water passage or in wetlands or in water bodies. Ecosystem processes involved range from physical processes (such as vegetation preventing erosion) to biochemical processes by microorganisms in soil, water or wetlands. The benefit of the service consists *inter alia* of decreasing water treatment costs, increasing the aesthetic value of water for swimming and tourism and supporting fish stocks harvested for commercial or recreational purposes (Loomis 2000; Brauman et al. 2007). Relevant substances for water quality-related ESs include the macronutrients, nitrogen and phosphorus, as well as ‘down-the-drain’ chemicals, heavy metals and pesticides. All these substances have effects on in-stream ecosystems, water quality and/or availability and are at least partly or temporarily removed from soil or from the river

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ISSN 2151-3732 print/ISSN 2151-3740 online

http://dx.doi.org/10.1080/21513732.2011.631940
http://www.tandfonline.com

system. With respect to ecological water quality, supporting services like the provisioning of habitat for aquatic species need to be considered as well (Brauman et al. 2007).

ES maps based on representative sampling of the entire study region are limited to well-studied areas – for example, the map of recreation (Eigenbrod et al. 2009) or the maps on taxon richness (Holland et al. 2011). Alternatively, ES maps can be based on empirical relationships or on model results (de Groot et al. 2010). Empirical approaches usually fit a regression type model to the existing data on service supply and landscape characteristics. The regression model is then used to forecast service supply based on the landscape properties in the model (Alessa et al. 2008; Willemen et al. 2008, 2010). An alternative way of mapping ESs is to employ a calibrated simulation model to quantify the service (Swallow et al. 2009; Jenkins et al. 2010). The improvement of a simulation model over an empirical model is its increased flexibility for forecasts under changing conditions; the drawback is the risk of overparameterization, which may increase the uncertainty of forecasts. Following good practice rules such as those suggested by Jakeman et al. (2006) during model development may safeguard against such risks. If data for model calibration are missing, one approach is to transfer empirical relationships or fitted simulation models from other regions to the case study region or to use proxy variables likely to affect ES supply (Chan et al. 2006; Naidoo and Ricketts 2006; Troy and Wilson 2006; Önal and Yanprechaset 2007; Egoh et al. 2008; Tallis et al. 2008; Kienast et al. 2009; Lonsdorf et al. 2009; Doherty et al. 2010).

Process models have been used to estimate water regulation services, but from the 40 studies that consider water quality regulation in Seppelt et al. (2011) only 16 used any kind of model, while the majority (27 studies) used look-up table approaches. Look-up table approaches most often reported the economic values of ESs in summary tables (e.g. Loomis 2000; Wang et al. 2010), while maps were used less frequently. Examples for the use of maps in the context of water regulating services include Tong et al. (2007) or Bryan and Kandulu (2009), who overlaid several look-up table or benefit-transfer-based ESs to map cost-efficiency or suggested management actions, although without reporting the service maps themselves. An example for the application of process models is the study of Swallow et al. (2009), who used the integrated hydrological model Soil Water Assessment Tool (SWAT, Arnold and Fohrer 2005) to quantify and map agricultural production and sediment yield without actually mapping the ESs. Likewise, Yates et al. (2005) used the WEAP21 model to calculate classical water quality parameters without further investigating these services. Jenkins et al. (2010) used the Environmental Policy Integrated Climate (EPIC) model together with empirical models on denitrification to map the economic value of nitrogen mitigation together with greenhouse gas mitigation and waterfowl recreation.

Some properties of the hydrological system complicate the estimation of water-related services. Water-related services tend to be regional services in the sense that location of the service provisioning and the place of consumption of the service are different (Brauman et al. 2007). Downstream users might benefit from the services that ecosystems provide further upstream. However, upstream users might also impact downstream users by decreasing the potential of ecosystems to provide services. In addition, hydrologic ESs including water quality regulation vary temporally and single events such as floods, droughts or accidents with hazardous substances tend to have strong effects on these processes (Brauman et al. 2007). Lakes and wetlands might even switch from a nutrient-retaining state to a nutrient-emitting state (Gordon et al. 2008). As a consequence of temporal variability and single-event impacts, monitoring is not easy and periods covered by time series are often not long enough to contain sufficient information to appropriately parameterize statistical or simulation models for extreme conditions.

The study shows how water-related ESs can be used in addition to classical water quality indicators to support river basin managers. Three case studies are used to show how the concept of mapping ESs can be applied to water quality issues. The case studies used simulation models and expert models to quantify ES-related indicators. In addition, we bring attention to the question of the scale at which a service is mapped – case studies 2 and 3 analyse and map ESs at a series of hierarchical scales. Since management questions and scale of analyses are related, we investigate how a hierarchical approach could be used in the context of water regulation services and water quality.

Methods

Method overview

The three case studies for which we mapped water quality-related services differ in area size as well as on the ES studied (Table 1). The case studies focus on pesticides (case study 1) as well as on nitrogen. Nitrogen-related services are considered at different spatial scales (case studies 2 and 3, Figure 1). We defined the service for case studies 1 and 2 based on the comparison of model runs with and without ecosystem functioning. The difference between both model runs is defined as the service supply. Case study 3 directly estimates the ESs.

All the three case studies map ESs based on supply-side state indicators. The supply side of ESs focuses on the potential good or service produced by the environment (Figure 2). To generate a benefit for the society, the supply of the service has to be linked to a demand by society. For some services such as food production, the benefit derived also depends on other aspects such as fertilization, irrigation or mechanic labour, which complement natural ES supply.

Table 1. Case study overview.

Case study	Location	Area (km^2)	Ecosystem service indicator used	Spatial resolution	Type of model used
1	EU-27	4,324,782	Habitat provisioning/ nursery	Grid based, 10 × 10 km	GIS model based on empirically fitted equations
2	German Elbe catchment	95,220	Nitrogen retention in the catchment	Polygon based, 132 sub-basins	Grey-box model (Elbe-DSS, ordinary differential equations as well as nutrient balance model)
	Parthe basin	318	Nitrogen retention in the catchment	Polygon based, 53 hydrological response units	Grey-box model (SWAT, ordinary differential equations)
3	Lödderitzer Forst	10	Nitrogen retention in the floodplain	Polygon based, 49 hydrogeomorphologic units	Expert knowledge model

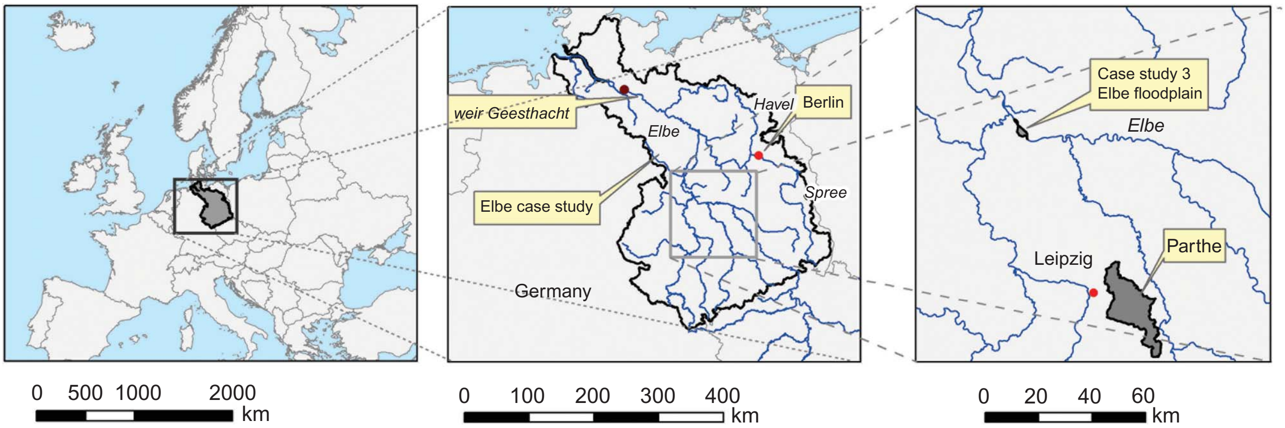

Figure 1. Location maps for case studies 2 (German Elbe basin, including Parthe basin) and 3 (Elbe floodplain 'Lödderitzer Forst').

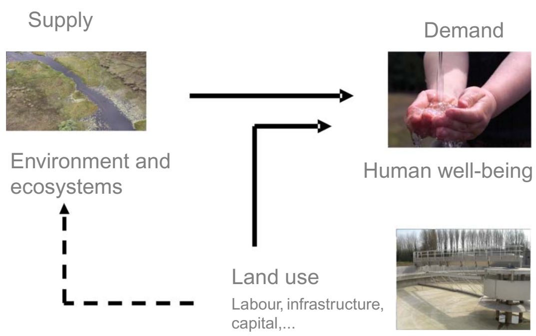

Figure 2. Overview about the demand and the supply side of ecosystem services. Photos: Sven Lautenbach (left), MADe (Wikipedia commons, right, bottom), André Künzelmann (top right).

Case study 1: mapping risk reduction of pesticides at the European scale

The first European case study focuses on pesticides as a substance class that is used in conventional food production. Pesticides have major impacts on terrestrial ecosystems (Geiger et al. 2010) as well as on aquatic ones. With respect to the aquatic ecosystems, pesticides might limit their potential to provide the services of water quality regulation, breakdown of leaf litter or outdoor recreation (Brauman et al. 2007; Schäfer et al. 2007; Rasmussen et al. 2008). Pesticides, in particular insecticides, have adverse short- and long-term effects on freshwater communities (Liess and von der Ohe 2005; Downing et al. 2008). They prevent the achievement and maintenance of a good chemical and ecological status of surface water bodies, as aimed for by the EU Water Framework Directive (WFD, European Commission 2000). The desired status is defined as the occurrence of only slight differences in the composition and abundance of species when compared with water bodies that are undisturbed by human activities.

We used a spatially explicit model to predict the potential exposure of small streams to insecticides (run-off potential – RP) as well as the resulting ecological risk (ER) for freshwater fauna on the European scale (Schriever and Liess 2007; Kattwinkel et al. 2011). The key environmental characteristics of the near-stream environment that influence RP are topography, precipitation, soil type and soil organic carbon content (Schriever and Liess 2007). Anthropogenic factors are crop type and levels of application of insecticides – given data limitation, the level of pesticide application is only available data for a country (Schriever and Liess 2007).

RP is correlated with patterns of macroinvertebrate communities of small agricultural streams (Schriever et al.

2007): the higher the value of RP, the lower the proportion of species sensitive to insecticide exposure within the community. The recovery of community structure after exposure to insecticides is facilitated by the presence of undisturbed upstream stretches that can act as sources for recolonization (Niemi et al. 1990; Hatakeyama and Yokoyama 1997). In the absence of such sources for recolonization, the structure of the aquatic community at sites that are exposed to insecticides differs significantly from that of reference sites (Liess and von der Ohe 2005). Consequently, such exposed sites do not meet the requirements of the WFD for good ecological status.

Hence, we calculated the ER depending on RP for insecticides and the amount of recolonization zones. ER gives the percentage of stream sites in each grid cell (10×10 km) in which the composition of the aquatic community deviated from that of good ecological status according to the WFD. In a second step, we estimated the service provided by the environment comparing the ER of a landscape lacking completely recolonization sources with that of the actual landscape configuration. Hence, the ES provided by non-arable areas (forests, pastures, natural grasslands, moors and heathlands) was calculated as the reduction of ER for sensitive species. The service can be thought of as a habitat provisioning/nursery service that leads to an improvement of ecological water quality.

Case study 2: mapping nitrogen retention in the German Elbe basin

The Elbe case study builds on results compiled during the Elbe-DSS project (Berlekamp et al. 2007; Lautenbach et al. 2009) as well as on results generated by Strauch et al. (2009). The case study maps the service provided by ecosystems in the catchment with respect to nitrogen retention. Nitrogen leaching from arable fields is a common problem in many intensively managed agricultural areas (Scanlon et al. 2007; Stoate et al. 2009). In addition, nitrogen reaches the river system via point emissions from households and industry. Simulations of nitrogen retention were conducted at four different scales (Figure 3): the German Elbe catchment (95,220 km^2), five first-order management/reporting units (average area: 19,044 km^2) defined by the WFD (European Commission 2000), 132 sub-basins (average area: 716 km^2) and the Parthe basin (318 km^2) as a subunit of one of the sub-basins. We calculated the effect of retention in the catchment using Equation (1) based on the amount of nitrogen applied to arable fields (APP), the nitrogen removed during harvest (HAR) and the amount of nitrogen that enters the river system via non-point emissions (NPE):

$$RET_{catchment} = APP - HAR - NPE. \qquad (1)$$

The results for the first three scales were generated using an integrated model that consists of MONERIS (Behrendt et al. 1999, 2003), GREAT-ER (Matthies et al. 2001; Heß et al. 2004; Koormann et al. 2006) and LFBilanz (Bach and Frede 1998, 2005). LFBilanz calculates nutrient surplus on arable farmlands based on a database on average fertilizer use, crop rotation systems and nitrogen removal during harvest. This surplus was then used in the deterministic nutrient balance model MONERIS. It calculates average perennial means of diffuse inputs (total phosphorus and total nitrogen) caused by erosion, surface run-off, groundwater flow, tile drainage, atmospheric deposition and impervious urban areas for 132 sub-basins. The data on farming practice and climatic conditions refer to the situation around the year 2000. All calculations were performed under the scenario assumption of constant nitrogen surplus.

In addition to the retention in the catchment, retention in the river network was estimated. The average concentrations and loads in the river network were calculated by the aquatic fate and exposure assessment model GREAT-ER. GREAT-ER incorporates input from about 1900 wastewater treatment plants as well as from the diffuse emissions that are calculated by MONERIS. It uses the inputs together with substance-specific degradation rates and discharge statistics for the calculation of concentration and load in the river network. The river network consists of approximately 33,000 river stretches with an average length of 1.5 km. The nitrogen retention rates per river stretch were estimated based on the empirical relationship between retention and hydraulic load described in Behrendt and Opitz (2000).

For the finest scale, the Parthe basin, the Soil Water Assessment Tool (SWAT) (Arnold and Fohrer 2005) was used. The Parthe basin is an agricultural basin in the neighbourhood of Leipzig, Eastern Germany (Figure 1). The Parthe catchment itself is not included directly in the 132 sub-basins of MONERIS; it covers 84% of the MONERIS sub-basin ‘Leipzig-Thekla’. The area was divided into 6 sub-basins and 53 hydrological response units. In contrast to MONERIS, SWAT is a process-based model that calculates time series of discharge and water quality indicators at the sub-basin outlets. Compared to MONERIS, a more detailed specification of management and tillage practice is possible in SWAT.

Case study 3: mapping nitrogen retention in floodplains at the ‘Lödderitzer Forst’

Case study 3 deals with flood retention at a site of the EU-EVALUWET project, the ‘Lödderitzer Forst’ (Figure 1). This alluvial forest is divided into a flooded (400 ha) and non-flooded (600 ha) areas by a dyke. The potential nitrogen retention was calculated for the current situation as well as for the situation after a planned dyke replacement, which would reconnect the remaining alluvial forest to the hydrological system of the River Elbe.

Floodplains play an important role in reducing nutrient loads in rivers via overbank flooding (Pinay et al. 2002). Flooding directly affects nutrient cycling in alluvial soils by controlling the duration of oxic and anoxic phases (Pinay et al. 2002). Denitrification is one of the most

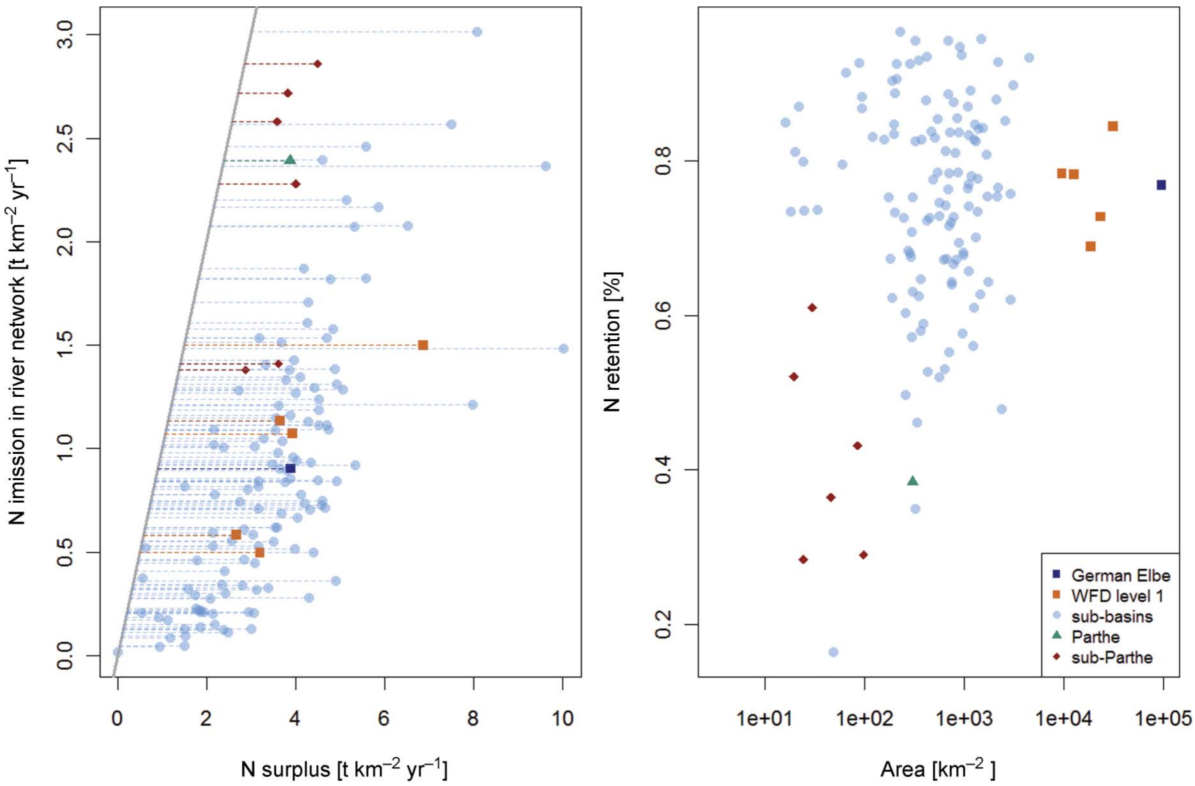

Figure 3. Nitrogen emissions and nitrogen retention services in the Elbe catchment.

important processes that permanently remove bioavailable nitrogen from ecosystems (Venohr et al. 2003; van der Lee et al. 2004; Verhoeven et al. 2006; Bondar et al. 2007; Mulholland et al. 2008). Denitrification rates are likely to be higher in floodplains than in the main channel due to lower oxygen levels (Olde Venterink et al. 2003). However, floodplain areas of many European rivers have been reduced in the past due to embankment (Tockner et al. 1999).

The amount of the nitrogen retention by floodplains was estimated by using the hydrogeomorphological unit (HGMU) approach (Brinson 1993, 1996; Maltby et al. 2006). A conservative estimate of 100 kg N ha^{-1}yr^{-1}, as taken from a literature review (Trepel 2009), was used as a base value, later modified by factors based on environmental conditions in HGMUs. The following factors were considered: flooding frequency, flooding regime, groundwater level and vegetation type. The procedure resulted in retention values from 50 to 250 kg N ha^{-1}yr^{-1}.

Results

Case study 1: mapping risk reduction of pesticides at the European scale

The ES estimation is based on a comparison of a model run with and one without recolonization sites. The changes in ER as a function of recolonization sites and RP describe the service. We predicted RP for insecticides to be very high in many parts of Italy as well as some regions of France, Spain and Greece, which could be traced back mainly to comparatively high rates of insecticide application, loamy soils and steep slopes (Figure 4a). Due to medium to high rates of insecticide application combined with moderate to high amounts of arable land and some steep slopes, large parts of the United Kingdom, southern Germany, France, Spain, Austria and Slovakia exhibited high RP for pesticides (categories high and very high, Figure 4). A low to medium RP for insecticides was predicted for Denmark, Finland, the Baltic States, Poland and northern Germany. It can be assumed that this was due to medium rates of insecticide application and to a lower degree due to moderate slopes and sandy soils that reduce surface run-off due to higher infiltration.

The distribution of predicted ER mostly matched that of RP for insecticides (Figure 4b). Likewise, the ES reduction in ER for sensitive species was also estimated to be above 50% in Italy, northern Spain, southern France, northern France/Belgium, followed by England, southern Germany, Slovakia and parts of the Czech Republic (Figure 4c). However, there was not always enough recolonization area present to compensate for high-risk potential like, for instance, in south-western France and the Po valley in Italy.

The higher the potential exposure of streams to insecticides, the higher the reduction in ER that can be provided by the environment (Figure 5). Hence, the service of ER reduction achieved per recolonization area available was highest in Italy and France, followed by Germany and

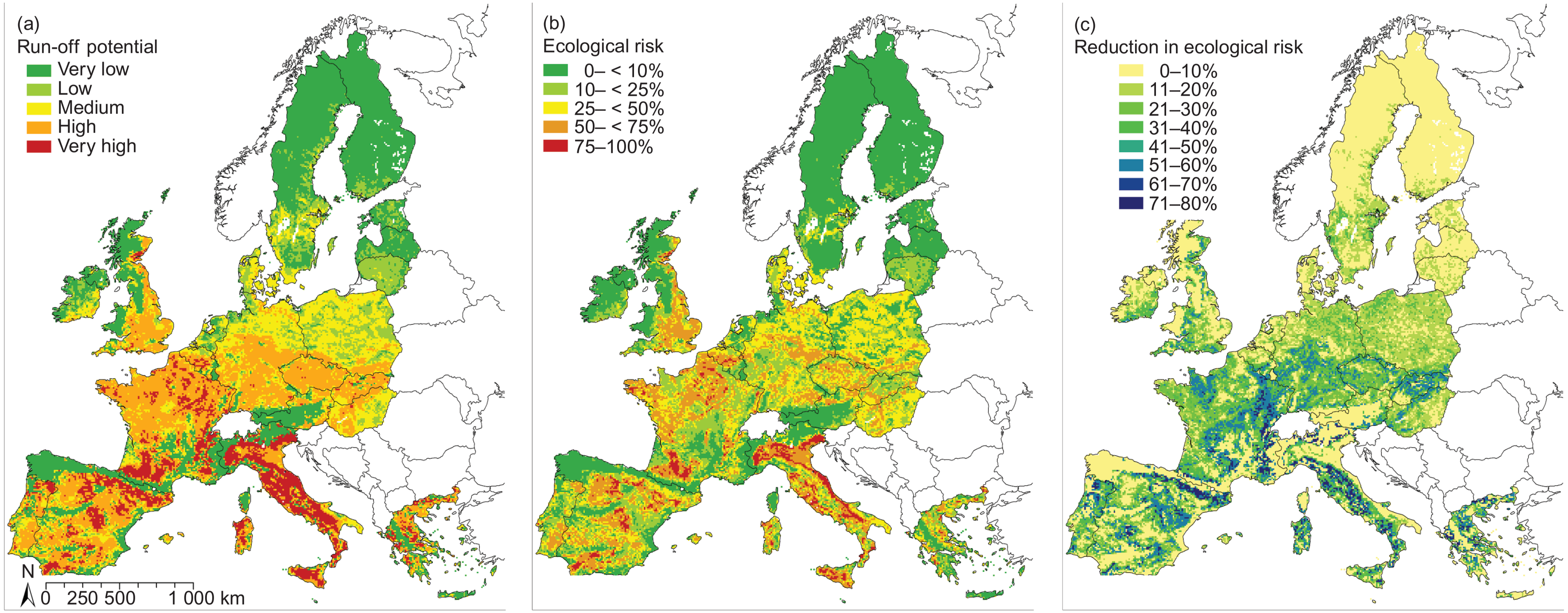

Figure 4. Run-off potential of pesticides (a), related ecological risk (ER) for sensitive species (b) and ecosystem service in terms of reduction of ER (c). ER gives the percentage of stream sites in each grid cell (10 × 10 km) in which the composition of the aquatic community deviated from that of good ecological status according to the WFD.

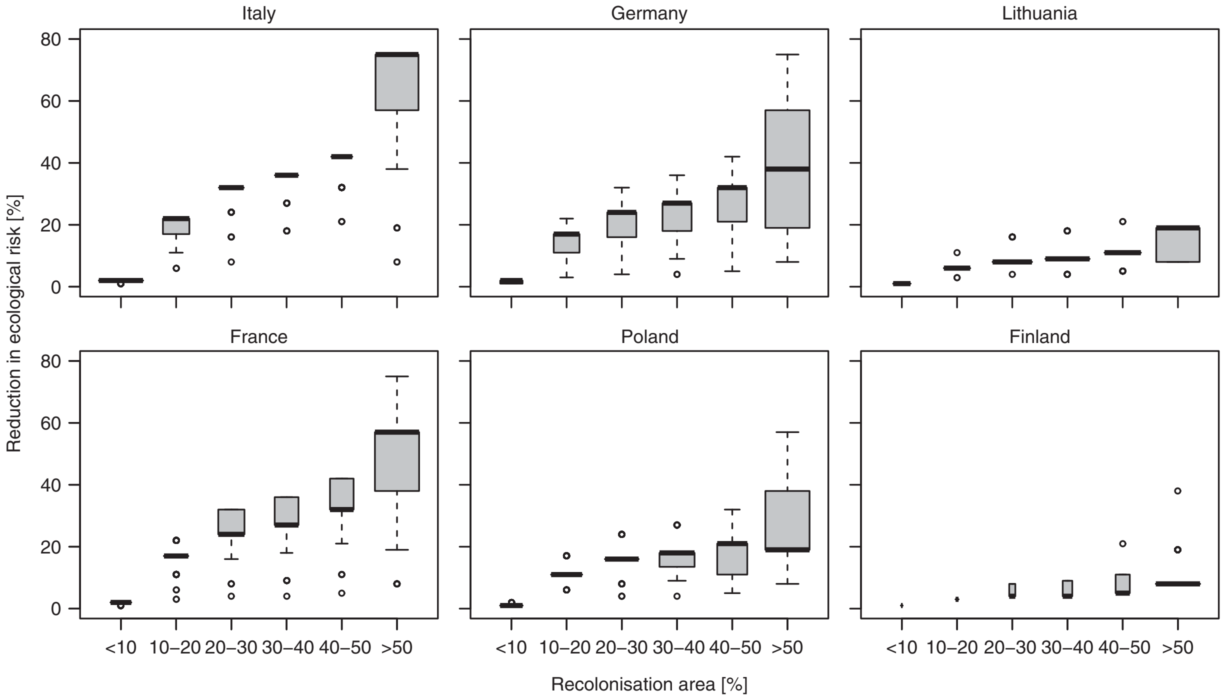

Figure 5. Examples of reduction of ecological risk (ER) for sensitive species due to insecticide exposure as a function of the percentage of recolonization area per grid cell. The higher the potential exposure of streams by insecticides due to high application rates, a high amount of arable area, steep slopes or high precipitation, the higher the reduction in ER that can be provided by the environment.

Poland and comparably low for Lithuania and Finland, where the risk potential is already very low.

Case study 2: mapping nitrogen retention in the German Elbe basin

The ES is estimated based on a comparison of model runs with and without nutrient retention in the catchment and in the river network. From the 748,000 tons of nitrogen applied in the German Elbe basin 470,000 are removed with the harvest (Figure 6). From the remaining 278,000 tons, only 106,000 tons reach the river system – the rest (172,000 tons or 62%) is withheld in the catchment. This can be considered as a regulating service by the environment. From the 106,000 tons that reach the river network by NPEs plus the 23,000 tons that reach the river system from point emissions and 59,000 tons that reach the German part of the catchment from the Czech part, which is not included in the model, 48,000 tons (25.5%) are retained in the river network itself.

At finer spatial scales (Figure 7), we detected a spatial pattern in the distribution of the service. The pattern of soil nitrogen surplus (Figure 7a) shows the highest values (8.6–13.0 t km^{-2} yr^{-1}, highest value class) in the sub-basins close to the mouth of the Elbe (Elbe Marshes, Altes Land and Stade Geest) into the North sea and in the regions of the Ore Mountains and the Vogtland in the southern part of the German Elbe basin. Both regions are characterized by a high density of livestock, which might explain the high nitrogen surplus. Highest nitrogen retention in percentage (of 91–97%, cf. Figure 7b, highest value class) was predicted for large parts of the Havel/Spree catchment in the eastern part of the Elbe basin; the lowest values were found in parts of Berlin due to the high amount of sealed urban areas in this sub-basin. The highest absolute values of nitrogen retention in the catchment (of 5.1–8.5 t km^{-2} yr^{-1}, cf. Figure 7c, highest value class) were located in the sub-basins close to the mouth of the Elbe. It is suggested that the combination of high input surplus together with high to average relative nitrogen retention leads to high values. For this case study, higher demand for the service (higher nitrogen surplus) does imply a higher service provisioning on an average. The different properties of the sub-basins regarding soil properties, topography and groundwater residence time are reflected in a significant scatter of estimated nitrogen retention across sub-basins. This rather diverse spatial pattern is diminished if we consider the estimated values for the whole Elbe basin or the first-order reporting units (results not shown).

A first estimate of the demand for water quality regulation services was compiled in form of the demand for drinking water (Figure 7d). These values consider the demand of households and small and medium businesses but did not incorporate the water demand of industry. Values were mapped to the sub-basins used in MONERIS. The highly populated areas especially in Berlin, around Hamburg and to a lesser extent in the parts of Saxony showed a high demand for drinking water.

The nitrogen surplus values in the Parthe basin were comparable to the corresponding sub-basin in the Elbe case study with values ranging from 2.9 to 4.5 t N km^{-2}. The relative retention values were lower than the values

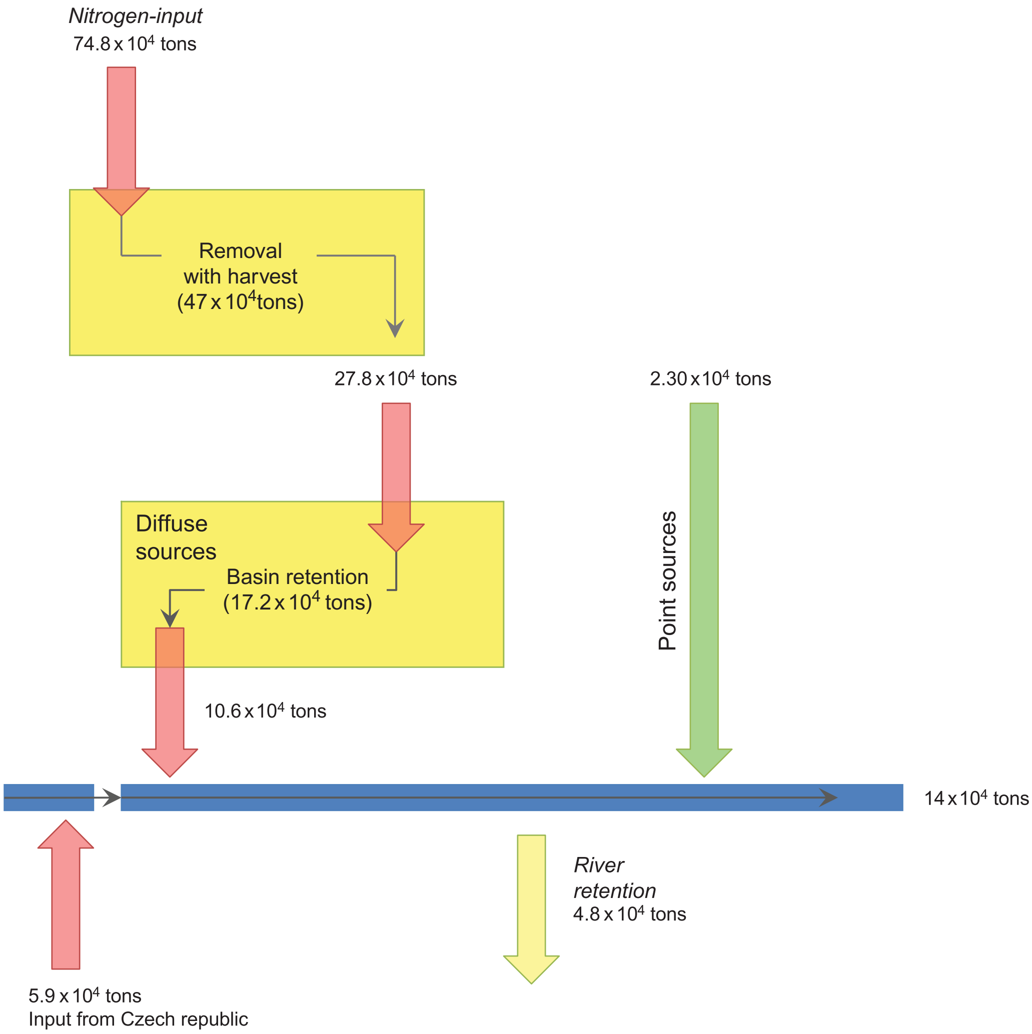

Figure 6. Nitrogen emissions and nitrogen retention services in the Elbe catchment.

estimated in the Elbe case study. The nitrogen surplus pattern is highly dependent on the crop rotation scheme applied in the hydrological response units. The nitrogen retention in the sub-basins is also dependent on the surplus due to management scheme as well as on the spatial pattern of soil types – more fertile cambisols in the North of the basin and more gleyic soils in the South of the basin. Since these spatial structures get blurred and disappear at higher spatial scales, the variability of nitrogen retention decreases with increasing scale. Comparing nitrogen retention values across different catchment sizes (cf. Figure 3, right panel) indicates that variability of nitrogen retention decreases with increasing area. It can be assumed that this is an effect of averaging across physical conditions and management practices at larger scales.

Case study 3: mapping nitrogen retention in floodplains

Case study 3 directly estimates the ES nitrogen retention based on the expert model. For the 'Lödderitzer Forst', the possible nitrogen retention potentials for the HGMUs ranged from 0.01 to 8.96 t N yr^{-1} for the status quo and from 0.01 to 24.2 t N yr^{-1} for the scenario option (Figure 8). In total, the foreland (the area before the current dyke line) retained about 38 t N yr^{-1} and the hinterland (which is currently behind the dyke but would be opened in the planned dyke-shift) about 96 t N yr^{-1}. This leads to an increase of nitrogen retention potential through the dyke replacement of about 253% 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.

Discussion

The three case studies have been presented with two goals in mind: first, to show the potential of ES indicators in addition to classical water quality indicators and second, to illustrate the benefits of a hierarchical approach in such a context. We will first discuss the results of the different case studies and then move on to more general topics on the mapping of water regulation-related ESs in water management tasks. We draw conclusions based on model results since there are no direct measurements of the services. The

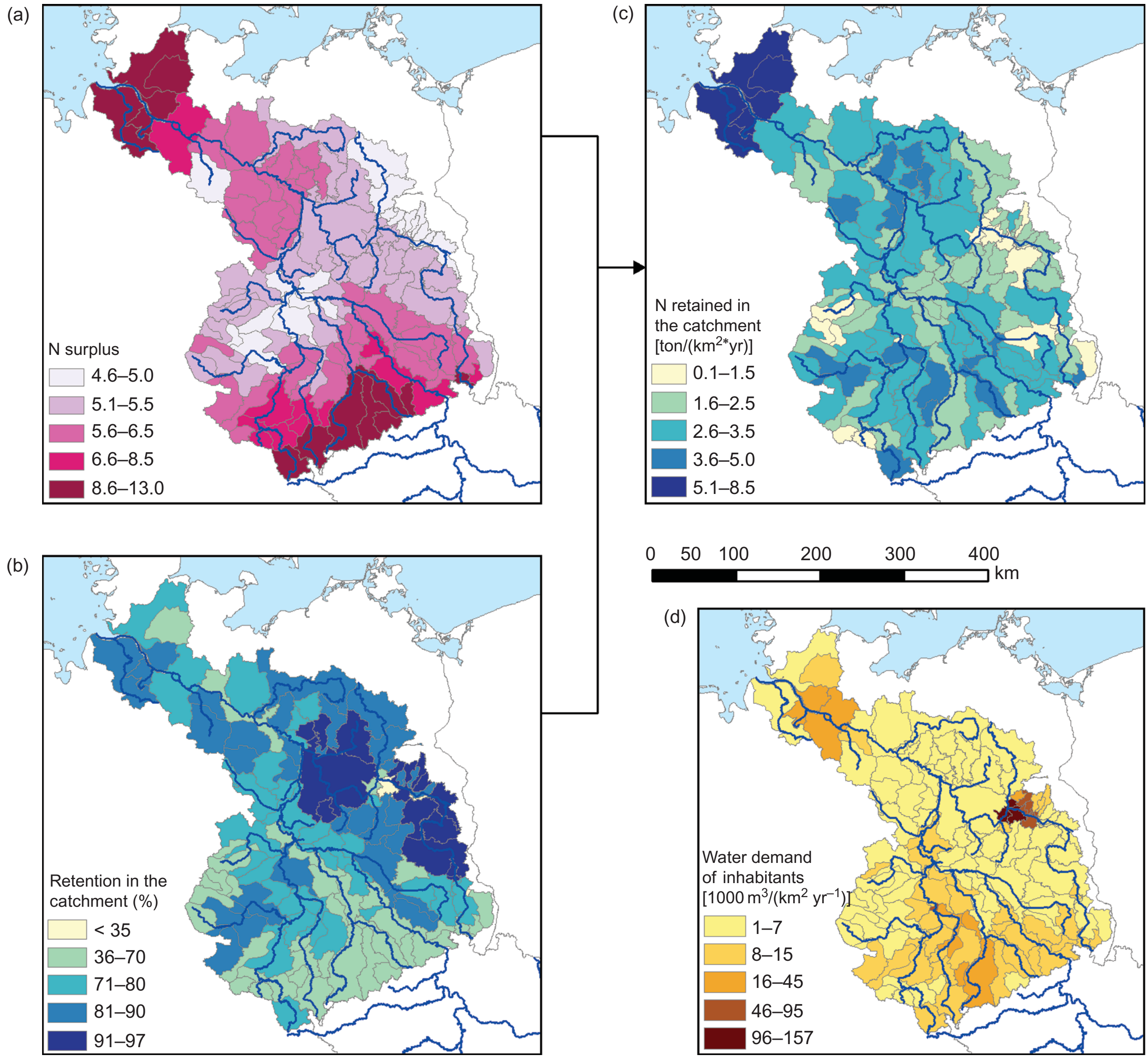

Figure 7. Nitrogen surplus (a) and nitrogen retention in the catchment (b, c) as well as the water demand in the Elbe basin (d) at the resolution of the 132 MONERIS sub-basins.

models for the case studies 1 and 2 have been compared to observed data, which allow an estimation of the uncertainty attached to the results. For case study 3, these comparisons with observed data are still under progress, so the results might be considered more as a proof of concept even if the results are plausible.

Case study 1

The results of case study 1 on pesticide effects at the European level should be read with a focus on general patterns and not on local details. The model relies on the transfer of functional relationships derived from field data to the European scale regarding insecticide exposure, recolonization areas and resulting shifts in community structure. Therefore, model results do not describe the situation at a specific stream site, but report the percentage of adversely affected sites within the 10 × 10 km grid cells. Input data for slope, land use, and soil information were only available at this scale. Likewise, the input data for pesticide use were only available at the national scale; regional differences in insecticide use have therefore not been taken into account. It is important to note that the run-off model takes into account neither substance-specific properties for transport, sorption or degradation nor specific application dates and practices. Such information lacks unfortunately for many pesticides. Hence, the model computes potential insecticide run-off and its effects. In spite of the various simplifications, model predictions were found to match well with measured peak

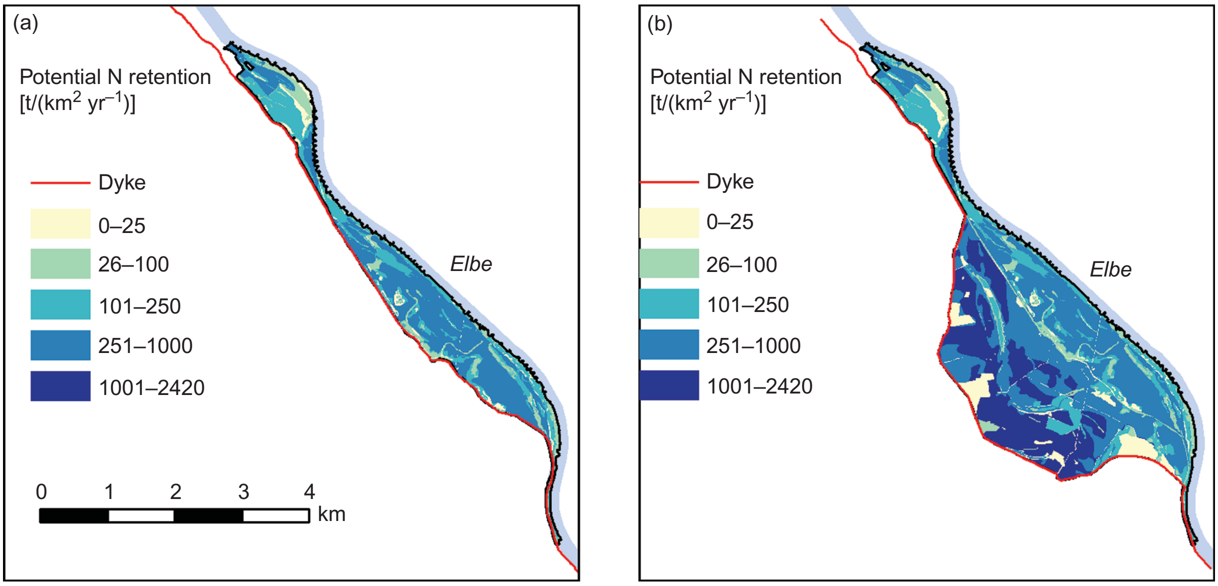

Figure 8. Potential nitrogen retention capacity at the 'Lödderitzer Forst' given the current location of the dyke line (a) and the situation after a dyke-shift scenario (b).

concentration and ecological effects in the field (Schriever et al. 2007a,b).

The aim of case study 1 was to identify streams at high ER as well as areas in which an improvement of water quality by establishing new nursery habitats is most promising. The results indicate that 26% of the cultivated grid cells face at least an ER of 50% for potential long-term effects on freshwater communities due to run-off of insecticides. Over all, the EU-25 countries, 33% of all stream sites in cultivated areas, were predicted not to meet the requirements of the EU-WFD for only slight deviation from undisturbed conditions due to run-off of insecticide. Hence, there is a great potential to improve the ecological quality of agricultural streams to comply with the EU-WFD. In addition to improvements in the timing and efficiency of pesticide application (Caruso 2000) and the establishment of buffer strips along water bodies (Reichenberger et al. 2007), this could be achieved by enhancing the ESs provided by recolonization sites, that is, by establishing new and protecting existing nursery habitats for aquatic species. The areas in which high benefits in the form of a reduction of ER are to be expected can be identified given the model results.

Case study 2

The aim of case study 2 was twofold: first, to distinguish areas according to their service provisioning and second, to estimate the effects of scale on the results. Depending upon which indicator we focus, different regions of the case study region are highlighted as important. Results indicate that the retention in the catchment (during groundwater passage, soil passage or overland flow) is higher than in-stream retention. Still, both systems are important service providers. Point source nitrogen emissions or emissions through sewer overflows can only be handled by the in-stream processes, while the catchment retention is important for the NPEs from agriculture.

The regional modelling approach in case study 2 is based on models of intermediate complexity. These grey-box models (Seppelt 2003) are based on physical processes but have to rely on effective parameters. These effective parameters cannot be measured directly but have to be estimated by confronting the model with observed data (Blöschl and Sivapalan 1995). The Elbe-DSS has been calibrated and validated against discharge data as well as against nitrogen and phosphorus concentrations in the river network – the goodness of fit was determined as good for discharge data and satisfactory for the nutrient concentrations (Lautenbach et al. 2009). The calibration and validation results for the SWAT model for the Parthe catchment were satisfactory as well. The Nash–Sutcliffe model efficiency for discharge was 0.45 for the calibration period (1992–2002) and 0.68 for the validation period (2003–2007), R^2 was 0.54 and 0.72, respectively. Due to the availability of water quality data, nitrogen concentration could only be calibrated for the period from 1999 to 2002 – model efficiency was 0.63 and R^2 was 0.78.

The comparison of values at different aggregation scales shows that information is lost then calculation units are spatially aggregated. Depending on the level of detail of the application, this information loss may or may not matter. Results for the Elbe case study have to be considered as aggregated values for the particular reporting units – that is, drawing conclusions on subunits is not justified.

Uncertainties exist because most of the models require input in form of economic farm data (crop types, crop rotation schemes, management schedules including fertilizer rates, etc.) that are, due to confidentiality laws, only available at aggregated levels (e.g. for municipalities, counties and federal states). Therefore, it is difficult to assess management strategy effects on microscale economic and ecological conditions such as retention capacity. This problem could be solved by an improved cooperation between the relevant authorities, NGOs and research institutes. In addition, there is a lack of long-term water quality time series data on a daily basis and of high spatial resolution, which complicates simulation evaluations.

Case study 3

Case study 3 used an expert model, which incorporates existing knowledge in a look-up table that is used to predict processes based on geodata. The aim of the case study was to examine changes in the nitrogen retention potential in floodplains due to dyke shifts. The results indicate a high effect of such a dyke shift: the nitrogen retention potential in the case study is predicted to increase by 253%. This provides valuable input for land use planers and river basin managers.

The nitrogen retention values are based on a conservative estimate, but the literature reports wide ranges of observed nitrogen retention in floodplains (Jansson et al. 1994; Kronvang et al. 1999; Olde Venterink et al. 2006). Furthermore, the assumption that nutrient retention does substantially depend on the hydrological situation in floodplains is not sufficiently backed-up by observations. Measurements about the size of the flooded area as well as the flood duration exist only locally. Nitrogen retention is also expected to depend on the discharge as well as on the load of flooding events. Since these events are not easy to monitor, it is difficult to reduce model uncertainty. We only know very superficially how changes in the hydrological system, caused by different management options such as changes from farmland into alluvial forest or into grassland, will affect ecosystem functioning. For time being, we can only assume that the future functional relationship in the restoration area will remain the same as in the present active floodplain. We currently have no means to take into account the resilience of a modified landscape and how long such a re-development would take. Hence, there is a strong need of more field investigations for doing a further back-up through data in calibrating models.

Overall discussion

Moving the focus from the water quality *per se* to ESs leads to a different perspective regarding potential management actions or priority areas for nature conservation. Our case study applications identified locations with high ES supply as well as those with a rather low ES supply. The spatial variance of the results is an argument against policy instruments that do not consider spatial diversity. Synergies between water quality-regulating ESs and other ESs such as flood regulation or biodiversity conservation are now identifiable – in contrast to classical water quality indicator maps. Scenario calculations as shown in Figure 8 combined with maps on potential biodiversity and on flood regulation potential allow decision-makers to choose areas for dyke shifting, which increase those objectives simultaneously. The results of Figure 7 could, for example, be used to identify areas in which management actions like erosion prevention are likely to improve water quality based on the level of ES supply. Areas with low service supply might profit most from an increase of the service potential.

However, there are different consequences decision-makers could draw from ESs maps such as the ones presented here: on the one hand, priority areas might be defined for regions with a high service supply to protect the service. On the other hand, priority might instead be given to areas with a low service supply since these locations might profit most from increasing water quality regulation services. One might also argue that high levels of water quality regulation services indicate a high impact on the ecosystems in this location – in other words, these ecosystems might be at an increased risk of collapsing in the near future. In case study 1, large parts of Italy are an example of such vulnerable regions. High potential insecticide exposure is therefore linked to high ESs provisioning. Wide areas in Scandinavia were reported with a very low service supply, but this is due to the low ER in these regions. To tackle these ambiguities, it is important to consider the absolute service value as well as the service value relative to the (potential) pollution.

It is also important to consider the scale at which results have been calculated and at which therefore conclusion should be drawn. Our results for case study 2 show, as expected, that model results at finer scales provides more details than models at coarser scales. Such details might be of great value for smaller-scale decision-making. Following the approach proposed by Steinhardt and Volk (2002), one would identify hot-spots at the large scale, quantify processes at the mesoscale and simulate and implement detailed effects of management actions at the local scale. To support management decisions, ES estimation could be incorporated in decision support systems such as the Elbe-DSS (Lautenbach et al. 2009) or Flumagis (Volk et al. 2007). In such a framework, ES supply for case studies 2 and 3 could be calculated under assumptions such as climate change or major changes in crop rotation schemes, for example, due to increasing biofuel production.

So far, stakeholders and policymakers rated model results as helpful for their work. Stakeholder feedback given during the development of the Elbe-DSS indicated that the spatial and temporal resolution of the results was in accordance with the requirements that stakeholders had in mind (Berlekamp et al. 2005). Concerning the floodplain case study stakeholder involvement resulted in the maintenance and development of the cultural landscape within the Biosphere Reserve Middle Elbe, but also lead to the acceptance of natural floodplain habitats including measures such as dyke relocations (Wycisk and Weber 2003).

Models have been calibrated or tested against hydrologic, water quality and ecological quality measurements but not against ESs. This is due to a lack of data on water regulating services. State agencies measure classical discharge and water quality indicators. Measuring the service itself, for example, nitrogen retention is much more challenging than measuring concentrations in the river system. Therefore, it has to be assumed that such data will not become available in the near future. While model results should be interpreted with respect to the fact that a direct calibration and validation against service data has not been performed, model results seem to be the only available source of information to at least estimate the services.

The shortcomings of model-based ESs mapping have been discussed. However, it should be taken into account that the widespread proxy-based quantification of ESs has recently been shown to be very vulnerable to data deficiency (Eigenbrod et al. 2010). Model applications are typically more explicit about their shortcomings compared to proxy-based studies, due to the widespread practice of describing the goodness of fit of the models during the calibration and the validation periods. Proxy-based approaches tend not to explicitly compare results with observed data.

Future steps for the improvement of mapping water quality regulation ESs should include the incorporation of demand-specific indicators such as the actual demand for water that fulfils specific quality standards. Since water is transported between catchments, it is only a first step to map the demand by households and industry to the same units that are used to model water quality aspects and related ESs. Depending on the way the water is processed, it might be worth focusing on ground water or on surface water. In addition, costs related to technical water purification differ between regions, depending on the technique used as well as on the available infrastructure.

Another important issue for future work is the identification of trade-offs and synergies between water quality regulation ESs and other ESs such as food production and flood regulation. Mapping ESs is an important step but does not automatically identify trade-offs. Extended model analysis as well as the use of optimization techniques could be a useful road to explore trade-off relationships (Lautenbach et al. 2010; Seppelt and Lautenbach 2010).

Maps have been shown to be a valuable instrument to inform decision-makers and stakeholders. They should be added as a tool for use by scientists in the ESs community. Nevertheless, it is important to know about the shortcomings of maps as well and to accompany maps with other forms of information display like scatterplots or time series plots. It should always be taken into account that there are multiple ways of manipulating maps to convey an intended message (Monmonier 1996). Data classification, colour schemes and projection issues are the most common ways of manipulating the perception of maps.

Conclusions

Direct mapping of water regulating ESs delivers important information for policymakers and decision-makers. Production of these maps shows additional information compared to classical water quality indicator maps. It seems favourable to use classical water quality indicators together with ESs focused indicators to support decisions. Unfortunately, studies mapping water quality-related ESs are lacking. We aimed at filling that gap by presenting three case studies. Furthermore, scale is an important issue regarding the spatial variability of results as well as for the processes incorporated in the models. Depending on the management question, finer or coarser scales might be preferable, but, in general, we recommend using a hierarchical approach.

Acknowledgements

This work was partly funded by the PEER project PRESS (http://www.peer.eu/projects/press/) as well as by the Helmholtz Programme 'Terrestrial Environmental Research' (Seppelt et al. 2009). The authors thank all members of the PRESS project for intense discussions during our project meetings. The authors are also grateful to three anonymous reviewers who helped to improve the readability of the text.

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