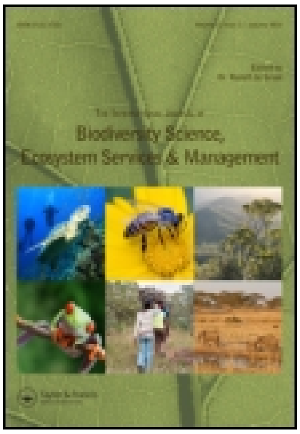

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This case study tested an approach for large-scale monetary valuation of water purification services based on spatially explicit, biophysical mapping of nitrogen retention in river networks. The study area for the case study was the Mediterranean bio-geographical region. The monetary value of nitrogen retention was assessed using replacement costs (RCs), based on the construction, operation and maintenance of constructed wetlands (CWs). The value of nitrogen retention by rivers in 2005 was estimated at €2167 km^{-1}, on average. If the capacity or the potential of rivers and lakes to retain nitrogen was considered, the monetary value was €35,000 km^{-1} on average, much higher than the effective retention. The coupling between a biophysical model simulating the transport and retention of nitrogen in the aquatic environment and monetary estimates of potential and realized nitrogen removal resulted in detailed spatial information of one of the benefits that are provided by rivers and streams.

Keywords: water purification; nitrogen; Mediterranean region; monetary valuation; replacement costs

Introduction

Biodiversity provides vital goods and services, such as food provision, carbon sequestration and water regulation, which underpin economic prosperity, social well-being and quality of life (Millennium Ecosystem Assessment 2005; The Economics of Ecosystems and Biodiversity 2010). In 2001, the European Union (EU) set itself the target to halt biodiversity loss in the EU by 2010. Despite efforts to date, there are clear indications that the target has not been reached (Mace et al. 2010). Several factors have led to this target not being reached, among which the lack of current policies to address the protection and sustainable use of ecosystems and the maintenance of their services. In 2010 the European Commission (EC) proposed a renewed vision and targets for biodiversity for the ensuing period, building on and contributing to the international deliberations on a global vision for biodiversity beyond 2010, which is part of a revised and updated strategic plan for the United Nations Convention on Biological Diversity (EC 2011). In this new strategy, ecosystem services play a key role in the protection of biodiversity. Biodiversity delivers value through the goods and services provided by ecosystems. In particular, biodiversity and ecosystems are linked to human well-being through the flow of ecosystem services. Ecosystems provide the necessary structures and processes which define the capacity or potential to deliver services. In turn, ecosystem services result in benefits for the society (Kienast et al. 2009; De Groot et al. 2010; Haines-Young and Potschin 2010). The proper valuation of ecosystem services is essential to support the EU biodiversity strategy to 2020 that explicitly addresses the spatial assessment of the economic value of ecosystem services as a supporting action to achieve the policy target of maintaining and restoring ecosystems and their services (EC 2011).

Geographical Information Systems (GIS) are indispensable tools to improve the monetary valuation of ecosystem services. In particular, GIS proved to be useful for two frequently applied methodologies in spatially explicit monetary valuation: the application of value transfer as well as the direct inclusion of spatially explicit environmental data as explanatory variables in valuation models and methods. Value transfer (Eade and Moran 1996; Costanza et al. 1997; Troy and Wilson 2006; Boyle et al. 2010; Bateman et al. 2011; Brander and Koetse 2011) involves the adaptation of existing valuation data to new policy contexts where valuation data are absent or limited. The increasing availability of high-quality land-use and land-cover data as well as of different sorts of spatially explicit environmental and socio-economic data has strongly enhanced the applicability of spatial data transfer (Troy and Wilson 2006). Furthermore, GIS is increasingly used to improve the spatially explicit economic valuation techniques of, in particular, recreation services offered by forests and natural areas (Termansen et al. 2008; Bateman 2009). In these studies, travelling distance, travelling time and accessibility represent key variables that explain the variance of monetary values assigned to wild parks, nature reserves or forests, respectively.

Further incorporation of spatial complexity in economic valuation is desirable given the inherently spatial nature of natural resources and ecosystems

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(Bateman 2009). The objective of this study is primarily to test a methodology for monetary valuation of water purification services at the European continental scale supporting the implementation of the EU's biodiversity strategy to 2020. This methodology implies value transfer based on spatially explicit, biophysical mapping of nitrogen retention in surface waterbodies, which is argued to be more accurate than value transfer based on land-cover data only.

The purpose, scale and final users have great influence on the monetary evaluation approach to be used. In addition, valuation depends on the availability of information at the scale and resolution of interest. In this article, we developed a test case which should be applicable at the European scale supplying EU environmental and water policies with information for decision-making and policy implementation. In particular, we applied a spatially explicit valuation methodology on water purification services delivered by rivers, streams and lakes of the Mediterranean bio-geographical region. This service relates to the role that aquatic ecosystems play to filter and decompose pollutants in water. Our specific focus went to problems related with excess nitrogen in surface waters. Rivers and lakes remove nitrogen from the surface water through uptake by plants and phytoplankton, sedimentation and denitrification, hereafter collectively called nitrogen retention. The removal of nitrogen, in particular through denitrification, contributes to an increased water quality in downstream reaches of the catchment and is therefore considered as ecosystem service as it provides benefits to water users. The capacity of surface waters to retain and remove nitrogen was quantified by using a spatial model (Grizzetti et al. 2008). These spatially explicit estimates were subsequently used in the monetary valuation. We applied and compared three different approaches to estimate the value of nitrogen retention by river networks. All three valuation methods were essentially based on value transfer but they differ in the degree of complexity. The first method involved the spatial transfer of monetary values available for different land-cover classes to the whole study area. Two other methods made use of the results of a nitrogen budget model as they adjusted monetary values derived from the literature according to spatially explicit estimates of nitrogen retention capacity and realized nitrogen removal.

Material and methods

Study area

We selected the Mediterranean bio-geographical region as defined for reporting the conservation status of habitats and species under the EU Habitats Directive (EEA 2002) as the study area to test three monetary valuation approaches.

Biophysical assessment of nitrogen retention

We used the model GREEN (Geospatial Regression Equation for European Nutrient losses; Grizzetti et al. 2011) to estimate total nitrogen retention in the rivers and lakes of the study area. GREEN is a statistical model developed to estimate total nitrogen and total phosphorus fluxes to surface water in large river basins. Marine and estuarine waters are outside the scope of the model. The model was developed and used in European basins with different climatic and nutrient pressure conditions and has been applied to the whole Europe (Bouraoui and Grizzetti 2011). The study area was divided into a number of sub-catchments that are connected according to the river network structure. The sub-catchments constitute the spatial unit of analysis. In the application at European scale, a catchment database covering all Europe was developed based on the Arc Hydro model with an average sub-catchment size of 180 km^2. For each sub-catchment the model considered the input of diffuse sources and point sources of total nitrogen and estimates the nitrogen fraction retained during the transport from land to surface water (basin retention) and the nitrogen fraction retained in the river segment (river retention). Diffuse sources included mineral fertilizers, manure applications, atmospheric deposition, crop fixation and scattered dwellings, while point sources consisted of industrial and wastewater treatment discharges. In the model the nitrogen retention was computed on an annual basis and included both permanent and temporal removal. Diffuse sources were reduced by the processes both occurring in the land (crop uptake, soil denitrification and soil storage) and occurring in the aquatic system (aquatic plant and microorganism uptake, sedimentation and denitrification), while point sources are considered to reach directly the surface waters and therefore are affected only by the river retention. For more details on model parameterization and calibration, see Grizzetti et al. (2011).

Monetary valuation based on land-cover data (Method 1)

In the first approach to attribute monetary value to water purification services, we used the CORINE (Coordination of Information on the Environment) land-cover (CLC) database made available by the European Environmental Agency (EEA) coupled with a valuation database. The value (€ ha^{-1}) for water purification was retrieved from the cost of policy inaction (COPI) study (Braat et al. 2008). Monetary values were provided for ecosystem services classified per geographical area and per land-cover type (forests, grasslands and steppe, scrubland, wetland, other woodland). We applied a simple unit value transfer (Boyle et al. 2010), which subsequently served as a reference. This reference was compared with two other valuation procedures for which the outcomes of the GREEN model provided the basis for monetary valuation.

Monetary valuation based on actual nitrogen retention (Method 2)

The second approach for the monetary valuation of water purification was based on replacement costs (RCs) and used the results of the GREEN model. We estimated the value of nitrogen retention by using the RCs based on an assessment of constructed wetlands (CWs). CWs are engineered systems that have been designed to use the natural processes in treating wastewater and they are considered as the best technical alternative to retain and remove nitrogen at low concentrations (Vymazal 2007; Li et al. 2009; Masi 2009). The choice for using costs of CW for valuing in-stream nitrogen retention is related to the large contribution of two particular nitrogen sources. Diffuse nitrogen emissions from the agricultural sector and point emissions from effluents of wastewater treatment plants are dominant contributors to the combined sum of nitrogen emissions to river networks (Grizzetti et al. 2011). Nitrogen from these two sources arrives in rivers at low concentration that justifies the choice for using CW-based costs in our analysis. CWs are indeed applied worldwide to purify wastewater derived mainly from agriculture and the residual effluents from wastewater treatment plants treating wastewater from industry and households (Kadlec and Wallace 2009). Cost estimates were derived from Cuttle et al. (2007) who presented a UK inventory of method to control diffuse water pollution from agriculture (Table 1). The inventory was used to calculate the costs for CWs per kilogram nitrogen. The handbook specifies for different farm systems the capital and annual costs necessary to build and manage a CW. We applied the costs per hectare of the arable option after applying a ratio to the sub-catchment in order to consider only the retention performed by waterbodies and not by the land. On average, this resulted in a conservative value of €2463 ton^{-1} (in year 2000). This value was subsequently multiplied with the total amount of nitrogen (ton) that was retained in each sub-catchment of the study area.

Monetary valuation based on potential nitrogen retention (Method 3)

The third approach addressed a different conceptual valuation framework and a different way to calculate monetary values. We estimated the capacity of river networks to remove nitrogen, which we called potential nitrogen retention and we used it as the basis for the valuation. Potential nitrogen retention was expressed as a fraction of nitrogen loading. The product of potential nitrogen retention and the total nitrogen river loading (ton year^{-1}) is equal to the actual nitrogen retention which was used for the second approach. The capacity of aquatic ecosystems to remove nitrogen is a function of nitrogen loading itself where increasing nitrogen concentration reduced the nitrogen uptake and denitrification velocity (Mulholand et al. 2008). We accounted for the negative effect of nitrogen loading on removal capacity assuming a slight decrease of the potential nitrogen removal as a function of nitrogen input:

$$N_p = N_{p^0} \times \left(1 - 0.0019\left[\frac{I}{2.1864}\right]\right)$$

where N_p is the nitrogen retention capacity, N_{p^0} the fractional nitrogen retention based on the GREEN model and I the input of nitrogen to the sub-catchment (10^4 kg). The result of this function is that at maximum nitrogen input (600 kg km^{-1} year^{-1}) N_p corresponds to 50% of the fractional retention capacity in the absence of nitrogen input.

The spatially explicit estimates of N_p were subsequently coupled with monetary values by assessing construction, operation and maintenance costs for CWs. There are several types of CWs: surface flow wetlands, subsurface flow (SSF) wetlands and hybrid systems that incorporate surface and SSF wetlands. A surface flow wetland, hereafter referred to as free water system (FWS), consists of a shallow basin, soil or other medium to support the roots of vegetation and a water control structure that maintains a shallow depth of water. They are mainly used to treat agricultural run-off. An SSF wetland consists of a sealed basin with a porous substrate of rock or gravel and the water level is designed to remain below the top of the substrate. They are mainly used as secondary treatment for domestic and industrial wastewaters. Using GREEN, we calculated per sub-catchment the proportion (α) of point sources of nitrogen and the proportion of diffuse sources of nitrogen ($1 - \alpha$). We used α to split N_p over nitrogen retention capacity to reduce diffuse sources (N_{pd}) and nitrogen retention capacity to reduce point sources (N_{pp}) so that

$$N_{p^d} = (1 - \alpha) \times N_p \, and \, N_{pp} = \alpha \times N_p.$$

Then, we attributed monetary value to N_{pd} assuming operation and maintenance costs of FWS; likewise the valuation of N_{pp} was based on a cost assessment of SSF. An important cost that we do not consider in our study is the price of land; the monetary proxies we present represent a minimum threshold of value that could be greatly increased if the price of land would be added. The price to be considered would mostly be based on the value of agricultural land, which in this case is the opportunity cost because the construction of FWS requires several hectares of land, while the construction of SSF has only low requirements in terms of space.

Table 1. Value of nitrogen retention for different case studies.

Methodology	Value (€ ton^{-1})	Reference
Replacement cost/constructed wetland	2463	Cuttle et al. (2007)
Replacement cost/contingent valuation method	21,180	Gren (1995a)
Replacement cost/denitrification and load reduction	3706	Gren (1995b)
Energy analysis/replacement cost	17,792	Gren et al. (1994)

Differently from the second method, we calculated the construction, operation and maintenance costs based on a number of biophysical features that were derived from the GREEN model. We used an engineering compendium of all the costs related to different building and management steps (Kadlec and Wallace 2009). A general price list for building and maintaining CWs does not exist. Instead we collected case studies through a survey of literature (Table 2). For FWS we considered experiences in the United States due to a lack of European applications; for other systems we retained applications situated in the Mediterranean area. The main differences that could incur when considering two different countries in transferring one price from the study site to another site mainly refer to labour force, which for FWS does not represent the main cost item. The rest of the costs are very variable but not in relation to the country, but rather to the specific local site where the CW is built (also within the same country).

There are considerable price differences depending on the system design, material needs or geo-morphological conditions. We decided to create three price classes (low, moderate, high) within which we calculated an average price for the construction and for the operation and maintenance (Table 2). The annual total costs were calculated considering a time horizon of 20 years and a 5% interest rate. We included the economy of scale in CW construction since a negative relationship was assumed between wetland size and construction cost per unit of area. Hereto, we performed a regression analysis of the cost per unit area using area as explanatory variable. Finally, we divided the sub-catchments into three groups according to their retention capacity, on the basis of the 25th and 75th quartiles (i.e. <25; 25> and <75; and >75 percentile). The three groups determined the sub-catchments that receive a low, moderate or high monetary value according to Table 2.

Table 2. References for case studies used to calculate the cost of construction, operation and maintenance of CWs and classification of the costs in three different price classes.

FWS	Reference
Carolina Bays (USA)	US-EPA (1993)
Houghton Lake (USA)	US-EPA (1993)
Cannon Beach (USA)	US-EPA (1993)
Vermont Ville (USA)	US-EPA (1993)
Las Gallinas (USA)	US-EPA (1993)
Hayward Marsh (USA)	US-EPA (1993)
Orlando (USA)	US-EPA (1993)
Lakeland (USA)	US-EPA (1993)
Incline Village (USA)	US-EPA (1993)
Venice (Italy)	Mannino et al. (2006)
Gustine (USA)	US-EPA (1999)
Arcata (USA)	US-EPA (1999)
Ouray (USA)	US-EPA (1999)
SSF systems	
Gomati (Greece)	Tsihrintzis et al. (2007)
Catalonia (Spain)	Puigaguta et al. (2007)
Andalusia (Spain)	Puigaguta et al. (2007)
Galicia and Castilla–Leòn (Spain)	Puigaguta et al. (2007)
Average values for three price classes (FWS)	€ ha^{-1}
Low	2200
Moderate	8000
High	12,300
Average values for three price classes (SSF)	€ ha^{-1}
Low	179,000
Moderate	203,000
High	218,000

Note: CW, constructed wetland; FWS, free water system; SSF, subsurface flow.

Where necessary we converted foreign currency to euro using the average annual exchange of the reference year. We accounted for inflation using the historical consumer price index for the Eurozone. All the monetary values were reported in terms of constant price referred to the year 2000.

Results

Three maps covering the Mediterranean bio-geographical region present a monetary valuation of nitrogen retention services at the landscape scale of (Figure 1) and at catchment scale (Figures 2 and 3).

The results of the first monetary valuation method, based on land-cover data, are available for 1990 and 2000, corresponding with the years during which updates of the CLC data were available (Figure 1). They were obtained by multiplying the monetary value per hectare attributed to the ecosystem service by the number of hectares corresponding to the different land cover. On average, the value of water purification was €139 ha^{-1} for 1990 and decreased to €127 ha^{-1} for 2000. The decrease is the result of land conversion with increasing artificial and agricultural land use at the costs of forests and wetlands.

The results of the second valuation method are shown in Figure 2 for the years 1990 and 2005. Here we multiplied the quantity of nitrogen removed (ton km^{-1}), estimated by the model GREEN, with the cost per ton of replacing the nitrogen retention service of rivers with artificial wetlands. The value of nitrogen retention was, on average, €2218 km^{-1} of river stretch for 1990 and decreased to €2167 km^{-1}, on average, for 2005. Figure 2 shows that the value attributed to nitrogen retention increased in downstream direction with values near to €0 km^{-1} for head waters and high values for downstream catchments. Maximum values were for catchments discharging into the sea reaching a high of €318,000 km^{-1} for 1990 and €238,000 km^{-1} for 2000.

In the third valuation method we valued the capacity of river networks to retain nitrogen. We obtained monetary valuation estimates for each catchment depending on its size, on the ratio between diffuse and point sources and on the value of nitrogen retention capacity. The resulting map is depicted in Figure 3. On average, we estimated that the value of nitrogen retention capacity was €31,560 km^{-1} for 1990 and increased to €35,000 km^{-1} for 2005. Decreasing nitrogen emissions increased the retention capacity and hence, the monetary value that was assigned.

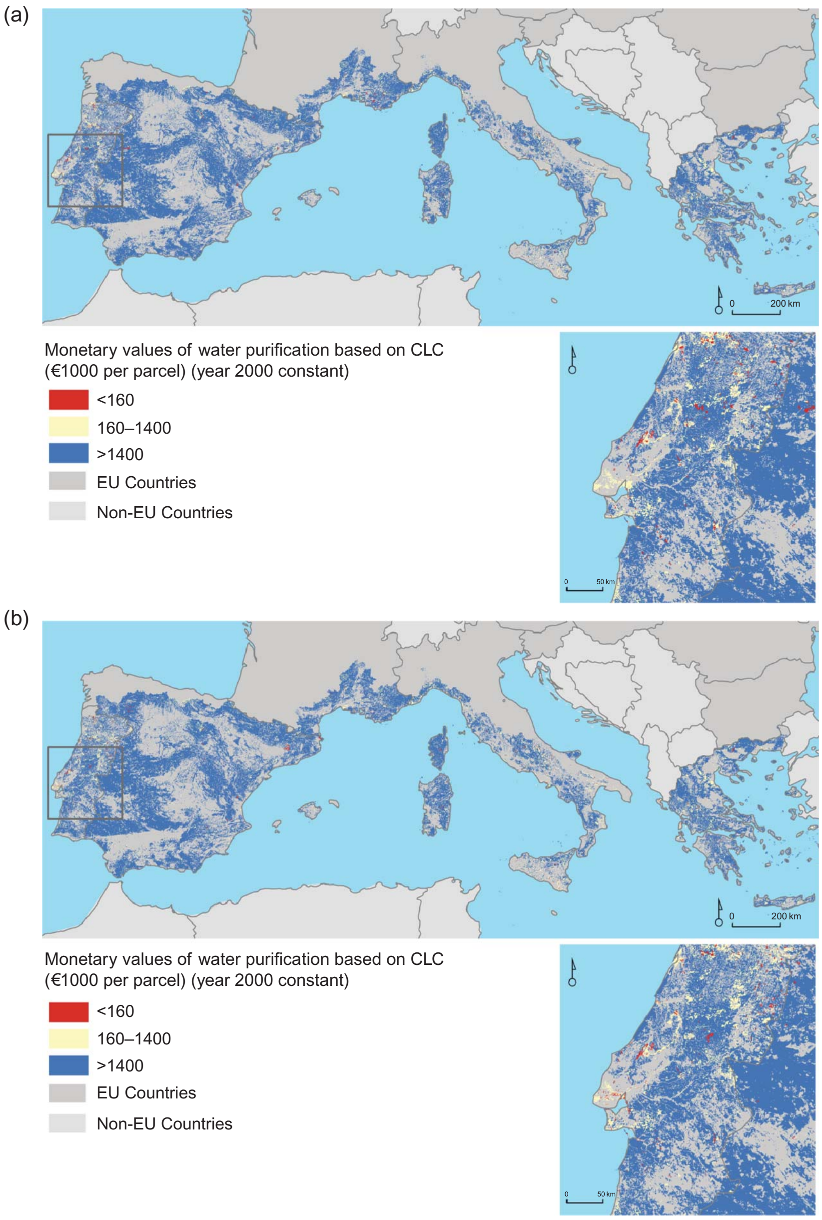

Figure 1. Spatially explicit monetary valuation of water purification services according to Method 1 (value transfer) for 1990 (a) and 2000 (b) based on the CORINE land-cover (CLC) map.

Aggregated over the study area using 1990 as the reference year, the value of water purification was estimated at €5.910^9 according to the first method based on land-cover and value transfer data, €4.110^9 according to the third method based on capacity of river networks to remove nitrogen and €0.1310^9 following the second method when only actual removal of nitrogen by rivers and lakes was considered.

Discussion

Spatial explicitness of monetary valuation of ecosystem services is of key importance to deliver evidence-based information for policies addressing the use of natural resources. There is a need to understand better where ecosystem services are produced and where the benefits are enjoyed as well as to take these spatial service flows into account in cost–benefit analyses or ecosystem-accounting frameworks (Naidoo et al. 2008; The Economics of Ecosystems and Biodiversity 2010).

In this article we complemented the spatial valuation approach based on value transfer across similar land-cover classes with a value transfer approach based on the results of a biophysical model that simulates the nitrogen losses which occur in river networks. Value transfers have become a standard practice to evaluate benefits of ecosystem services (Bateman 2009; Boyle et al. 2010), but the methodology is not uncontroversial (Troy and Wilson 2006). Clearly, primary valuation studies are the best available methods to estimate the value of ecosystems services but such studies are time-consuming, in particular if the scale of our study site is considered. An important problem in spatial value

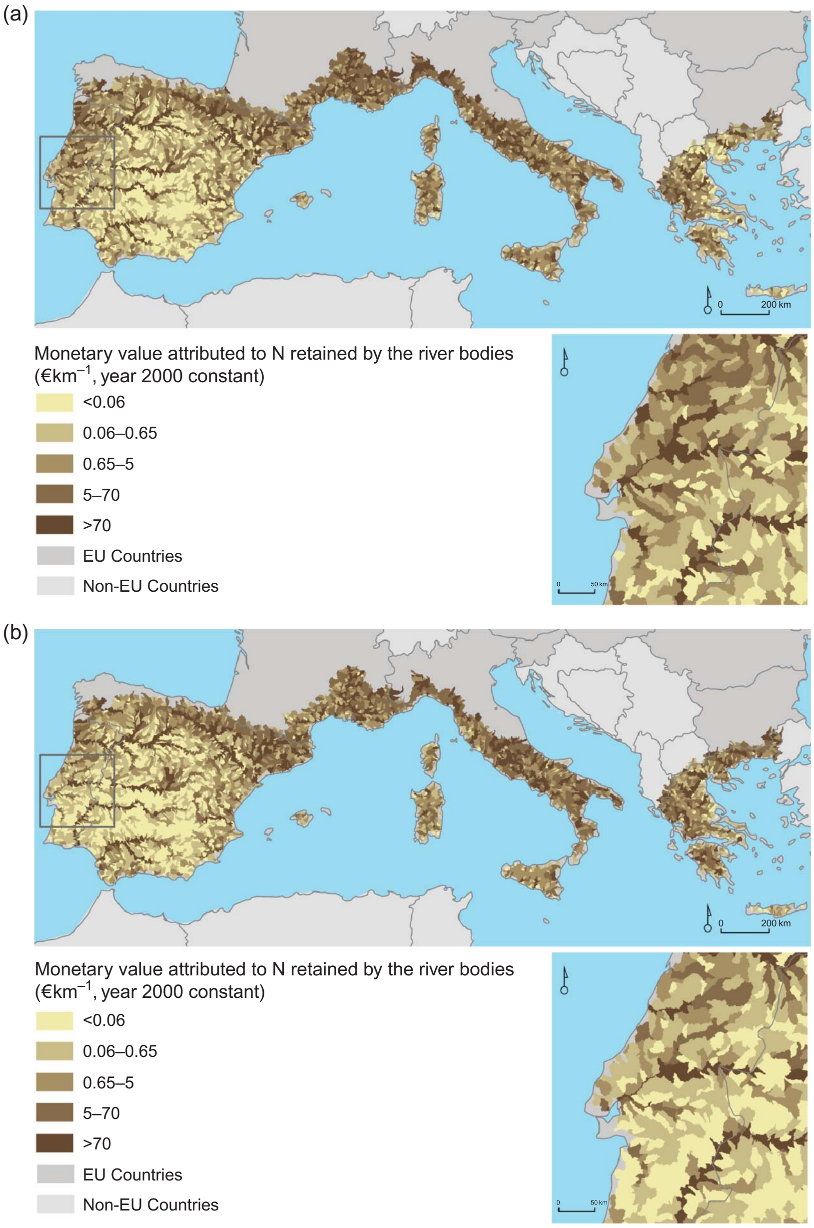

Figure 2. Spatially explicit monetary valuation of nitrogen retention per sub-catchment according to Method 2 (nitrogen retention assessment using the GREEN model) for 1990 (a) and 2005 (b).

transfer is that values are taken from study sites with a very different context than the site of interest (Troy and Wilson 2006). An advantage of our study was the application of a validated biophysical model which enabled transfer values for nitrogen retention across the study site using an equation that scales monetary value to biophysical flows. We believe that this approach is more accurate than the approach based on value transfer using land-cover data.

The advantage of a land-use-based approach with value transfer is its simplicity, which permits a fast and cheap assessment. However, the specific biophysical processes that result in a particular ecosystem service are not captured well by land-cover data alone. The difference between the values over the time period considered was due to changes in land use with an increment of artificial and agricultural land use. Nitrogen inputs were not considered in the methodology so the land-use-based valuation approach assumed uniform distributions of the polluting source. Furthermore, the effect of upscaling the results of single, local case studies to the regional or continental scale is unknown and may lead to erroneous estimates.

The second and third methods applied in this study overcome to some extent of these problems. Our valuation was based on a validated biophysical model at the scale of Europe operating at a relatively fine spatial resolution. To value the biophysical estimates of nitrogen retention in river networks, we used a cost-based approach. The main argument for not using a damage-based approach (Crutchfield et al. 1997; Meyerhoff and Dehnhard 2007; Dodds et al. 2009) lies in the difficulties of exhaustively

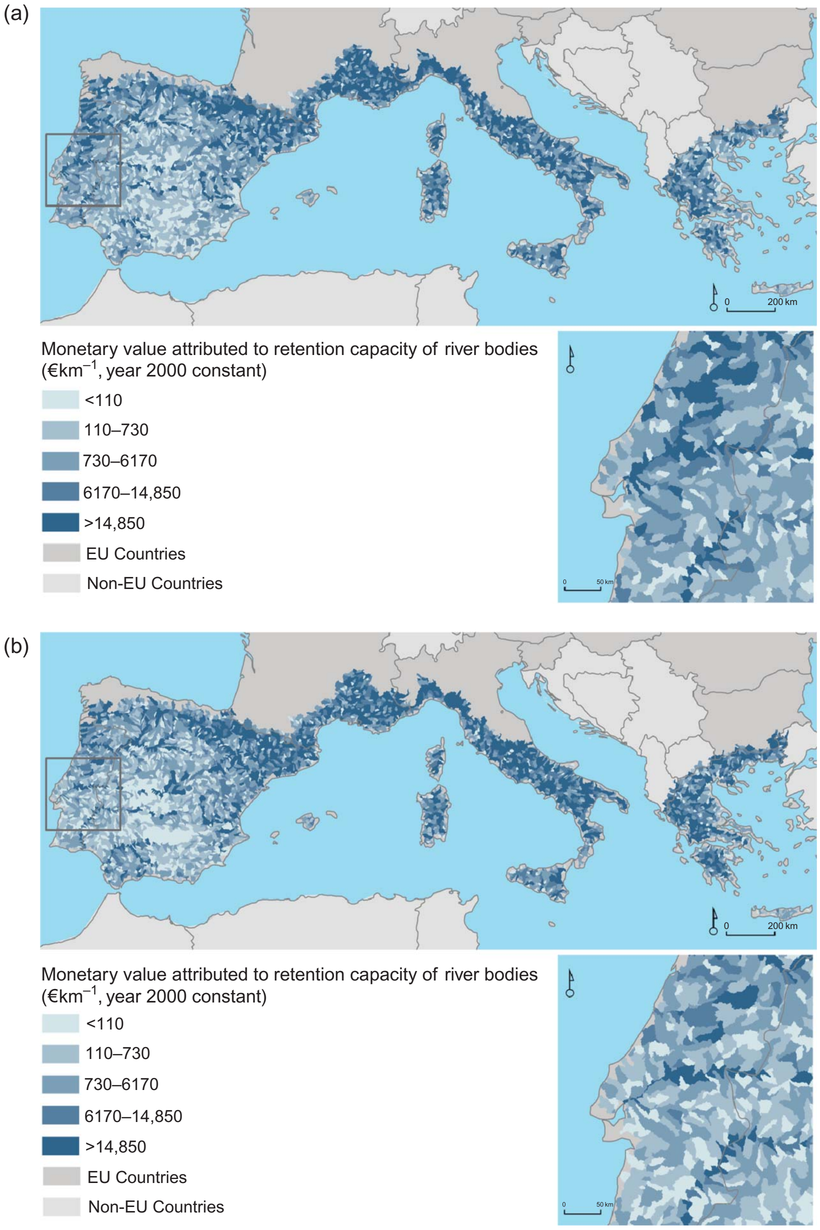

Figure 3. Spatially explicit monetary valuation of capacity of river networks to retain nitrogen per sub-catchment according to Method 3 (potential nitrogen retention assessment using the GREEN model) for 1990 (a) and 2005 (b).

identifying all the benefits that could be lost if river networks do not contribute to water purification. These benefits range from the availability of clean water for drinking, irrigation, recreation, fishing or simply the aesthetic perception that influences both recreational activities and real estate markets. The risk of not considering all the benefits involved would make the valuation incomplete. We thus preferred to value nitrogen retention by considering the hypothetical cost to replace the service that the ecosystem now provides at no cost.

The approach we used to value nitrogen retention (RC-CW) is based on a conservative estimate out of a limited range of other values reporting on the implementation of wetlands for nitrogen abatement (Table 1). In these studies (Gren et al. 1994; Gren 1995a, 1995b), RCs were compared with the benefits of improved water quality which positively influenced aquatic biodiversity. These studies considered also the values related to biodiversity and self-organizing capacity of the ecosystem. The value that we have used in this study considered only nitrogen retention by inland waterbodies as a proxy for the water purification ecosystem service and we thus referred to a very specific, be it indirect, benefit which justifies using a conservative estimate for nitrogen removal. The CW-based valuation approach represents only hypothetical costs targeted to replace the service of water purification in case this is not any longer supplied by a degraded ecosystem. CW could provide additional ecological benefits, but those can be accounted for when a project is effectively carried out in a specific context; otherwise, we consider the positive externalities of CW as an additional reason to choose this typology of costs to value an ecosystem service.

The third valuation approach that was applied in this study attempted to overcome some of the problems that arise when nitrogen retention is used as indicator for water purification services. Increasing nitrogen input from different sources increases the nitrogen loading to rivers and in turn, the total nitrogen retention increases accordingly. Since we preferred to value the service (nitrogen retention) and not the benefit (clean water), increasing nitrogen input to watersheds increased the monetary value that is attributed to the service. Given the results of the second method, there was a decrease in the monetary value following a decrease in nitrogen inputs to the study area and thus an associated decrease in nitrogen retention by the river network. Moreover, as rivers became loaded with nitrogen in downstream direction, so increased the retention of nitrogen. Therefore, in the third valuation approach, we considered the capacity of river networks to remove nitrogen irrespective from nitrogen input and nitrogen loading. Put another way, the second valuation approach addressed the actual service flow provided by river networks, whereas the third approach valued the potential service or the capacity to provide a service given environmental conditions. The main difference between both approaches was that the change in monetary value took opposite directions if the nitrogen input changed. Increasing nitrogen pressure resulted in increasing value per kilometre river stretch under the second approach and decreasing value per kilometre river stretch under the third approach. The increment of the monetary value for the third method over the considered time period is a result of reduced nitrogen input that was modelled to positively affect the capacity of rivers to retain nitrogen. The results of the comparison suggest that if we considered only the nitrogen retained as the proxy for the flow of the ecosystem service the latter would be underestimated. In fact, we would value mainly the flow of the ecosystem service in response to a human pressure without considering the ecosystem capacity to provide the service, and how this capacity can deteriorate under high pollution pressures.

Summed over the whole study area, the capacity of rivers and lakes to remove nitrogen yielded an estimate for total value that had the same order of magnitude than the estimate based on value transfer (first method) for wetlands and several other terrestrial ecosystems. However, rivers and lakes are not the only sinks of nitrogen that are present in landscapes. Ecosystems with wet soils such as forests and grasslands but also soils agricultural land are important denitrifiers capturing, processing and removing a substantial part of the nitrogen present in run-off water (Pinay et al. 2000). Nitrogen removal that takes place in these ecosystems was not captured by the modelling approach used in the second and third methods. Including terrestrial nitrogen retention would considerably increase the total value of water purification services delivered by landscapes.

This study is a first attempt to propose and compare methodologies to value water purification as ecosystem service on a large spatial scale. At present, the valuation still lacks a sufficient number of studies reporting on construction and operation and maintenance costs of CWs at appropriate geographical resolution. Further research must also address the sustainable use of water purification services. The main question is when the capacity of aquatic ecosystems to remove nitrogen is fully reached and at which critical loadings or threshold concentrations unsustainable quantities of nitrogen are retained.

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