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Biophysical Assessment and Monetary Valuation of Ecosystem Services

Scenario analysis for the case of water purification in Europe

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ACRONYMS

BAU	Business As Usual Scenario
CICES	Common International Classification of Ecosystem Services
CWs	Constructed Wetlands
DALY	Disability Adjusted Life Year
EEA	European Environmental Agency
Eurostat	European Statistical Office
FWS	Free Water System
GREEN	Geospatial Regression Equation for European Nutrient losses
GDP	Gross Domestic Product
HF	Horizontal Flow
LUMP	Land use Modelling Platform
MA	Millennium Ecosystem Assessment
MANU	Manure Scenario
N	Nitrogen
O&M	Operation & Maintenance costs
P	Phosphorous
PE	Population Equivalent
REF	Reference Scenario
TEEB	The Economics of Ecosystems and Biodiversity
WWTP	Waste Water Treatment Plants

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1. Introduction: ecosystem services assessment and valuation

According to the Millennium Ecosystem Assessment (MA, 2005) there is a direct link between ecosystem services and human well-being: ecosystems provide goods and services which are beneficial for people whilst human actions have a direct or indirect impact on ecosystems. The conceptual shift with respect to human impacts concerns the move from sectoral problems such as air pollution, water pollution, resource depletion, and renewable resource overexploitation to more complex approaches considering global problems related to climate change, the nitrogen cycle and biodiversity.

The EU Biodiversity Strategy to 2020 (COM(2011)244 final) emphasises the importance of biodiversity in delivering ecosystem services that underpin human well-being. It is recognized that those services have significant economic value but this value is seldom captured in markets. Target 2 of the Strategy requires to maintain and restore ecosystems and their services and in order to do that Action 5 explicitly calls for the improvement in the knowledge that lies in the mapping of ecosystems and their services and the assessment of the economic value of such services. Moreover the Roadmap to a Resource Efficient Europe (COM(2011)571 final), when it comes to nature and ecosystem services (§ 4.1), establishes as milestone the mapping of ecosystem services and their economic valuation. The biophysical assessment through mapping and its economic valuation are thus considered of great importance when it comes to sustainable development and conservation policies. However, the definition and assessment of ecosystem services needs some further introduction.

The notion of ecosystem services, raised by Costanza and Farber (1997), Vitousek (1997), Daily (1997), Mooney and Ehrlich (1997) during the end of ’90 and spread by the MA (2005), has been used as conceptual basis for the TEEB initiative (ref. http://www.teebweb.org/). One main conceptual issue addresses the pathway from ecosystems to humans by differentiating biophysical structures, functions, services, and benefits. In particular, functions are defined as what is needed to deliver services; services are what ecosystems do for the people; and benefits effectively satisfy human needs and wants (TEEB, 2009). Figure 1 shows this cascade-frame endorsed by TEEB (this is not the one that is in the TEEB).

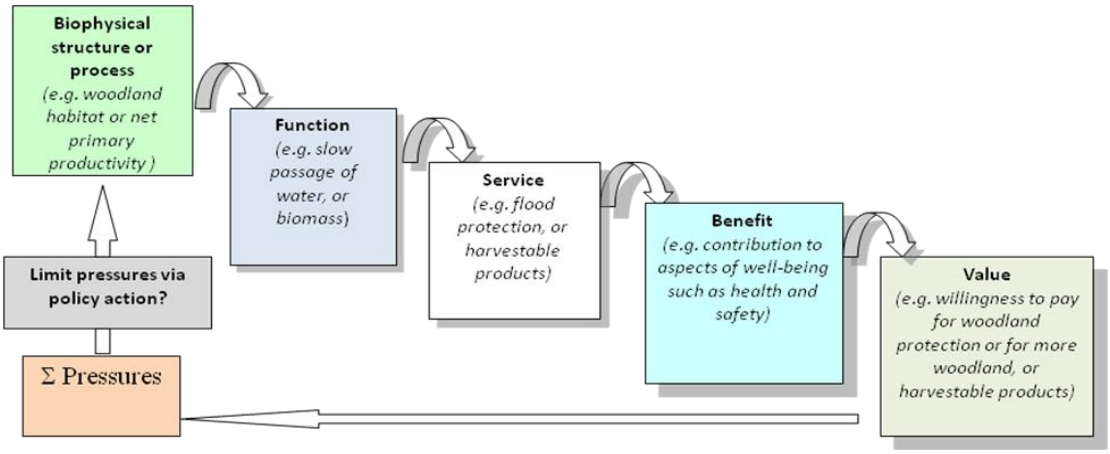


Figure 1. The cascade framework

We use the ecosystem services cascade framework to argue that we can value ecosystem services rather than the benefits that are derived from them (which in most cases is represented by resources and commodities) because this allows a straightforward link to the biophysical assessment. We thus take a different point of view with respect to other researchers (Wallace 2007, Boyd and Banzhaf 2007, and Fisher and Turner 2008). Benefits are crucial to understand why ecosystem services are important but the service is the flow to be assessed in order to be consistent with the initial purpose of valuing for conservation purposes (as required by COM(2011)244 and COM(2011)571). Moreover, from the sustainability point of view, especially when dealing with regulating services, what we need to look at is not only the effective flow of the service, i.e. what is currently used by the society (economic sectors and households), but also the capacity of the ecosystem to generate that service. The initial cascade model is thus employed in our exercise as follows (Figure 2).

In the case study we present in this report we value the ecosystem service *water purification*, according to *The Economic of Ecosystems and Biodiversity* classification of ecosystem services (TEEB [eds. P. Kumar], 2012) and the *Common International Classification of Ecosystem Services* (CICES, ref. http://cices.eu/) supported by the European Environmental Agency (EEA).

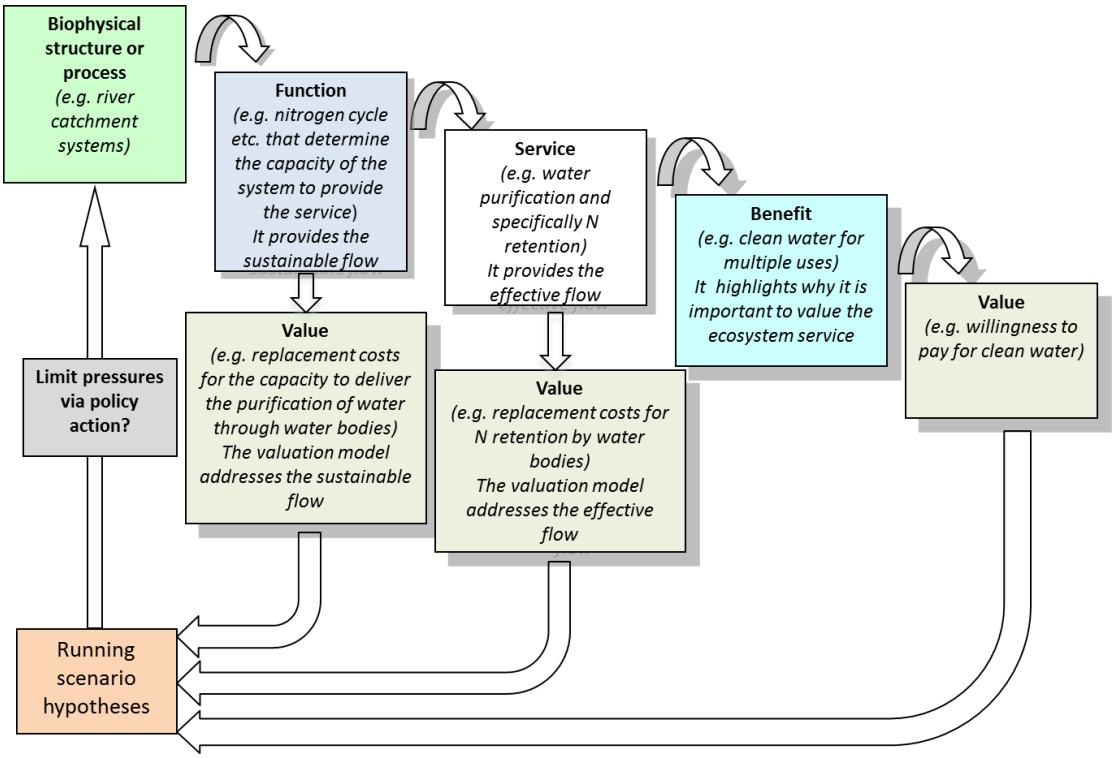


Figure 2. The *adapted* cascade framework applied to water purification services using nitrogen as common water quality metric

The estimated economic value will serve as a tool for scenario analysis. What matters are not the absolute value that is calculated but rather the amount and direction of the change when comparing different policy options. We will thus calculate the economic value for water purification for the baseline year (2005) based on results from a biophysical model and using two scenarios: a business as usual in 2020 and a manure optimization policy in 2020 in both physical (§ 2) and monetary (§ 3) terms. The obtained results (§ 4) will be presented and commented together with the draft assessment of an additional valuation item (i.e the cost of land acquisition) and a sensitivity analysis. This study represents an initial attempt to assess and value ecosystem services at European scale in response to Action 5 of the Biodiversity Strategy. It is thus important to track the important lessons we learn from it, about its weaknesses and about its useful inputs for further research development (§ 5).

2. Biophysical Assessment

Water purification is an important ecosystem service. Ecosystems, in particular wetlands, lakes, rivers and floodplains, have the capacity to retain, process and remove pollutants, sediments and excess nutrients. This avoids pollution of downstream waters and more importantly, it contributes to the provision of clean water for multiples uses.

The service of purifying water takes place in surface waters where usually primary activity (agriculture and breeding) discharge together with primary treatment residuals from industries and households. Considering that fresh water for drinking is usually extracted from groundwater tables, the water purification service is in many cases more related with pollution mitigation from economic activities than with the availability of freshwater for people.

In this study nitrogen serves as a common water quality metric. Excessive nutrient loading is a leading cause of water pollution worldwide (Cardinale 2011). The nitrogen balance of the planet is disturbed as a result of increased nitrogen fixation for the production of artificial fertilizers and through combustion of fossil fuels. This excess nitrogen runs in rivers, streams and lakes where it contributes to eutrophication. Especially shallow coastal zones such as estuaries, deltas and marine embayments are vulnerable to eutrophication which leads to anoxia and the bloom of harmful and toxic algae. Given the impact of the altered nitrogen cycle on the environment, policies are put in place to control the input of nitrogen to river basins. At European scale two important directives are the Nitrates directive designed to protect the EU's waters against nitrates from agricultural sources and the Water Framework Directive aiming at good water quality by 2015. Large-scale nitrogen budgets show that an average of about 20–25 per cent of the nitrogen added to the biosphere is exported from rivers to the ocean or inland basins (Mulholland et al. 2008), indicating that substantial sinks for nitrogen must exist in the landscape. Streams and rivers may themselves be important sinks for nitrogen owing to their hydrological connections with terrestrial systems, high rates of biological activity, and streambed sediment environments that favour microbial denitrification (Mulholland et al. 2008).

A biophysical modelling approach is developed which will serve as a solid basis for the valuation of this service and for scenario assessment studies (§ 2.1). We estimate 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. This study extends the above mentioned approach with an analysis of three scenarios ((§ 2.2):

- The current state situation (scenario REF2005) characterizes the most current level of pressures and is thus considered as a starting point for scenario building. This reference is used for comparative assessments as it represents the most recent situation reported for Europe.
- A Business As Usual reference (scenario BAU 2020) is built as a generic scenario for 2020. It aims at propagating the current trend of anthropogenic pressures but considers the *status-quo* in the mitigation of land based nutrient emissions. It includes changes in population count and distribution, prospects for food production and consumption and associated change in land distribution.
- A scenario supporting a reuse of animal manure in Europe (scenario Manure 2020) intends to improve nutrient supply based on an optimal reuse of organic manure and the adjustment of minimized mineral inputs. It emphasizes the possibility of redistributing the manure locally produced, according to the (crop) demand for both N and P in surrounding areas.

2.1 The GREEN model

GREEN (Geospatial Regression Equation for European Nutrient losses) is a statistical model developed to estimate nitrogen and phosphorus fluxes to surface water in large river basins (Grizzetti, 2006). The model was developed and used in European basins with different climatic and nutrient pressure conditions (Grizzetti et al., 2005) and was successfully applied to the whole Europe (Grizzetti et al., 2008; Bouraoui et al., 2009). The model contains a spatial description of nutrient sources and physical characteristics influencing the nutrient retention. The area of study is 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 (Bouraoui et al., 2009). For each sub-catchment the model considers the input of nutrient diffuse sources and point sources and estimates the nutrient fraction retained during the transport from land to surface water (Basin Retention) and the nutrient fraction retained in the river segment (River Retention). In the case of nitrogen, diffuse sources include mineral fertilizers, manure applications, atmospheric deposition, crop fixation, and scattered

dwellings[1], while point sources consist of industrial and waste water treatment discharges. In the model the nitrogen retention is computed on annual basis and includes both permanent and temporal removal. Diffuse sources are reduced both by the processes occurring in the land (crop uptake, denitrification, and soil storage), and those 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 each sub-catchment i the annual nitrogen load estimated at the sub-catchment outlet (L_i, ton N/yr) is expressed as following:

$$L_i = (DS_i \times [1 - BR_i] + PS_i + U_i) \times (1 - RR_i) \qquad \text{Equation 1}$$

where DS_i (ton N/yr) is the sum of nitrogen diffuse sources, PS_i (ton N/yr) is the sum of nitrogen point sources, U_i (ton N/yr) is the nitrogen load received from upstream sub-catchments, and BR_i and RR_i (fraction, dimensionless) are the estimated nitrogen basin retention and river retention, respectively. In the model, BR_i is estimated as a function of rainfall while RR_i depends on the river length. For more details on model parameterisation and calibration see Grizzetti et al. (2008) and Bouraoui et al. (2009). Although simple in its structure the model GREEN is able to provide spatially distributed estimates of nitrogen river and basin retention at large scale.

For the application of the economic valuation methods proposed in the present study we specifically considered the following model outputs:

1. The nitrogen retained per sub-catchment by river retention (N retained river, ton N/yr):

$$N_{retained\ river} = N_{river\ input} - N_{river\ output}$$

$$N_{retained\ river} = (DS_i \times [1 - BR_i] + PS_i + U_i) \times RR_i \qquad \text{Equation 2}$$

2. The nitrogen retained per sub-catchment by basin retention (N retained basin, ton N/yr)

$$N_{retained\ basin} = N_{basin\ input} - N_{basin\ output}$$

$$N_{retained\ basin} = DS_i \times BR_i \qquad \text{Equation 3}$$

[1] GREEN assumes a constant reduction of 50% for scattered dwelling and the remaining part is then subject to aquatic retention.

We have to consider that the residence time increases in low flow conditions, enhancing the denitrification process. Nitrogen removal decreases in deeper channels as the contact surface between sediment and stream water is reduced and the stream depth generally decreases in low flow conditions. Therefore, the in-stream retention processes were related to the low water flow in the model (Wollheim et al. 2006).

Once we get all these output from model we need to translate them in monetary terms.

2.2 The MANU scenario

2.2.1 Scenario building

The MANU scenario lies on a simple assumption: mineral application of nutrients can be adjusted according to plant demand for both nitrogen and phosphorus, while animal manure application has to be used with a specific N:P ratio, directly related to the type of breeding activities.
The optimisation procedure starts with the calculation for each sub-basin in Europe of:

- The *net demands* of the plant (either grass or crop) for nutrient inputs (both nitrogen and phosphorus), calculated according the crop export, after substituting the inputs corresponding to atmospheric deposition, fixation and crop residues.
- The quantity and quality (N:P ratio) of the animal manure locally produced, according to the intensity of breeding activities.

Next, allocation rules consider that animal manure is applied on grassland area and the remaining part is then used as crop fertilizer, without exceeding the *net demand* of the plant for any of the two nutrient elements. The potential amount of residual manure (not applied at the sub-basin scale) is either cumulated at the basin level, and redistributed to sub-basin candidates according to their residual demand, or redistributed to all sub basins according to their size, and weighted with respect to their initial excess of manure. Finally, if such organic supplies do not cover the *net demand* of plants, adjustments in nitrogen and/or phosphorus “mineral” inputs are calculated for each individual sub-basin.

As it was designed, the MANU promotes the use of nutrient inputs locally produced. It considers the basin scale as a coherent unit to manage manure availability and crop demand for both nitrogen and phosphorus. The first objective of this theoretical exercise is to demonstrate that animal manure can be considered as a resource for agricultural purposes and not as a waste. Whereas during the last

decades, industrial agriculture has led to a strong decoupling between breeding and cropping activities, this scenario demonstrates that there is a possible adjustment between areas requiring fertilizer inputs (manure deficient) and areas having an excess of manure. It has to be acknowledged that in reality animal manure handling over long distances (e.g. across the Danube, the Seine or the Rhine river basins) is not obvious. Especially for slurry that can contains 95% of water and requires some pre-concentration treatments prior to any transportation or storage, in order to be economically sustainable. In addition, animal manure may undergo specific treatments related to odour nuisance, water pollution, prevention of disease etc.. However, this theoretical exercise performed at the basin scale, does not consider transportation of manure as the only option for a better reuse of animal manure. Rather than promoting long distance exportations, the underling idea of the MANU scenario is clearly to rethink current agricultural practices and promote a better balance between cropping and breeding activities at the farm (or local) scale. Moreover, a specific focus is added on the suitability of the manure locally produced and its capacity to cover the demand of surrounding crops for both N and P nutrients.

2.2.2 Scenario results

The total amount of manure applied at the scale of European river basins is preserved across BAU and MANU scenarios, only its distribution at the sub-basin scale is affected. However, the impact on mineral supplies is important especially for nitrogen for which mineral applications are widely exceeding the plant demand. As a consequence, an overall decrease of nitrogen diffuse sources is observed under an optimized manure management scenario (Figure 3).

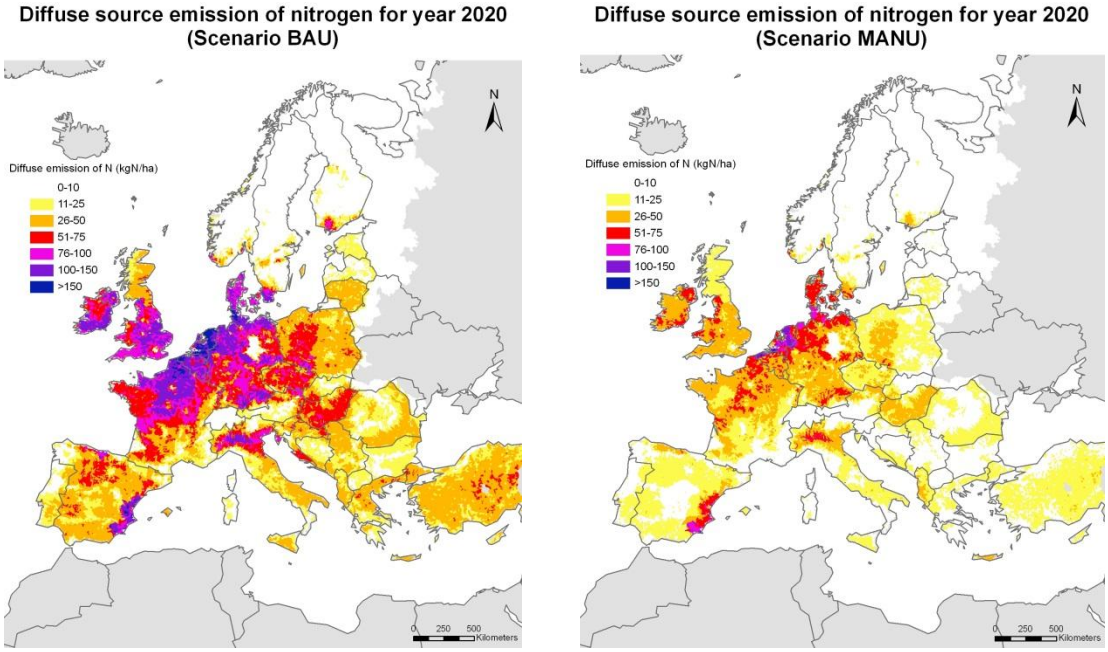


Figure 3. Map of diffuse source of nitrogen (ton N) per sub-basin for BAU and Manure scenarios

Compared to the BAU situation, the Manure scenario could reduce nitrogen exports to European seas –with 36% to 41%, on average, given hydrological conditions. By definition, this scenario preserves agricultural activities, promotes the use of nutrient inputs locally produced, and considers the basin scale as a coherent unit to manage manure availability and crop demand for both nitrogen and phosphorus. Even theoretical, it represent an efficient option to mitigate nitrogen emissions, and probably also a cost-effective one, as it only implies storage and transport costs, and potentially an increase in P-fertilizers application.

2.3 Sustainability indicators

Nitrogen loading is a key pressure to ecosystems. While enhanced N fixation has undeniable societal benefits, N is also a powerful environmental pollutant. The intensification of the N release to the environment has resulted in important and growing effects on human and ecological health (Table 1; Johnson et al. 2010), affecting essential ecosystem services such as the provision of clean air and water, recreation, fisheries, forest products, aesthetics and biodiversity (Compton et al. 2011). Table 1 identifies the trade-offs that arise between different ecosystem services and biodiversity resulting from nitrogen loading and the challenge for policy and river basin management is to find an optimal balance. Figure 4 shows the resulting impact of N loading on various ecosystem services.

Table 1. Ecosystem services and human benefits affected by increasing nitrogen

Ecosystem Service	Impact on benefit	Mechanism of impact
Production of food and materials	+	Increased production and nutritional quality of food crops
	+	Increased production of building materials and fibre for clothing or paper
	-	Stimulation of ozone formation, which in turn can reduce agricultural and wood production
	-	Soil acidification, nutrient imbalances and altered species composition and diversity in forests and other natural ecosystems, which ultimately impact stability and resistance to disease, invasive species and fire
Fuel production	+	Increased use of fossil fuels to improve human health and wellbeing across the globe
	+/-	Increased N inputs required for some biofuel crops can affect other services
Clean air	-	NOx-driven increases in ozone and particulates exacerbate respiratory and cardiac conditions
	-	Increased allergenic pollen production
Drinking water	-	Increased nitrate concentrations lead to blue-baby syndrome, certain cancers
	-	Increased acidification and mobility of heavy metals and aluminum
Swimming	-	Stimulation of harmful algal blooms that release neurotoxins (interaction with phosphorus)
	-	Increased vector-borne diseases such as West Nile virus, malaria and cholera
Fishing	+	Increased fish production and catch for some very N-limited coastal waters
	-	Increased hypoxia and harmful algal blooms in coastal zones, closing fish and shellfish harvests
	-	Reduced number and species of recreational fisheries from acidification and eutrophication
Hiking	-	Altered biodiversity, health and stability of natural ecosystems
Climate regulation	+/-	Variable and system-dependent impacts on net CO_2 exchange
	-	Stimulation of N_2O production, a powerful greenhouse gas
UV regulation	-	Increased N_2O release, which has strong ozone-depleting potential
Visibility	-	Increased NO_x in air stimulates formation of particulates, smog and regional haze
Cultural and spiritual values	-	Altered biodiversity, food webs, habitat and species composition of natural ecosystems
	-	Damage to buildings and structures from acids
	+/-	Long range trans-boundary N transport and associated effects (both negative and positive)

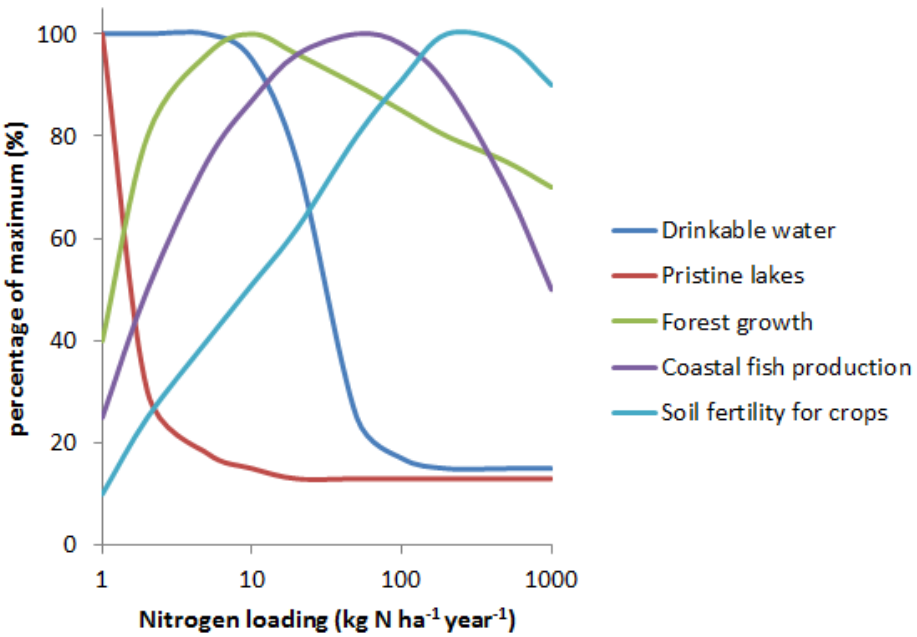


Figure 4. Ecosystem service production functions as function of nitrogen loading

At the ecosystem scale, geology and hydrology interact to control the residence time of water and thus the processing time of nitrogen within an aquatic system. This, in turn, affects the proportion of N inputs that are removed. At the same time, nitrogen loading limits the amount of nitrogen available for removal. With increasing residence time of water in a system, a higher proportion of the available nitrogen can be removed. But also the higher the nitrogen loading, the more nitrogen is removed through denitrification and this is observed across whole range of lakes, rivers, wetlands, estuaries (Seitzinger et al., 2006). Increased nitrogen removal as a result of increased nitrogen loading trades off with several other ecosystem services but the inflection point at which increasing loading results in negative effects varies among ecosystem services (Figure 4).

Ideally, any scenario analysis which investigates the impacts of policy changes on the delivery of ecosystem services should assess multiple services. This was not possible for this study and therefore, we introduce two functions that assume that the benefits of nitrogen loading follow the unimodal response that is suggested in Figure 4.

Firstly, this requires setting a certain criterion for sustainable removal of nitrogen by the river network. As an example, we used a total nitrogen concentration of 1 mg L^{-1} as maximum threshold concentration below which we do not expect harm to the environment. Clearly, this threshold concentration serves as an example for the purpose of this study only and will change depending on the vulnerability of different aquatic ecosystems to nitrogen loading. Sustainable targets for total

nitrogen concentration in freshwater systems can for instance be inspired on the requirements for good or high ecological status required by the Water Framework Directive.

Using data on average river flow (m^3 $year^{-1}$) in combination with the critical nitrogen concentration (1 mg L^{-1}), we can calculate the critical nitrogen loading (L_{crit}, ton $year^{-1}$) which we define as the loading of nitrogen that corresponds to a critical nitrogen concentration below which no harm to the environment is expected. Substituting the nitrogen loading L_i with L_{crit} in equation 1 and solving equation 2 for $N_{retained}$ gives:

$$N_{crit} = L_{crit} \times RR_i \times (1 - RR_i)^{-1} \qquad \text{Equation 4}$$

where N_{crit} is the critical nitrogen removal by the river network assuming a critical loading L_{crit}; RR_i is the river retention coefficient (as a proportion).

Next, we used the critical nitrogen load and the critical nitrogen removal in two functions that assume that nitrogen retention results in maximum benefits for human well-being as long as nitrogen loading is below the critical loading. However, increases in nitrogen loading far above the critical loading will result in costs due to the degradation of most services (Figure 4).Both functions estimate the sustainable removal of nitrogen in ton per year as a function of increasing nitrogen loading.

The second function assumes that at critical nitrogen loading, the removal of nitrogen by the different ecological processes that take place in the ecosystems is sustainable and results in optimal use of the ecosystems from an ecosystem services point of view. This hypothesis allows for nitrogen inputs from anthropogenic sources to an optimal level at which nitrogen concentrations reach a critical threshold. In the subsequent valuation study, nitrogen removal will be valued the most at critical nitrogen loading. Function 2 has the following equation:

$$N_{sust} = N_{crit} \times \exp(-0.5 \times [L_i - L_{crit}]^2 \times [1.5 \times L_{crit}]^{-2}) \qquad \text{Equation 5}$$

where N_{sust} is the sustainable removal of nitrogen, N_{crit} is the critical removal of nitrogen, L_i is the nitrogen loading for river basin I, and L_{crit} is the critical loading of nitrogen.

Function 1 assumes a more conservative approach to sustainability. River networks with zero or very low nitrogen loading are assumed to be sustainable systems and are valued the most but increasing nitrogen concentration leads to decreasing values for sustainable nitrogen removal. Function 1 has the following equation:

$$N_{sust} = N_{crit} \times (1 + L_i / L_{crit}) \qquad \text{Equation 6}$$

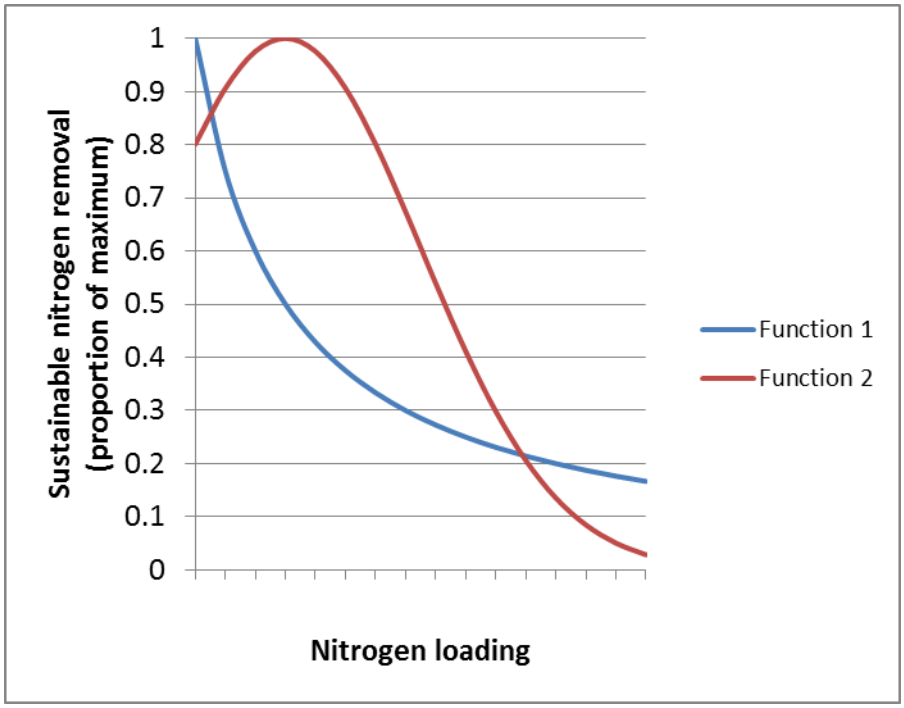


Figure 5. Sustainable removal of nitrogen as a function of nitrogen loading

3. Monetary Valuation

There is a long history of valuing nature and many different approaches have been used in the past. Liu et al. (2010) track the milestones in the history of ecosystem service valuation and Bartelmus (2008) shows the different possible conceptual stages between the two extremes (the pure economist and the pure environmentalist). Despite the critics, there is no economic or monetary estimate of ecosystems or ecosystem services with absolute validity: any valuation exercise is always context related. The aim is not always to attribute an absolute monetary value to nature but monetary valuation of ecosystem services is useful if the purpose of the valuation of ecosystem services and the subsequent application of the results are clearly defined. Here we propose a framework for valuing ecosystem services with the purpose of supporting new EU biodiversity policies aimed at mainstreaming biodiversity policy into EU policies such as agriculture, fisheries, forestry or regional development using the concept of ecosystem services and their monetary valuation.

A vast literature has been developed (for instance de Groot et al. 2002, Turner et al. 2003, Liu et al. 2010, Seppelt et al 2011). All these studies have in common that they present arguments for the practical integration of natural sciences and social sciences, in particular economics. Such integration represents an ambitious research agenda. Before any attempt to value ecosystem services, it is necessary to clarify which are the economic concepts we use, and specifically we need to point out the difference between environmental and ecological economics and, in terms of valuation purposes, the difference between positive and normative analysis

3.1 Background

Environmental economics finds its origin in mainstream neoclassical economics, already at the beginning of the XIX, to correct market failures in the provision and use of environmental goods and services. Its core is the theory of externalities and its aim is the optimal allocation and the efficiency in the use of scarce resources (van den Berg, 2001). An externality refers to the cost or benefit of an activity spill over on a third party, for instance ecosystems. The main focus is the interaction between economic agents, and nature is only implicitly described because the environment is considered as a sub-component of the economy. From a methodological point of view

environmental economics is based on the same concepts and tools on which neoclassical economics are based. Among the main economic concepts we can name individualism, rationality, marginalism, efficiency criterion and general equilibrium models extended to environmental issues. The major advantage of environmental economics lies in its analytical rigor. The advantage of referring to a single discipline with its rules, principles and concepts lies in the inner consistency of results, however in some cases a single discipline may not be able to correctly represent the object of interest. Environmental economics is, in general, precise but lacks realism (Bartelmus, 2008). Eppink and van den Berg (2007) illustrate that economic models applied to biodiversity conservation show a common trend: the overall attention for biodiversity declines when the economic complexity of the model grows. While cost-effectiveness and resource extraction models manage to cover biodiversity at some level, macroeconomic growth and general equilibrium models do not. It would be impossible to include ecological complexity in economic models and still obtain analytical solutions because economic models would loose one of their main characteristics which is analytical tractability.

Ecological economics is a more recent discipline (1980s). The role of the economic system is reversed with respect to the environment: the former is included in the global ecological system, which is characterized by limited resources and much more complex than assumed by environmental economists. Under the ecological economics paradigm, different scientific disciplines need to interact and the final result is not necessarily a monetary valuation of natural capital but other useful units can be used as well, for instance Ecological Footprint, Habitat Equivalency Analysis, Emergy, or health related units such as DALY (disability adjusted life year). The inconsistency between all approaches that are grouped under the umbrella of ecological economics represents its major drawback. Among the possible future developments of ecological economics, two paths are identified by van den Berg (2001). The first one requires a strong and intense co-operation between natural and social scientists to build joint theories and models, and the second one aims at focusing on social sciences to provide an alternative paradigm to the neoclassical methodology (Venkatachalam 2007, Spash 2008).

There are essential differences in the way environmental and ecological economists interpret sustainability: the former aims to keep the natural OR human-made capital intact so that economic growth will not decline; the latter aims to lower the pressure on natural systems, they hold a precautionary principle in dealing with complexity and uncertainty and aim at dematerialising the economy (Bartelmus, 2008). However among these extreme positions (deep environmental versus

deep ecological economists) there is a continuum of intermediate positions where most of the researchers act.

The approach we used is very close to ecological economics of which we support the opportunity to develop a strong co-operation between natural and social scientists in order to build joint theories and models. However, we do not reject (as some ecological economists do) the possibility of using the valuation in monetary terms to convey sustainability messages to policy makers. Although we agree with some of the main critics made on monetizing ecosystem services (Frame and O'Connor 2011, Spanenberg and Settle 2010, Farley 2010) we still believe that monetary valuation represents an effective way to communicate to policy makers. Ecosystem services need to be more integrated into policy decisions and in order to do that they need to be expressed using a common and understandable *language*. Not valuing services too often corresponds with assigning a zero value to services. One of the reasons why ecosystem services are becoming increasingly degraded is the lack of valuation: it is impossible to manage what cannot be valued (Liu et al. 2010). The notion of value on which we base our judgments is the same that characterizes most economic valuation literature. We think that values to be used must be instrumental (instead of intrinsic) and anthropocentric (instead of ecocentric). The utilitarian meaning of value is traditionally identified with human utility and *welfare*. However, we include in the desired objective (to which value is attributed) also the integrity and healthy state of ecosystems and biodiversity in order to assure the *well-being* (not just the welfare) of present and future generations.

The disciplines we start from are in the domain of natural sciences with their particular models. We will thus use specific models[2] that address purposes such as explanation, prediction and decision support.

The role of economics is considered instrumental to natural science: it must consistently translate what natural science assesses in a decision-maker understandable language. From the perspective of our valuation exercise it is not possible to provide an answer to the question 'what are the economic consequences of global biodiversity loss?' because the risk to underestimate and simplify the role of ecosystems is too high; the question becomes rather 'how can we quantify the loss/gain related to a specific ecosystem service in monetary terms?'.

[2] Alternative models are classified by Baumgärtner et al. (2008) as general/abstract models that pursue theory development, testing and generalization.

3.2 Methodology

For the monetary valuation of water purification we decide to adopt a 'cost-based approach' instead of 'damage-based approach' for two reasons. The reason not to use a 'damage-based approach' lies in the difficulty of exhaustively identifying all the benefits that could be lost if the water purification service offered by the ecosystem is not present anymore. These benefits range from the availability of clean water for drinking or swimming, to the presence of fisheries, to the aesthetic perception that influences both recreational activities and real estate markets. The risk of not considering all the benefits derived from water purification by ecosystems would make the valuation incomplete and ambiguous. Moreover benefits overlap in many cases with benefits from other ecosystem services resulting in double counting. Moreover, by using a cost-based approach we can operationalize the underlying concepts of the physical model and monetary values, which is a crucial prerequisite for an integrated valuation. By cleaning (partially) the discharges coming from human activities, aquatic ecosystems provide for free an ecosystem service and thus avoid the degradation of the ecosystem that would deteriorate health and living conditions. If this ecosystem were not there anymore, there should be an artificial replacement which has a cost. Considering the kind of pollution sources (mainly agriculture and livestock activities together with already treated industrial and households discharges) the best proxy we can use as replacement cost are constructed wetlands. Wastewater treatment plants would be inappropriate because they are not applicable to the primary sector (i.e. agriculture and livestock activities) and what is discharged from the secondary sector (i.e. industrial activities) and households has already been treated by these plants. Constructed wetlands (CW) profit from the similar ecosystem functions as aquatic ecosystems do, and their costs refer to ecosystem engineering work which is to our view more objective than using a survey questioning citizens to put a price on nitrogen retention. The rationale is that artificial wetlands are also able to retain nitrogen that is delivered in relatively low concentrations, as opposed to urban wastewater treatment plants that need high concentrations for efficient removal. We thus use the cost of CWs as *proxy* for the valuation of nitrogen retention, which represents, in turn, a *proxy* for water purification. Specifically, the amount of nitrogen that is retained and removed by rivers and lakes will be converted to a CW area equivalent, i.e. the total area (ha) of CW that is needed to result in the same nitrogen retention as the river network in each sub-catchment. Once we have this CW area equivalent, we calculate the costs of the corresponding typology of CWs based on cost data. The typologies of CWs and an assessment of their sizing criteria is provided in § 3.2.1. Secondly, we present the costs for each type of CW and their implications (§ 3.2.2). Finally we apply it to estimate replacement costs for nitrogen retention in river catchments (§ 3.3).

3.2.1 General features and sizing criteria of CWs

The choice of the typology of CWs is made based on the types of pollutant sources that in our case are mainly agriculture and livestock activities (diffuse sources) which require primary treatment and industrial activities and households (point sources) which require secondary and tertiary treatment (since primary treatment is assumed to be performed by Waste Water Treatment Plants-WWTP). For this reason only two categories of CWs will be considered:

- Free Water Surface (FWS) CWs for diffuse pollution control;
- Horizontal subsurface Flow (HF) CWs for point emission sources.

FWS are the best choice for the treatment of nutrients from primary sector activities because of their suitability for watershed scale and because of their ability to deal with intermittent flows and low concentrations (Kadlec and Wallace, 2009).

The choice of HF for point source is driven by the assumption that the nitrogen to be retained is mainly in the oxidized forms (NO_2^-, NO_3^-), due to the general quality characteristics of wastewater treated in conventional WWTPs (which is generally limited during the denitrification phase) and runoff derived from paved and agricultural surfaces. The quality of industrial wastewater is not predictable, but we can suppose that these discharges should be pre-treated to meet the requirements of the EC Nitrates Directive. The HF CWs provide a higher denitrification efficiency than vertical subsurface flow (VF) CWs, due to the prevalent anoxic conditions in the HF beds. The usage of VF CWs for the purposes of this study and its application to modelling is not advised, due to the lack of the need for a strong oxidizing reactor, like a VF CW, generally aimed to enhance the nitrification process, having to face mainly nitrate removal (and so an anoxic or anaerobic environment is more advised for providing the right conditions for the denitrification reactions, i.e. an HF CW reactor).

Although the choice of CWs typologies does not depend on climate features, both HF- and FWS-CWs are able to operate in a range from cold to tropical climatic conditions.

In the following, we briefly present some features of FWS and HF CW. An exhaustive description of the two typologies of CWs, their pollutant removal capabilities, their design and maintenance requirements and their diffusion across Europe can be found in Annex I. What matters for our

valuation purpose is the size of CWs and the way economy-of-scale effects and regional variation in labour and material price influence Construction and Operation and Maintenance (O&M) costs.

FWS-CWs

Generally FWS-CWs are densely vegetated basins that contain open water, floating vegetation and emergent plants. They basically need soil to support the emergent vegetation. The FWS constructed wetlands reproduce closely the processes of natural wetlands, attracting a wide variety of wildlife, namely insects, molluscs, fish, amphibians, reptiles, birds and mammals (Kadlec and Wallace, 2009). The most common application of these systems is the tertiary treatment due to their power of denitrification and pathogens removal (due to the high exposure of the wastewater to the UV component of the sunlight). Sizing of FWSs depends on many parameters. The sizing procedure can be performed using well known and scientifically approved methods like the various first order kinetic equations (Reed et al.,1995, Kadlec et al., 2009) for the pollutants removal. Another simple method is to use design charts (Wallace and Knight, 2006). These charts provide FWS design criteria for BOD, TSS, TKN, TP and FC, which are based on empirical data and the Kadlec equation (Kadlec, 1996). The sizing criteria derived from this method are summarized in Table 2.

Table 2. Recommended FWS Wetland Areal Loading Rates

Parameter	Effluent Goal	Influent Areal Loading Rate	Comments
BOD_5	30 mg/l (90% of the time)	60 kg/ha·d	25 mg/l requires a reduction in loading rate to approximately 30 kg/ha·d.
TSS	30 mg/l (90% of the time)	70 kg/ha·d	25 mg/l requires a reduction in loading rate to approximately 35 kg/ha·d.
TKN	10 mg/l (90% of the time)	15 kg/ha·d	Performance will be dependent on temperature and oxygen transfer.
Fecal Coliforn (FC)	2-log reduction of C_{in}	Min retention time 3 days	/

Generally for tertiary treatment we can use an areal coefficient of 0.5-2 m^2/PE.

HF-CWs

HF constructed wetlands consist of waterproofed beds planted with wetland vegetation (generally common reed species) and are filled with gravel. The wastewater is fed by a simple inlet device and flows slowly in and around the root and rhizomes of the plant and through the porous medium under the surface of the bed in a more or less horizontal path until it reaches the outlet zone. The filling material (coarse gravel, fine gravel and coarse sand) has to offer an appropriate hydraulic conductivity but also a large surface for the biofilm growing. HF beds are typically comprised of inlet feeding system, a clay or synthetic liner, filter media, emergent vegetation, berms, and outlet piping

with water level control. Because the water is not exposed during the treatment process, the risk associated with human exposure to pathogenic organisms is minimized. Properly designed HF beds do not provide suitable habitat for mosquitoes or other vector organism and permit public access in the wetland area. Sizing of horizontal flow subsurface systems depends on many parameters. Again, the sizing procedure can be performed using the various first order kinetic equations commonly used (Reed et al.,1995, Kadlec et al., 1996, 2009) for the pollutants removal and the Darcy law for the hydraulic aspects. As alternative and more simple way it is possible to use “rule of thumb” approaches to the design, based on areal coefficients like “area per PE” or “area per gram of COD”. Some examples across Europe are presented in Table 3.

Table 3. Examples for the design of horizontal flow systems

Parameters	Germany[3]	Austria[4]	Great Britain[5]	Denmark[6]	France[7]	Italy[8]
Surface Area m^2/PE	5 minimum size 20 m^2	5 (secondary treatment) 2 (tertiary treatment)	5 (secondary treatment) 0.5-1 (tertiary treatment)	5 minimum size 25 m^2	5 (BOD 150√300 after a 1° septic tank or Imhoff tank) 2-3 (for BOD 100 mg/l, after vertical filters as1st stage)	4-6
Filling material **Main Layer**	>50 cm	> 50 cm sand: 0/4 (greywater) > 50 cm sand:1/4 (3°) >50 cm sand: 4/8 (2°)	gravel: 3/6 mm or 5/10 mm or 6/12 mm	0.3 mm<d10<2 mm 0.5mm<d60<8 mm	Pea gravel: 4/8 1/4 in second stage vertical filters	80 cm gravel 8/10mm
Permeability of the main layer Kf (m/s)	10^{-4}-10^{-3}	≈ 10^{-4}	≈ 10^{-3}	10^{-3}	≈ $1x10^{-3}$ – $3x10^{-3}$ in operation: $3x10^{-3}$ – $5x10^{-3}$	$5x10^{-3}$
U=d60/d10	< 5			<4		
Hydraulic surface load	<40mm/d	<50mm/d	< 50 mm/d (secondary) < 200 mm/d (tertiary)	/	/	/
Organic load		112 kg/ha*d				/
Depth	0.5 m	/	0.6 m	0.6 m	0.6 m	0.8 m

[3] ATV DVWK A 262 (draft 2004)
[4] ÖNORM B2505 (draft 2003)
[5] Cooper et al., 1996
[6] Brix and Johansen, 2004
[7] Molle et al., 2004
[8] Italian Guidelines on CWs, 2005

3.2.2 Costs of CWs

FWS construction costs

The estimation and the comparison of the construction costs of FWS systems are quite difficult, due to the different levels of complexity and engineering that these systems might provide. The final construction cost is mainly influenced by:

- Presence of waterproofing liner;
- Complexity of inlet and outlet devices;
- Presence of filtration zone with gravel.

Unfortunately, in Europe only few data are available on FWS construction costs: Tsihrintzis et al. (2007) reported that the construction costs for a 5500 m^2 FWS (1200 pe, realized in Greece) was 287,18 €/PE. Consequently we have to refer to a U.S. database to find cost related to FWS construction. Kadlec and Wallace (2009) reported the capital cost for a hypothetical 1 ha FWS CW, that might treat up to 300 m3/d, is 320,000 $/ha (about 23 €/m2). It is seen that over 50% of the direct cost is for earthwork and, and another 28% is associated with planting soil and plants.

Table 4. Estimated capital costs for a hypothetical 1 ha FWS

Component	Unit	Quantity	Unit cost ($)	Total cost ($)
Land acquisition	ha	1	10000	10000
Site evaluation	Lump sum	1	2000	2000
Clear and grub	ha	1	8000	8000
Earthwork	m^3	10000	7	70000
Liner	m^2	12000	8	96000
Planting soil	m^3	3000	10	30000
Plants and planting	plant	20000	3	60000
Structures	Lump sum	5	2000	10000
Conveyance	m	400	35	14000
Site work	Lump sum	1	20000	20000
			Total direct cost	320000

Based on 15 FWS contained in the North American Treatment Wetland Database (NADB), the median cost for an FWS wetland was $ 86,600/ha. When economies of scale are considered, larger systems will cost less per hectare than smaller systems. For example, a 175 ha FWS cost 43,400 $/ha and the cost of a 0.46 ha FWS was 177400$/ha (Wallace and Knight, 2006). Based on these data, we conclude that the construction costs of a FWS varies between 20 and 40 €/m2 depending on the size and complexity of the system.

HF construction costs

Based on literature data, we summarized the construction costs of HF beds the Table 5. The comparison of these prices is quite difficult due to regional differences and inflation.

Table 5. Capital costs for HF constructed wetlands treating mostly municipal or domestic wastewater

Country	Cost per m^2	Cost per PE
Poland	€ 31	€ 121
Belgium	€ 257	€ 1258
Spain		€ 503
Czech Republic	€ 157	€ 631
Portugal	€ 96	
Spain		€ 354
Germany		€ 150-1500
Italy	€ 115	€ 377

The different authors indicate that the most expensive part of investment costs is constituted by the filtration bed and in particular by the media (Tables 6 and 7).

Table 6. Relative costs (%) of the HF CW components

Country	Pretreatment	Filtration bed	Miscellaneous
Austria	28	51	21
Czech Republic	25	60	15
Portugal	20	60	20

Table 7. Capital costs (%) of the HF beds

Country	Excavation	Gravel	Liner	Plants	Plumbing	Control structures	Other
Spain	15	27	33	2	6	5	12
Czech Republic	7	53	13	7	12	/	8
Portugal	12.5	37.5	25	5	11		9

Masotti and Verlicchi (2005) reported a review on Italian construction costs of HF beds designed by Iridra srl (Figures 6 and 7)

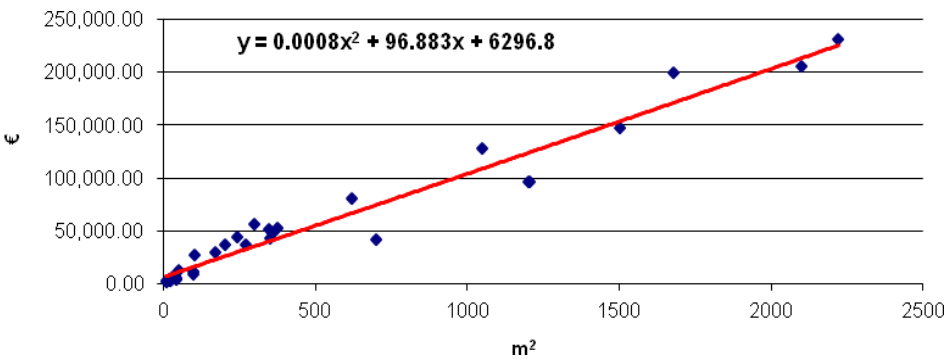


Figure 6. Construction costs of 34 HF beds realized in Italy between 2001 and 2003

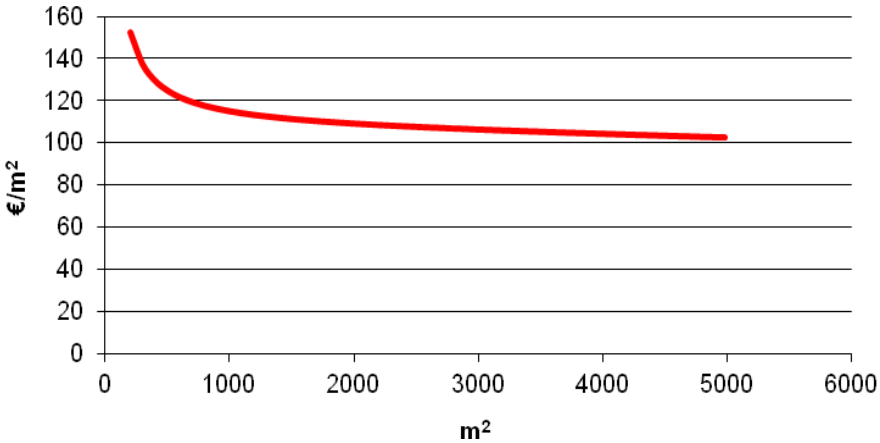


Figure 7. Dependence of specific construction costs on bed size

Vymazal (2008) reported the cost of HF beds designed by the ARM Ltd in UK (Figure 8).

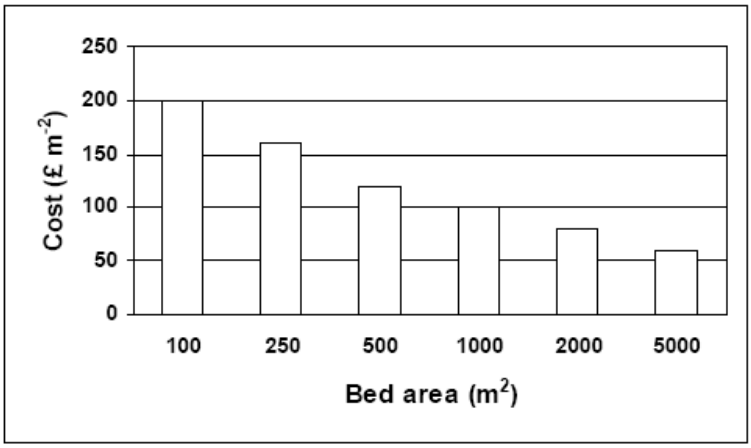


Figure 8. Cost of reed bed built by ARM Ltd (in Vymazal et al., 2008)

In addition to the construction elements of a wetland system, the project will incur other project costs:

- Engineering and permitting: these activities include conceptual design, final sizing, preparation of plans and specifications, preparations of the O&M manual, and permitting. These costs will depend upon the size, complexity and novelty of the project.
- Non construction contractor costs: these typically include mobilization, bonding, insurance, and construction surveying and staking. These costs range from 0,8% to 6,8% of the overall construction costs (Wallace and Knight, 2006).
- Construction observation and start-up services: construction observation, inspections, testing, startup assistance and operator training. These components constitute approximately 10% of the overall construction costs (USEPA, 2000).
- Contingency and escalation: escalation is an allowance for inflation. Contingency is a percentage of the base cost to cover error in human judgment. Contingency allotments of 10-30% are typically used.

Construction costs and economy of scale

A strong determinant in the construction cost of a wetland treatment is the size of the system. Larger projects benefit from economies of scale, resulting in lower unit costs (Figures 9 and 10). The following figures present the trends of costs with size or flow capacity for FWS and HSSF in USA (Kadlec and Wallace, 2009).

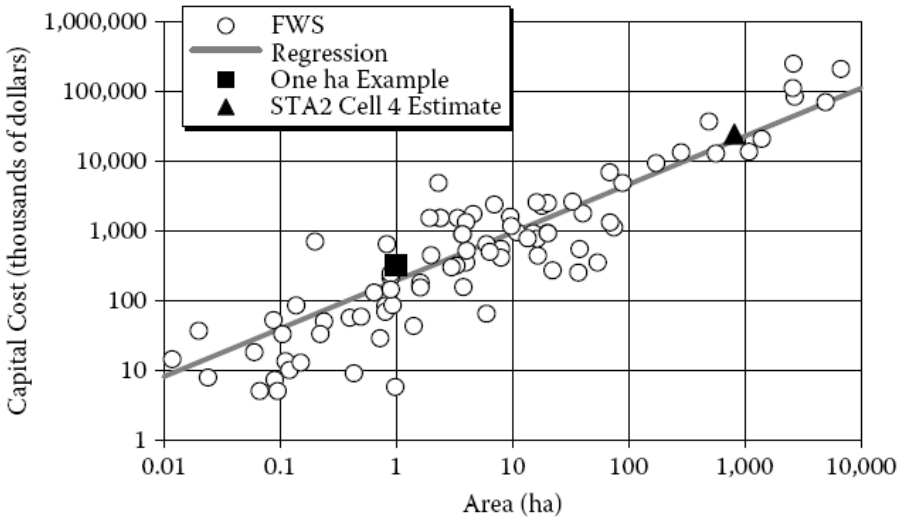


Figure 9. Cost of FWS as a function of size (Kadlec and Wallace, 2009).

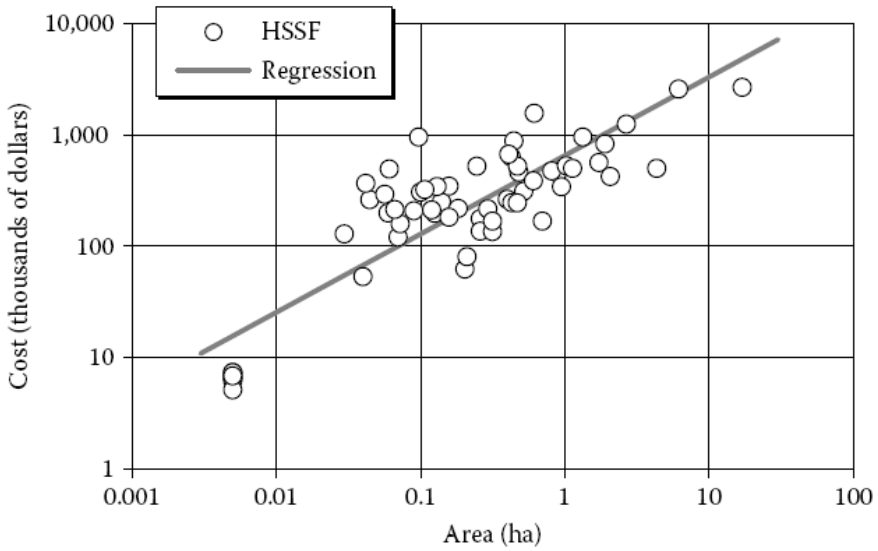


Figure 10. Cost of HF as a function of size (Kadlec and Wallace, 2009).

The above graphs are well represented by the following relationships:

FWS Cost (US $) = 194·A(ha)*$^{0.690}$ R^2=0.79*

0.03<A<10000 Equation 7

HF Cost (US $) = 652·A(ha)*$^{0.704}$ R^2=0.75*

0.005<A<20 Equation 8

Silva and Braga (2006) reported a good correlation between the size of HF constructed wetlands and the investment costs for systems up to about 1000 PE:

Costs (€/PE) = -297 ln PE + 2103 R^2 *= 0.58* Equation 9

Operation and Maintenance costs

Wetland systems have very low intrinsic O&M costs including pumping energy, compliance monitoring, maintenance of access roads and berms, harvesting of the vegetations and mechanical component repair.

The estimates of O&M costs for HF and FWS systems vary widely in the literature, as showed in the following Table 8.

Table 8. O&M costs of HF and FWS CWs.

Country	O&M Costs
	HF
USA	2510-4045 $/ha·y
Austria	300 (<50PE) ÷ 60 (50-500 PE) €/PE·y
Spain	58 €/PE·y
Portugal	2000-6000 €/ha·y
Italy	8 €/PE·y
	FWS
USA (EPA, 2000)	1533 $/ha·y

External experts estimated the O&M costs of a CW treatment plant (Imhoff tank+ 400 m^2 HF) for 100 PE (Table 9).

Table 9. O&M costs of a HF CW for 100 PE.

Operation	*Cost (€)*
Primary sludge disposal	777.60
Macrophytes harvesting	57.04
Grass mowing in plant area	145.03
Wells cleaning and disposal of removed sediments	30.00
Personnel	834.00
Total	1,843.67
Cost per m^2	4.61
Cost per PE	18.44

It is interesting to see the relationship established in Kadlec and Wallace (2009) about O&M costs (Figure 11).

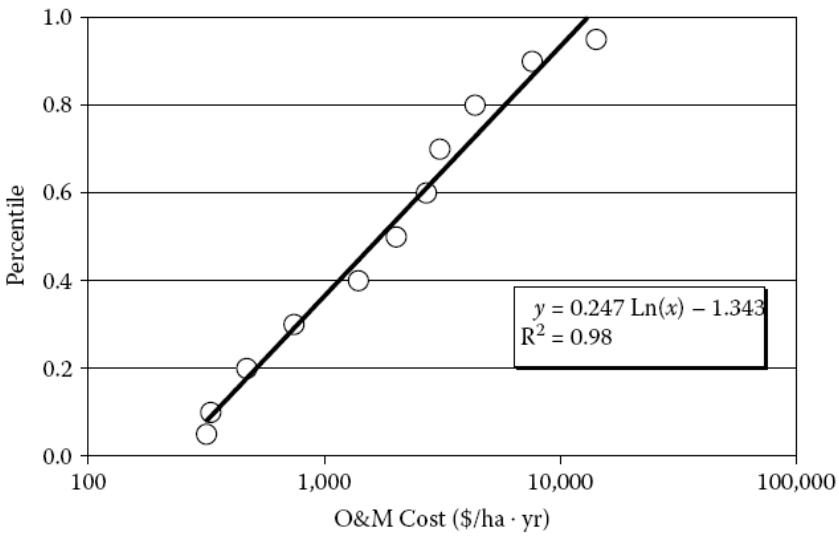


Figure 11. O&M costs for 21 FWS wetlands (Kadlec and Wallace., 2009).

3.3 Application

For the evaluation of the replacements costs, a direct relationship can be defined between the required area (ha) of constructed wetlands for replacing the hypothetical loss of natural nitrogen retention of rivers or lakes. Based on the assumption that oxidized nitrogen is the dominant form in the pollution sources considered in the study (mineral fertilizers, manure, atmospheric deposition, scattered dwellings, secondary effluent of WWTP, industries), the types of CWs that show the higher efficiency in nitrogen removal are HF CWs for the treatment of point-sources and FWS CWs for the treatment of diffuse sources.

The Kadlec and Knight (1996) method has been adopted for the sizing of CWs systems. The nitrogen removal can be described with first-order plug-flow kinetics:

$$\ln\left(\frac{c_e - c^*}{c_i - c^*}\right) = \frac{-k}{q} \qquad q = \frac{365.Q}{A_s}$$

Equation 10

$$A_s = \frac{365.Q}{k}\ln\left(\frac{c_i - c^*}{c_e - c^*}\right)$$

Equation 11

where:
A_s = surface of the CWs, m^2
c_e = outlet concentration, mg/L
c_i = inlet concentration, mg/L
c= background concentrations, that for nitrate can be assumed=0, mg/L*
K= areal constant of first order, m/year; for nitrogen removal k is temperature dependent: $K=K_{20}\cdot\vartheta^{(T-20)}$
q = hydraulic load, m/year
Q = mean flow, m^3/day.

The semi-empiric kinetic constants values are updated basing on the most recent publications (Kadlec and Wallace, 2009) and shown in Table 10.

Table 10. Kadlec Parameters for nitrate removal.

	HF	FWS
K_{20}, m/y	41.8	30.6
θ	1.102	1.102
C* mg/l	0	0

We do not use river flow data to estimate the mean flow Q of equation 11, as this leads to overestimating the surface of CWs that is needed to replace nitrogen retention as ecosystem service. Consequently the total flow has been split in two different flows:

- Diffuse sources flow as product of the surface basin and annual precipitation (supposing a completely impervious basin);
- Point sources flow, converting the point input sources to the river in terms of person equivalent (a person equivalent corresponds to 12 gN/day and 250l/day).

The consequence of this choice is that for any given catchment, the water flow arriving from an upstream catchment is not considered in the calculation of Q.

$$Q_{PS}(m^3/year) = \frac{Point InputSources(ton/year) \cdot 10^6}{12gN/PE \cdot day} \cdot 0.25m^3/day$$

Equation 12

The initial 12gN/day has been updated per country according to Table 11 extracted from Bouraoui et al. (2011).

Table 11. Daily N emission based on the protein intake (g N/day/capita).

COUNTRY	N emission
Albania	10.12
Austria	11.99
Belarus	9.46
Belgium	11.44
Bosnia and Herzegovina	7.59
Bulgaria	9.46
Croatia	7.37
Cyprus	11.33
Czech Republic	9.90
Denmark	11.66
Estonia	9.79
Finland	11.00
France	12.87
Germany	10.45
Greece	13.75
Hungary	10.34
Ireland	12.43
Italy	12.65
Latvia	8.69
Lithuania	11.44
Luxembourg	11.44
Moldova	6.71
Netherlands	11.88
Norway	11.44
Poland	10.89
Portugal	12.98
Romania	11.11
Russia	9.46
Serbia and Montenegro	8.58
Slovakia	8.47
Slovenia	11.55
Spain	12.10
Sweden	11.11
Switzerland	10.23
Turkey	10.67
Ukraine	8.80
United Kingdom	10.89

It's assumed that the nitrogen load removed by HF and FWS should be proportional to the ratio between non-point and point input sources to the basin. Consequently the following calculations are made:

Ci/Ce diffuse sources =

$$\frac{(L_i+(DS_i*(1-BR_i))}{L_i+(DS_i*(1-BR_i))-(\%N_{FWS}*N_{river})}$$ Equation 13

Where:
Li= Load at catchment inlet
DSi= Diffuse sources at catchment
BRi= Basin retention
% N $_{FWS}$ = 1 - Percentage of point sources
N_{river}= Total nitrogen removed by the river

Ci/Ce point sources =

$$\frac{(L_i+PS_i)}{L_i+PS_i-(\%N_{HW}*N_{river})}$$ Equation 14

Where:
Li= Load at catchment inlet
PSi= Point input sources to the river at catchment
% N_{HF} = Percentage of point sources
N_{river}= Total nitrogen removed by the river

With these ratios, it is possible to calculate the requested areas for HF and FWS by the Kadlec and Knight formula (above described).

The building value that we calculate refers to the whole building project. What we need in our valuation is an annual flow, we thus need to calculate it. For the estimation of the annual flow from the total building costs, we can use the standard equation.

$$a = \frac{Y \times i \times (1+i)^N}{(1+i)^{N-1}}$$ Equation 15

Where:
a=yearly amount of building costs
i= discount rate, is the excess of the interest rate over inflation, we assume 3%.
N=life expectancy, we assume 20 years.

We take into account on one hand the economy of scale effect, and on the other hand the fact that different countries in Europe have different costs.. We calculate separately the economy of scale effect (that we will define as 'scale costs') and the price difference effect (that we will define as 'differentiated costs') and subsequently combine them.

In order to take into account economy of scale for the constructions costs we will implement the relationships between surface/construction costs presented by Kadlec and Wallace (2009) with a factor of 0.77 for the conversion $/€.

$FWS\ Cost\ (€) = 0.77 \cdot 194 \cdot A(ha)^{0.690}$ $R^2=0.79$
$0.03<A<10000$ Equation 16
$HF\ Cost\ (€) = 0.77 \cdot 652 \cdot A(ha)^{0.704}$ $R^2=0.75$
$0.005<A<20$ Equation 17

The indirect costs (not including the cost of land acquisition) have been included as standard percentage (30%) of construction costs.

Because wetland systems are constructed using local labour and local materials, it is not possible to offer universal cost estimates that will apply to all European countries. The main components of a wetland (as earthwork, gravel, plants) are produced in regional markets: for instance, the installed cost per cubic meter of gravel is highly dependent on the distance between the gravel pit and the site of wetland construction. Labour costs are also highly variable. To correlate the construction costs to a country, it is necessary estimate the cost of each component within a regional market. The basic cost components of a wetland treatment system include: land, site investigation and system design, earthworks, liners, media, plants, water control structure and piping, site preparation fencing and access road, and finally human use facilities.

We could implement in our calculation for each European country a different cost considering labour and filling. The labour cost values have been extracted from the Eurostat data, which reports the costs from 1997 to 2009. For updating these values to year 2011, an inflation coefficient equal to 2% per year has been chosen. For countries with missing data, we extrapolated based on values of adjacent countries with similar economic conditions. The costs of filling materials have been obtained by a direct survey to CW designers and builders from different European countries and by data available in the international peer-reviewed literature.

In order to calculate the 'differentiated costs', the construction costs has been divided in three components: 1) a fixed value (depending on waterproofing, excavation, plants, concrete elements, piping, etc.); 2) the labour cost and 3) the filling materials costs. Referring to Italian situation, we assume a base cost of 120 €/m^2 for HF and 40 €/m^2 for FWS. The fixed values represent 58% and 50% of the total, respectively, for HF and FWS. For each country the total cost (€/m^2) is finally obtained as sum of fixed costs + labour cost + filling material cost for HF and as sum of fixed costs +

labour cost for FWS. Basing on the values reported in Annex I, an O&M cost equal to 3850 €/ha for FWS and 7700 €/ha for HF is assumed.

As shown in Annex II several combinations of scale and differentiated costs have been tested. Compared to the 'differentiated costs' the 'scale costs' are much lower for the building component (that strongly influences the outcome), and this is due to the scale effect. The extremes of the interpolation line are higher for small sized wetlands and lower for large sized wetlands, producing costs that can be considered out of the market, even in underdeveloped countries. The 'differentiated costs' approach is more conservative and closer to reality: for large surfaces it is feasible to realize more than one small wetland instead of a single large wetland. For the purpose of this valuation we show in the following paragraph the results of a combination that considers a 70-30 break down (70% of the cost is based on an assessment of 'differentiated costs' and 30% of the cost is based on the economy of scale model.

Figure 12 frames the adopted methodology with all the implied formula.

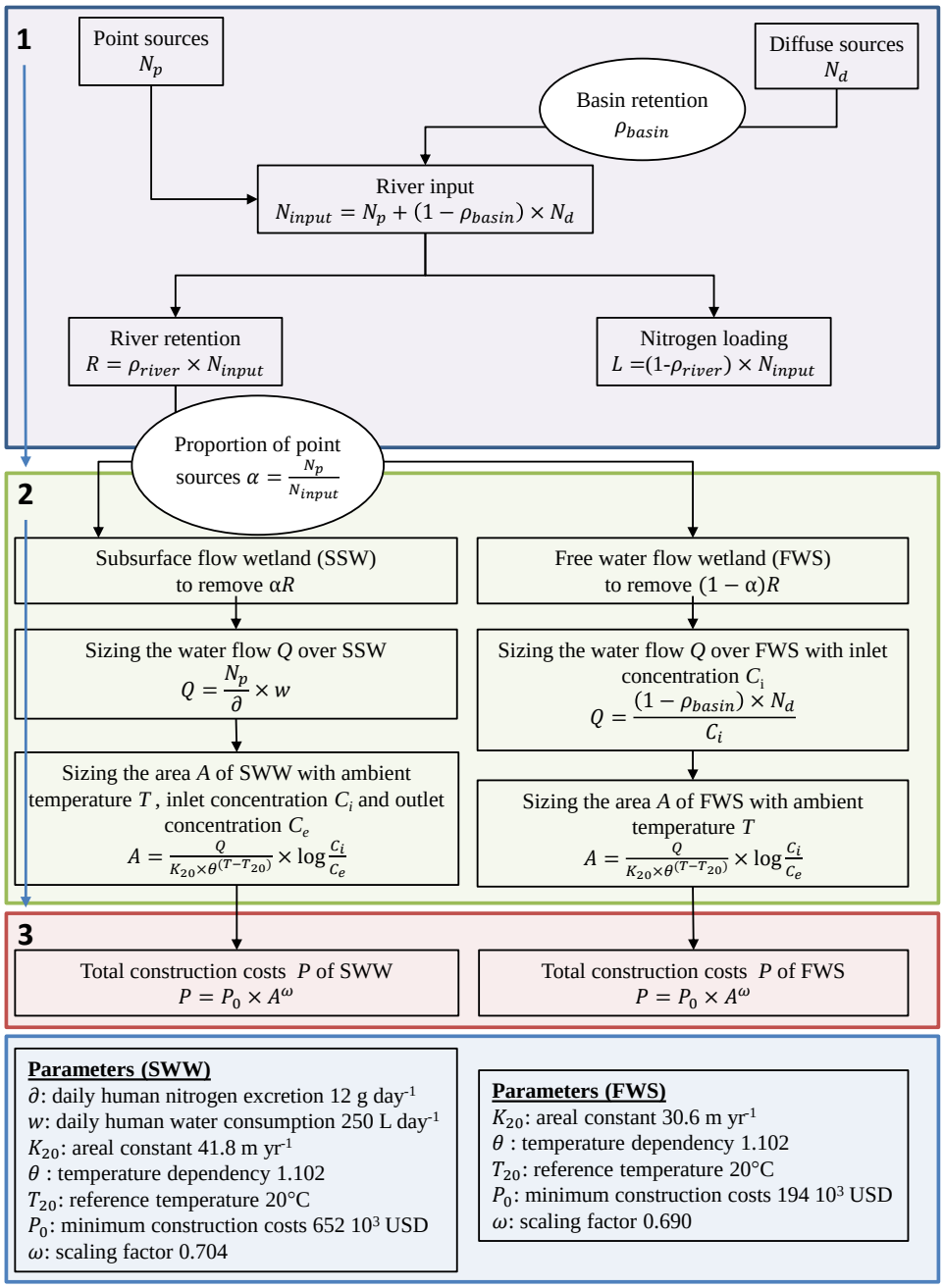


Figure 12. Flow diagram representing the three step approach to assess replacement costs of nitrogen retention in river networks

4. Results

In this section we apply the methodology of monetary valuation of nitrogen retention developed in this report on a biophysical assessment of nitrogen retention in the European river network. In particular, we compare the value of nitrogen retention under two scenarios (MANU2020, BAU2020) with a baseline (REF2005). We do this assessment based on sustainable nitrogen retention (sustainable flow) which considers the capacity of the river network to remove nitrogen and based on effective nitrogen retention (effective flow) which considers actual nitrogen inputs in the environment.

4.1 Scenario outcomes

Table 12 contains the results for the different scenarios in terms of nitrogen emission and nitrogen retention for each considered scenario. Under the MANU2020 scenario, diffuse source emissions decreases with 1 million ton per year whereas under the BAU2020 scenario diffuse emissions further increase relative to the reference. The effective removal of nitrogen by the river network follows this pattern with a reduction under MANU2020 and an increase under BAU2020.

Table 12. Nitrogen retained by rivers in physical terms (1,000 tonnes).

	Diffuse source emissions to river	point source emissions to river	nitrogen removed by the river
Reference 2005	4,018	1,048	1,272
BAU 2020	4,149	1,095	1,326
MANU 2020	3,075	1,095	1,046

We base the monetary valuation of nitrogen retention on the data presented in Table 12. An emission reduction implies a decrease of economic value if we consider effective flow. We could read the reduction with a reverse sign considering it as savings or we should consider the sustainable flow that not only runs in the opposite direction but also by holding a flow amount that is more than double compared to the effective flow.

Table 13. Monetary valuation of the water purification service.

sum 1,000 €	effective flow	sustainable flow
Reference Scenario 2005	24,750,080	50,958,540
Business As usual 2020	24,745,370	50,638,480
Manure Scenario 2020	24,607,310	52,797,660
average €/km		
Reference Scenario 2005	32,902	90,213
Business As usual 2020	32,895	89,622
Manure Scenario 2020	32,744	92,512

When we consider not the results in absolute value, but we in terms of difference between the scenarios (Tables 14 and 15) we can see that:

- the results in physical units (tons) of Table 12 do not turn into same percentage when we consider their monetary translation
- moreover the percentage difference is much lower when we consider the effective flow
- considering the baseline scenarios produce lower differences if we consider the sustainable flow, slightly higher if we consider the effective flow

Table 14.Differences of monetary values between scenarios of the water purification service.

differences Manure-BAU (€)	effective flow	sustainable flow
absolute values	-138,060	2,159,180
Percentage	-0.56%	4.26%
differences Manure-BAU (€/km)		
absolute values	-150	2,890
Percentage	-0.46%	3.22%

Table 15.Differences of monetary values between scenarios of the water purification service

differences Manure-Reference scenario (€)	effective flow	sustainable flow
absolute values	-142,770	1,839,120
Percentage	-0.58%	3.61%
differences Manure-Reference scenario (€/km)		
absolute values	-157.93	2,300
Percentage	-0.48%	2.55%

The numbers only refer to the value of water purification in inland water bodies. They do not consider the role played by terrestrial ecosystems in the basin catchment. If we look at the total nitrogen retention in basins, the river retention represents only 10% of the total retention process. We also need to consider that for water bodies these numbers represent the value estimated only for water purification: other ecosystem services exist and have to be valued in order to obtain a *total value*.

The spatial distribution of those values is shown in Figures 11 and 12. The Figures were made using €/km values and by cutting outlier values in the quantile class classification. Most of the difference values related to effective flow are negative and in many cases very close to zero. On the other hand, when considering the difference values related to sustainable flow not only they are positive but also reach considerable levels. Table 16 shows mean and median adopted for the classification in both cases in order to check the difference in the amount we are referring to.

Moreover, it is not the same considering whether the difference is calculated between the Manure and BAU scenarios (2020) or between the Manure (2020) and Reference scenario (2005), especially from the spatial distribution point of view of the effective flow. In fact, while the former looks more scattered (Figure 13), the latter shows some zones in which positive and negative values concentrates (Figure 14). When a message has to be conveyed to policy makers it is important that this aspect is taken into account.

Table 16. Difference of €/ha/year of scenario options considering mean and median.

	Mean	Median
Manure2020-BAU2020		
Effective flow	-67	0
Sustainable flow	756	180
Manure2020-REF2005		
Effective flow	-135	0
Sustainable flow	894	164

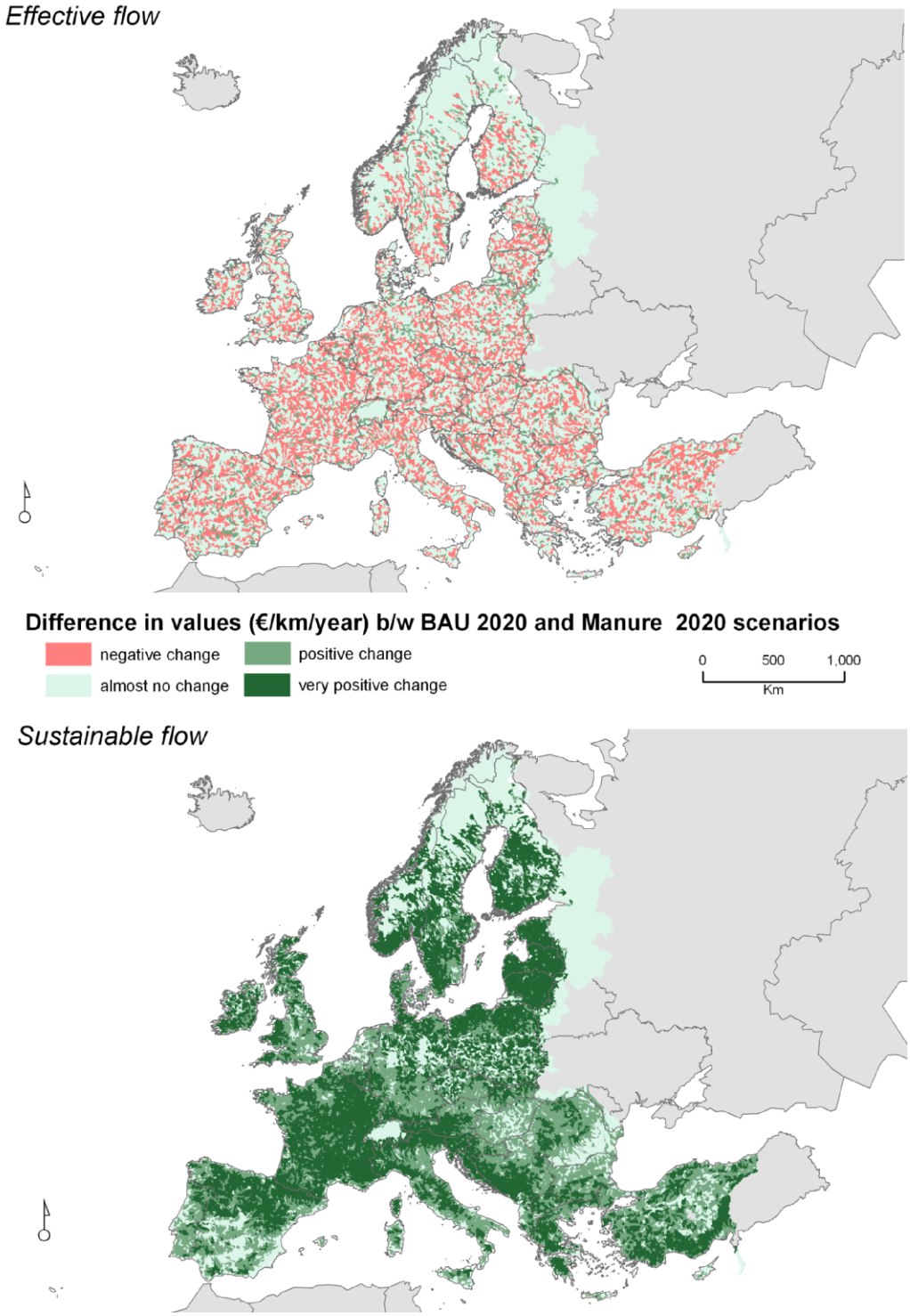
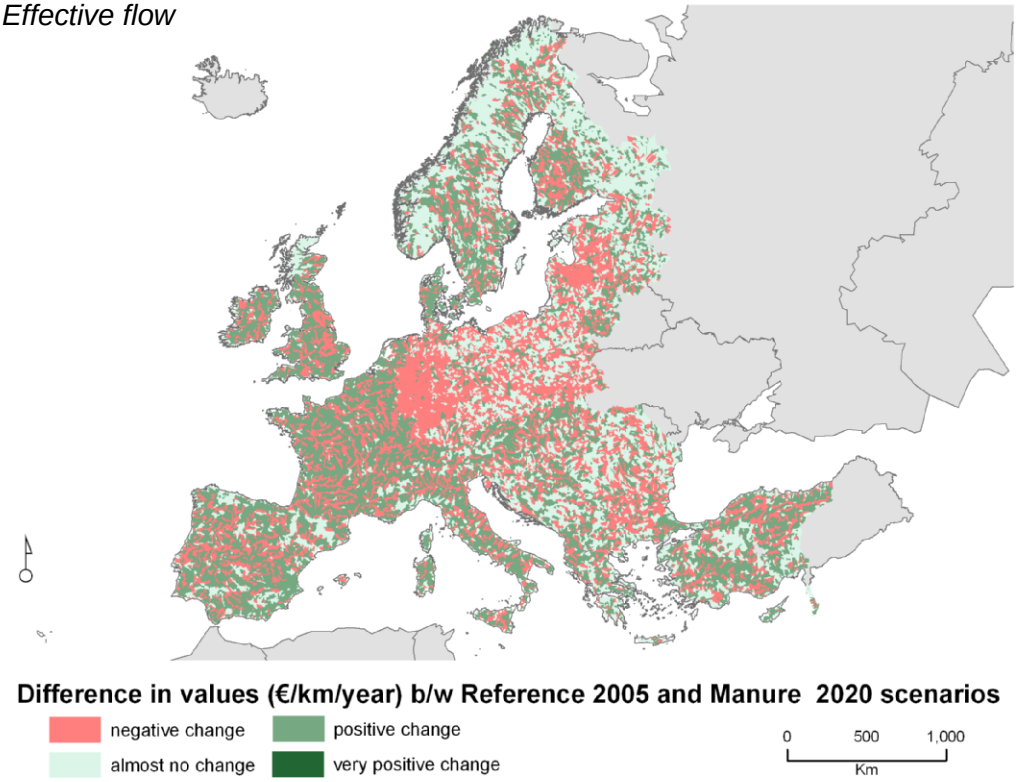


Figure 13. Spatial distribution of value difference between the BAU 2020 and Manure 2020 scenarios

Effective flow



Sustainable flow

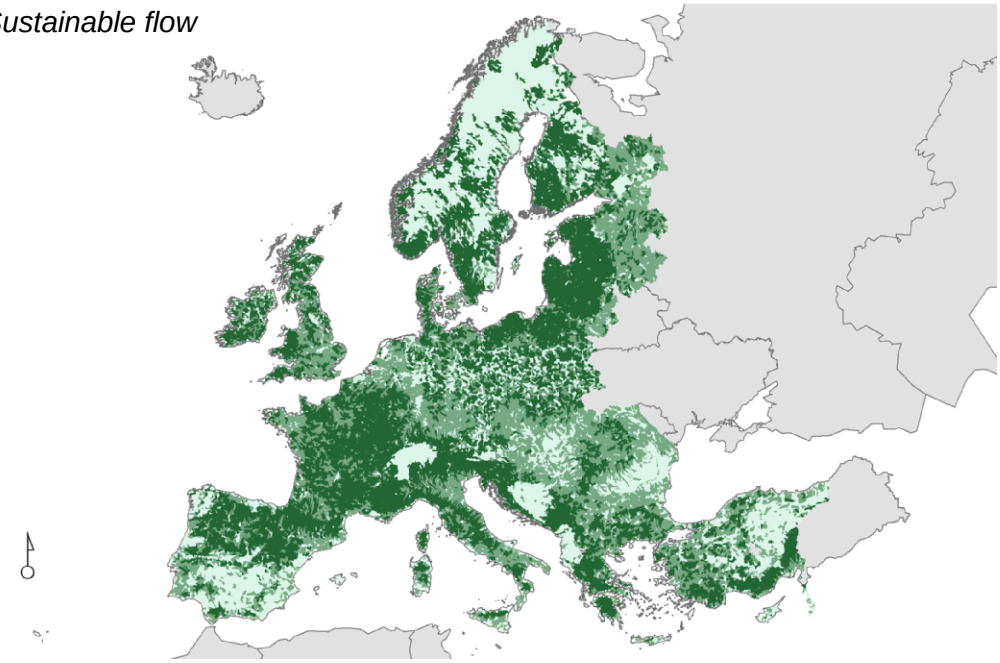


Figure 14. Spatial distribution of value difference between Reference 2005 and Manure 2020 scenarios

4.2 Cost of land acquisition

In the economic estimates of CWs costs (our proxy for water purification) we do not include an important variable: the cost of land acquisition. The reason why this estimate is not considered lies in the difficulties of the availability of an exhaustive database on the cost of land according to its uses for all EU member states. Some reports trace the way to go and provide some references (EEA, 2010) but the coverage is far from being complete and an *ad hoc* study should be undertaken in order to fill this information gap.

However, we aim to have at least a general reference on how higher would be the value of water purification if the estimated values had to include the costs of land acquisition. The procedure considers the EUROSTAT data on the price per hectare of agricultural land. The most recent data refer to year 2006 but are rather patchy across countries. We had to estimate the missing prices. We thus perform a multiple regression based on country surface and population density[9]. For some countries[10] we need to perform additional calculations to have an estimate of population density. Table 17 reports the cost per hectare.

[9] The variables GDP and inequality distribution were also tested but the correlation was not significant.
[10] Namely: Albania, Bosnia and Herzegovina, Kosovo, Montenegro and Serbia.

Table 17. Cost of land per hectare (€/ha), year 2005

Country	*€/ha*
Albania	4022.29
Andorra	9713.83
Austria	6268.12
Belarus	652.77
Belgium	22053.00
Bosnia and Herzegovina	3103.39
Bulgaria	3103.39
Croatia	5038.49
Cyprus	8418.91
Czech Republic	1621.08
Denmark	18787.41
Estonia	1167.89
Finland	5377.00
France	2798.02
Germany	8692.00
Greece	4170.96
Hungary	6861.85
Ireland	16230.00
Italy	12920.33
Latvia	2183.28
Lithuania	536.09
Luxembourg	14874.00
Macedonia	4022.29
Moldova	7608.08
Netherlands	30235.00
Norway	3350.50
Poland	6026.89
Portugal	7425.72
Romania	878.79
Russia	3685.24
Serbia and Montenegro	3103.39
Slovakia	980.60
Slovenia	8640.12
Spain	9713.83
Sweden	3350.50
Switzerland	15168.77
Turkey	1921.65
Ukraine	3685.24
United Kingdom	12974.55

In green: data available from the original Eurostat statistics

The €/ha value is then multiplied by the CW size (ha). Before comparing the valuation with the cost of land acquisition and without the cost of land acquisition we need to address two basic issues.

Firstly, we have to consider that the cost of land acquisition cannot be compared with the flow of the service but must be added to the stock of the construction costs. The whole construction costs (not the annualized values) will be considered. Moreover, Eurostat data refer to agricultural land so we will compare the building costs of (just) FWS with and without the cost of land acquisition. Secondly, in order to obtain significant information, we cannot consider a unique value for agricultural land for the whole Europe. There are considerable gaps among countries and that force us to consider groups of countries with similar geographical and social condition in order to understand how much does land acquisition weight on the whole building costs.

We divide the Europe in five groups:

- Mediterranean countries- Portugal, Spain, Andorra, France, Italy, Greece, Macedonia and Cyprus
- Central and Atlantic countries- Austria, Germany, Belgium, Luxembourg, The Netherlands, Denmark, Switzerland, United Kingdom and Ireland
- Northern countries- Norway, Sweden and Finland
- Eastern countries- Albania, Bosnia and Herzegovina, Bulgaria, Croatia, Hungary, Latvia, Moldova, Poland, Russia, Serbia and Montenegro, Slovenia and Ukraine
- Low land-value countries- Belarus, Czech Republic, Estonia, Lithuania, Romania, Slovakia and Turkey

Table 17 reports the results with and without the cost of land acquisition for the reference scenario 2005. We decided to use the values referring to the economy of scale because decreasing unit costs for increasing extension of property are likely to apply to real estate values. We consider the FWS building cost applied to sustainable flow data.

Table 18. Impact of land acquisition costs in the European regions

		Stock value (€)
Average values for EU-27		
	without land	42,873,994,078.21
	with land	51,849,073,826.26
	% diff.	20.93%
Mediterranean countries	*without land*	2,813,894,717.65
	with land	3,348,429,707.12
	% diff.	19.00%
Northern countries	*without land*	27,720,228,287.49
	with land	33,463,167,677.80
	% diff.	20.72%
Central and Atlantic countries	*without land*	3,391,027,825.88
	with land	4,994,096,008.70
	% diff.	47.27%
Eastern countries	*without land*	6,719,507,910.85
	with land	7,747,178,257.09
	% diff.	15.29%
Lower land value countries	*without land*	2,229,342,052.62
	with land	2,296,208,938.39
	% diff.	3.00%

The range of values to account for, when land acquisition is considered, is very low for some countries (+3%), while it adds almost half of the initial value for some other countries (+47%). The average for the whole EU-27 is about +21%. Ideally, these percentages represent the amount we would need to add to the values we have estimated if we want to take into account the cost of land acquisition. We should thus consider that the building and operation costs of CWs represent only the minimum value to be considered when assessing the value of water purification.

We are aware that the costs of land acquisition obtained represent a rough estimate. There is a need to refine those estimates and conduct a more accurate survey about the cost of land according to the uses in all EU member states. Moreover a reliable relationship should be established between the typology of land and the rents it generates.

Even if Eurostat statistics were complete, the costs of land could not be utilized the way they are because:

- Land in poorer countries records a much lower price irrespective to the environmental quality/ degree of ecosystem integrity

- If we want to include land in the valuation estimates of the service flow, we have to find a proper way to calculate the annual flow related to land
- Both the value of land and the rent it generates must be linked to land use: artificial land cover with all its uses (connected to point sources) must be treated differently from agricultural land from an actual and sustainability point of view.

4.3 Sensitivity Analysis

Sensitivity analysis is used to demonstrate if and to which extent changes to an individual input parameter produce impacts on the model response. In fact by changing input variables, and measuring how the outcomes are affected by that change, it is possible to analyse how sensitive the model is to the individual input variables.

In the calculation of monetary values many estimates and approximations have been undertaken: from the building and O&M costs to the discount rates and the number of years estimated as life expectancy. However, given the conceptual basis of the whole procedure, the major impact on the final result should be produced by the changes involved in the biophysical assessment rather than those depending on pure economic valuation figures. In fact monetary valuation should just translate the outcomes of biophysical assessment. We thus need to find out the parameters that mostly affect the final economic value obtained for water purification. The purpose of the sensitivity analysis here performed is to find out how both biophysical model and economic valuation inputs affect the final output.

Firstly, we identify the variables and parameters, specifically:

1. Diffuse input sources to the river
2. Point input sources to the river
3. River retention potential (%)
4. K_{20} FWS
5. Temperature parameter Θ
6. K_{20} HF
7. Nitrogen load per PE
8. Specific hydraulic load per PE
9. Building cost FWS
10. Building cost coeff FWS

11. Building cost HF
12. Building cost coeff HF
13. O&M cost FWS
14. O&M cost HF
15. Discount rate i
16. Life expectancy N

Secondly, we randomly draw 1000 new sets of parameters from a 10% interval around the mean parameter value to study how the outputs are affected.

Thirdly, we perform a general regression as approximation of model simulation output. Sensitivity coefficients based on the proportion of total variation explained by each factor/parameter were calculated from regression fits according to the equation 18.

$$SC_i = \frac{SS_i}{TSS} * 100 \qquad \text{Equation 18}$$

where:

SCi: sensitivity coefficient as relative sum of squares attributable to factor I (%)

SSi: sum of squares for a regression model with factor i

TSS: total sum of squares of the output variable

Table 19. Sensitivity coefficients for the relevant variables and parameters

	SSi	SCi
Diffuse input sources to the river	85.45	1.02
Point input sources to the river	1399.56	**16.671**
River retention potential (%)	3213.42	**38.277**
K20 FWS	41.46	0.493
Θ	2179.97	**25.966**
K20 HF	7.66	0.091
Nitrogen load per PE	66.35	0.790
Specific hydraulic load per PE	1.82	0.021
Building cost FWS	79.57	0.947
Building cost coeff FWS	550.97	6.562
Building cost HF	15.56	0.185
Buidling cost coeff HF	124.47	1.482
O&M cost FWS	421.09	5.015
O&M cost HF	133.47	1.589
Discount rate i	24.45	0.291
Life expectancy N	49.88	0.594
TSS	8395	

As reported in Table 19, 81% of the model results are explained by variables that depend of the biophysical assessment part. Specifically, 56% depends on the model input (diffuse and point emissions to the river) and parameters (river retention). 27% depends on the physical base of the replacement cost, namely the parameters used to size the area of CW necessary to retain the amount of N (especially Θ requires particular attention). Only 17% depends on the purely economic figures, i.e. building and O&M costs and their coefficients, the discount rate and life expectancy of the CWs.

We demonstrate that the drivers of changes in the final outcome mainly depend on the biophysical assessment and results are thus consistent with the conceptual basis.

5. Discussion

The chapter summarizes the main findings from the previous chapters, and introduces some issues kept in the background that were crucial for delivering the reported results. A series of lessons learned are listed and commented followed by the recognition of weak points of the study together with proposal to overcome the limits and go further.

The first lesson that can be learned from this valuation study is that there is no perfect economic valuation for ecosystem services, each valuation that is made is context specific and answers the needs expressed by the specific study/project/research undertaken. The purpose of economic valuation from our point of view is to include ecosystem services into biodiversity policies by demonstrating the value of ecosystems in order to justify investments in biodiversity protection as desirable objective of COM(2011)244 and COM(2011)571.

It is thus important to be specific about the theoretical background we move from and the purpose and context behind the valuation exercise. If we aim to value ecosystem services to provide policy makers with information on the valuable role of these services for human well-being the notions behind the valuation exercise are referring to concepts such as integrity and healthy state of ecosystems. These concepts are not contemplated if we use economics (in terms of monetary valuation) as the conceptual system that *explains* the ecosystem trend. We should then consider, and this is the second lesson, that different approaches of economic valuation may exist when the purpose is conservation and aims at providing a monetary value to those services that do not pass through the market, instead of purposes that are closer to tangible end user benefits or that involve an assessment of the consequences on the market. The role of economics in this study is *instrumental* to natural science. We do not use the economic logic to say what is scientifically correct about ecosystem functioning. Natural science does it. Economics must consistently translate what natural science assesses in a decision-maker understandable language.

The disciplines we move from are natural sciences with their models and the economic valuation methodology has been though within an interdisciplinary environment. The third lesson is in fact about the necessity of operating in an interdisciplinary environment when approaching issues such as valuation of ecosystem services. Interdisciplinarity can take place in different ways (Baumgärtner et al., 2008): from an independent assessment carried out without sharing concepts, methods or

theory (side-by-side), to a subsequent integrative analysis where a discipline feeds its results into another discipline (division of labour) to a sharing of theory and knowledge between disciplines (fully integrated cooperation). The more the monetary valuation technique is tailored on the meaning of the outcome of the biophysical model the more integrated is the cooperation. The relationship is two-sided: the biophysical model can be extended in order to satisfy the requirements needed for a comprehensive and consistent valuation, e.g. in our case the sustainability function processing.

Specifically about sustainability, the fourth lesson is that ecosystem services are not properly valued if only their effective use is considered. For conservation and protection policies it is important to consider the potential capacity of ecosystems to provide that particular service and thus take into account how healthy the ecosystem is and for how long it will be able to guarantee the flow the service to future generations. Moreover, especially for the regulating services, not considering the potential capacity, and thus the sustainability issue, can lead to a paradox in economic terms: the higher the human pressure on the environment (e.g. more pollution) the higher is the economic value attached to purification services, because only what is used is valued. Such a valuation does not support any policy aimed at ecosystem conservation and protection.

The main weakness of the valuation is the lacking of biodiversity features in the GREEN model. The integration of those features should become a crucial part of the model itself in order to create a stronger linkage with biodiversity issues. Moreover the sustainability functions can be improved, for example by having different threshold limits according to location of each category of catchments (upstream catchments versus downstream catchments).

The economic valuation should include the cost of land acquisition whose estimates at this stage were too rough to be included into the final values. Those costs would play a crucial role when scenarios involved land use change. In trade-off assessment among different ecosystem services, land use change scenarios are going to be one of the main driver of change that is considered, and the land use modelling platform (LUMP) its main tool (Lavalle et al., 2012).

A last step that would make this assessment and valuation even more usable for policy assessment is the framing of outcomes in both physical and monetary terms into a system for ecosystem accounting, with special reference to ecosystem services. The users and uses of this kind of

information for strategic planning are already set and are expanding further. The contribution of these kinds of studies would definitely be useful.

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ANNEX I

Assessment of construction costs of artificial wetlands

1 INTRODUCTION

Rivers, streams and lakes remove dissolved nitrogen from the surface water by plant uptake and denitrification, hereafter collectively called retention. This ecosystem service contributes to maintaining or improving 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. The BIOMES action of the JRC is performing a pan-European monetary valuation of this in-stream nitrogen retention capacity using replacement costs. This valuation is founded on bio-physical maps of retention of nitrogen in rivers based on the model GREEN. The model GREEN contains a spatial description of nutrient sources and physical characteristics influencing nutrient retention. For each sub-catchment the model considers the input of nutrient diffuse sources and point sources and estimates the nutrient fraction retained during the transport from land to surface water (Basin Retention) and the nutrient fraction retained in the river segment (River Retention). In case of nitrogen, diffuse sources include mineral fertilizers, manure applications, atmospheric deposition, crop fixation, and scattered dwellings, while point sources consist of industrial and waste water treatment discharges. Diffuse sources are reduced both by the processes occurring in the land (crop uptake, denitrification, and soil storage), and those 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.

This research concerns only river retention processes that need to be valuated in monetary terms. The assumption is that in absence of in-stream nitrogen retention, replacements costs have to be paid in order to maintain the supply of this ecosystem service. We assume that the construction of artificial wetlands can be used for calculating these replacement costs. The rationale is that artificial wetlands are also able to retain nitrogen that is delivered in relatively low concentrations, as opposed to urban wastewater treatment plants that need high concentrations for efficient removal.

Only two categories of CWs will be rationally considered for the above mentioned purposes:

- Free Water Surface CWs for diffuse pollution control;
- Horizontal subsurface Flow CWs for point emission sources.

This choice derives by the assumption that the nitrogen to be retained is mainly in the oxidized forms (NO2-, NO3-), due to the general quality characteristics of wastewater treated in conventional WWTPs (that generally shows major problem in the denitrification phase) and runoff derived from paved and agricultural surfaces. The quality of industrial wastewater are not predictable, but we can suppose that these discharges should be pretreated to meet the requirements of the EC Nitrates Directive. The HF CWs provides an higher denitrification efficiency than VF CWs, due to the prevalent anoxic conditions in the HF beds. The usage of VF CWs for the purposes of this study and its application to modeling is not advised, due to there is no need for a strong oxidizing reactor, like a VF CW, generally aimed to enhance the nitirification process, having to face mainly nitrate removal (and so an anoxic or anaerobic environment is more advised for providing the right conditions for the denitirification reactions, i.e. an HF CW reactor).

The typology of CWs that will be used will depend only on types of pollutant sources and NOT on climate features: both HF and FWS CWs are able to operate even in cold or torrid climate.
About the costs, the mainly factors that can be varying on regional scale are:

- Land acquisition;
- Labor;
- Filling Media

2. HORIZONTAL SUB-SURFACE FLOW WETLANDS (HF)

General Description

HF constructed wetlands consist of waterproofed beds planted with wetland vegetation (generally common reeds) and filled with gravel. The wastewater is fed by a simple inlet device and flows slowly in and around the root and rhizomes of the plant and through the porous medium under the surface of the bed in a more or less horizontal path until it reaches the outlet zone. The filling material (coarse gravel, fine gravel and coarse sand) has to offer an appropriate hydraulic conductivity but also a large surface for the biofilm growing.
HF beds are typically comprised of inlet feeding system, a clay or synthetic liner, filter media, emergent vegetation, berms, and outlet piping with water level control.

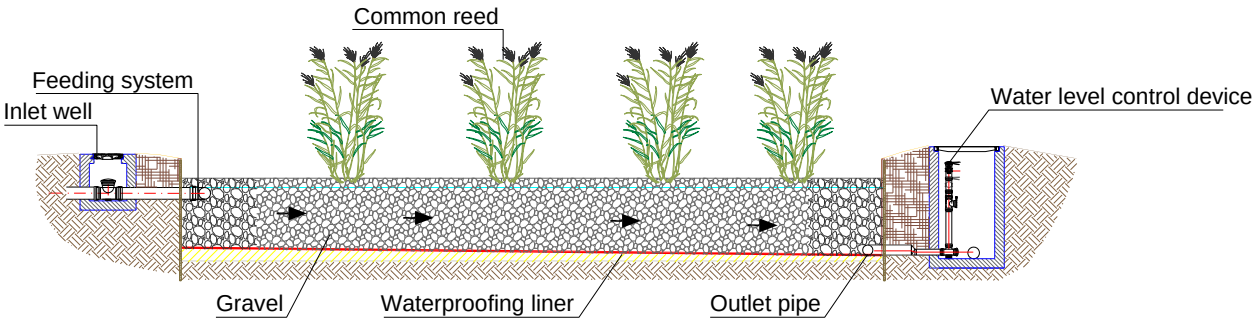

Figure 15. HF wetland schematic

Because the water is not exposed during the treatment process, the risk associated with human exposure to pathogenic organism is minimized. Properly designed HF beds do not provide suitable habitat for mosquitoes or other vector organism and permit public access in wetland area.

These systems are capable of operation under colder conditions than FWS, because of the thermal insulation offers by the filling media and the possibility of adding an insulation layer on the top of the gravel.

Pollutant removal capabilities

HF wetlands are able to remove or transform wastewater pollutants through physical, chemical and biological mechanisms.

Organic compounds are degraded aerobically as well as anaerobically by bacteria attached to the media surface and roots and rhizomes of the plant. However, due to the lack of oxygen, anoxic and anaerobic decomposition play a most important role in HF constructed wetlands. The major removal mechanism for nitrogen in HF CWs are denitrification reactions because, due to its prevalently anoxic conditions, nitrification is limited. Microbial pollution removal is mainly achieved through a combination of physical, chemical and biological factors.

The major removal mechanisms for HF systems are summarized in the following table:

Table 20. Removal mechanisms in HF wetland

Water contaminant	Physical	Chemical	Biologica
Suspended solid	Sedimentation, filtration	/	Microbial degradation
BOD_5	Sedimentation	/	Microbial degradation
COD	Sedimentation	/	Microbial degradation
Metals (Ag, AS, Cd, Cu, Cr, Hg, Ni, PB, Se, Zn)	Sedimentation	Precipitation, adsorption, ion exchange	Microbial Up-take, plant uptake
Hydrocarbons	Volatilization	adsorption	Microbial degradation, plant uptake
Nitrogenous compounds	Sedimentation	adsorption volatilization (ammonia)	Microbial degradation, plant uptake
P	Sedimentation	Precipitation adsorption	Microbial Up-take, plant uptake
Pathogens	Sedimentation, Filtration	/	Die-off, microbial predation, exposure to biocides excreted by wetland plants

The performance of HF systems are influenced by wastewater temperature and the hydraulic retention time (HRT): HRT must be minimal 2-3 days to permit removal performances of organic matter over 60-70%. High temperatures positively influence the natural purification processes.
In the following table the usual performance of correctly designed HF systems are shown:

Table 21. Characteristics removal efficiencies of HF treating domestic wastewater

BOD_5	85-95%
Suspended Solids	70-95%
Total Nitrogen	55-75%
Ammoniacal Nitrogen	50-70%
Phosphorus	50-90%
Pathogen micro-organisms	97-99,999%

Design and layout recommendations

Pretreatments

The goal of a pretreatment system is the removal of settleable and floatable solids to avoid the clogging of the CW bed. The type of pretreatment depends on the dimension of the plant:

- Imhoff tank or three chamber septic tank for single households or small settlement;
- For >200 PE and combined sewer system: coarse and fine screening, grit removal and Imhoff tank.

Waterproofing

Bottom and sidewalls of the filter bed have to be waterproof, if a contamination of the groundwater can be expected. In the case of sealing with natural soil a $K_f<10^{-7}$ m/sec is required and a minimum thickness of 30 cm should be given. Artificial sealing with impermeable layer: The material should be acid-resistant and alkali proof, frost-resistant, roots and rodent resistant, non toxic, easy to carry and move, made of recyclable materials (preferred material: HDPE or LDPE).

Filling Material

The filter media consists in a combinations of various size gravel. Commonly, the first meter of the bed is filled with coarse rock (average 100 mm diameter) and the remaining part with a fine well cleaned gravel (average 5-10 mm diameter). The media depth varies, depending on the used plants from 0,6 m to 0,8 m.

Vegetation

The filter beds are usually planted with the same types of emergent macrophytes as present in the natural wetlands. Most common is '*Phragmites australis*' (reed) but also '*Typha ssp*.' (cattail) and '*Scirpus ssp*.' (bulrush) can be used.

Hydraulic requirements

The bed bottom slope is designed in order to respect Darcy's Law and permits to drain the incoming hydraulic load, maintaining the flows under the surface of the bed. The bottom's slope ranges from 1% to 5%.

Lentgh/width ratio of the bed varies from 2 to 0,5 (depending of inlet hydraulic and organic load): in recent years several designers prefer to use a bigger width (l/w ratio < 1) in order to minimize the clogging risk in the inlet zone.
Maximum daily hydraulic load (HLR) for cross surface unit shouldn't be more than 12 m^3/m^2 per day to avoid overflow phenomena.

Inspection and maintenance requirements

Operation and maintenance routine for the HF systems is easy to do and requires no specialized personnel (in most cases there are no electro-mechanical units).
The main periodic checks are listed below:

- ✓ Wetland vegetation uniform diffusion;
- ✓ Presence of weeds;
- ✓ Presence of vegetal diseases or damages by insects or animals.

The plants density must be kept over 10 plants per square meter; otherwise it is necessary to plant new reeds.

Vegetation management: the first action is to cut the reeds mechanically or manually 3 years after the plant started. The reeds debris must be subsequently removed from the bed. Further vegetation cuttings will be performed every 2 years.

The inlet pipe and the first meters of the filling medium should be checked every 6 months, to verify if clogging by sedimentation of the suspended solids and biomass growing on medium surface has occurred. Lack of good distribution will lead onto problems with sludge deposition, surface flooding and outlet clogging. Consequently it is important keeping the distributor free from clogging and checking that flow distribution is even across the width of the bed.

Reasons for malfunctioning and the adequate trouble shooting can be:

- ✓ Superficial runoff, overload: Enlarge filter area or reduce hydraulic load;
- ✓ Sludge drift from the pre-treatment system: Empty pre-treatment or reconstruct pre-treatment;
- ✓ Clogged inlet pipes: clean pipe system;
- ✓ Consecutively the inlet coarse rock has to be removed and washed for the first 30 cm;
- ✓ Plant disease: If vegetal diseases or damages by insect or animals have happened, an intervention of specialized personnel is required to achieve the right solutions;
- ✓ Presence of weeds: remove them manually or by controlled flooding of the bed.

Types of wastewater treated in HF constructed wetlands

Constructed wetlands have long been used primarily for treatment of municipal or domestic wastewaters. However, at present, constructed wetlands are used for wide variety of pollution, including agricultural (diary, pig farms and fish farms effluent) and industrial wastewaters (textile, tannery, mining waters, food processing industries, distillery and winery), various runoff waters (both agricultural and urban runoff) and landfill leachate.

HF constructed wetlands are commonly used to treat municipal and domestic wastewaters as both secondary and tertiary treatment stages. Especially important is the fact that HF constructed wetlands can successfully treat wastewaters with very low concentrations of organics and nitrogenous compounds. It is well known that conventional treatment systems such as activated sludge cannot treat wastewater with such low concentrations (Vymazal et al., 2008).

HF CWs are been successfully used for CSO (combined sewer overflow) treatment in United Kingdom, as demonstrated by the plants of Stretton on Fosse, Honiley and Lighthorne Heath (Griffin, 2003), that achieve efficiencies removal >70% for BOD and TSS and >40-50% for nitrogenous compounds (NH_4 and oxidized nitrogen).

The use of HF constructed wetlands in Europe

The most common CW treatment systems in Europe are sub-surface flow systems, due to their applicability in urban or peri-urban areas.

In the late 1980s, the first HF beds were built in many European countries. By the 1990s, this technology had become a preferred method for wastewater treatment for small villages and other decentralized wastewater applications. The HF systems are mainly applied in the Mediterranean country, Grain Britain and Czech Republic, opposite the VF systems are preferred in the North-Central Europe countries: Germany, Austria, Denmark, France.

Basing on a recent work of Jan Vymazal (Vymazal et al., 2008), we try to schematize the diffusion of CWs in Europe in the following table:

Table 22. Diffusion of HF CWs in Europe

Austria	1400 (400 HF)
Belgium	107 (2 HF, 10 VF+HF)
Czech Republic	180 HF
Denmark	~ 120 HF
Estonia	14 HF, 10 hybrid system containing HF
France	HF constructed wetlands are mainly used as a part of various hybrid systems
Germany	50000 (comprising private households)
Ireland	140 HF
Italy	300 (comprising private households)
Lithuania	20 HF
Poland	100 (most of them single-stage HF)
Portugal	189 HF
Slovenia	28 HF or hybrid HF/VF
Spain	~50 HF
Sweden	~20 HF
United Kingdom	677 HF for tertiary treatment, 88 HF for secondary treatment and 45 for storm sewage overflow+tertiary

Longevity of a HF system

Two decades of treatment wetland literature have reported that the asset lifetime of horizontal subsurface flow treatment wetlands is highly variable mainly due to clogging problems: several systems resulted completely clogged after only 8-10 years, whereas others continue to properly function after 15-20 years (Knowles et al., 2010). Consequently clogging has influenced the newest design guidelines, that recommended the use of gravel instead of finer materials as sand or soil and the adoption of an adequate width of the HF cells (Cooper, 2010).

In addition to design errors, other factors that can be affect the longevity of HF systems are related to an inadequate maintenance. HF beds require very little maintenance, but in practice this can results frequently in the total absence of inspections or maintenance. Cooper et al. (2005) pointed out that the majors maintenance problems that can determine the failure of the HF systems are: inlet distribution problems, outlet collector problems, sludge deposition and surface flooding. Weed control, sufficient screening of the influent, a thorough maintenance of the inlet distribution system and a correct setting of the outlet level were identified as crucial factors contributing to the performance and the longevity of the beds (Rousseau et al., 2005).

Based on our experience and literature studies, we can conclude that an HF systems is able to operate at least for 15-20 years, if properly designed and maintained.

3 FREE WATER SURFACE WETLANDS (FWS)

General Description

Generally surface flow wetlands are densely vegetated basins that contains open water, floating vegetation and emergent plants. They need of soil or another suitable medium to support the emergent vegetation. When the FW systems are applied for the control of diffusion pollution, they don't need of waterproofing with plastic liner, due to the low risk of groundwater contamination.
The main components of a FW wetland are:

- ✓ An inlet distribution system, followed by an inlet deep zone to allow the removal of heavier sediments;
- ✓ Shallow marsh areas with varying depths (0,4 - 0,6 m) with wetlands vegetation;
- ✓ An outlet deep zone to clarify the final effluent;
- ✓ An outlet device to control the water level.

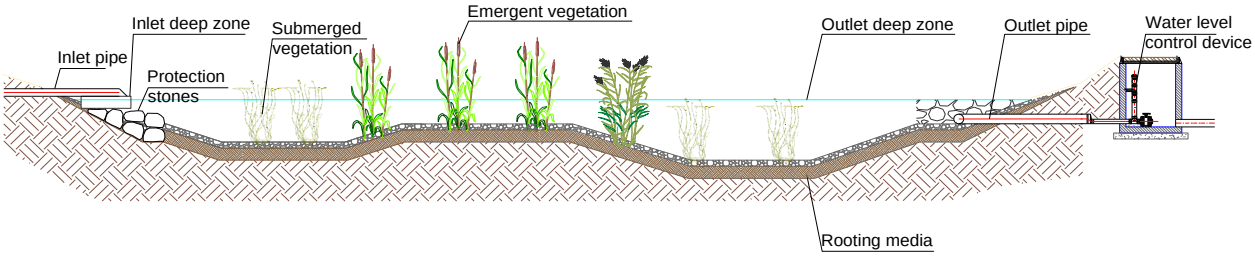

Figure 16. FWS wetland schematic

The FW constructed wetlands reproduce closely the processes of a natural wetlands, attracting a wide variety of wildlife, namely insects, mollusks, fish, amphibians, reptiles, birds and mammals (Kadlec and Wallace, 2009).
The most common application of these systems is the tertiary treatment due to their power of denitrification and pathogens removal (due to the high exposure of the wastewater to the UV component of the sunlight). FW systems are also largely used to control diffuse pollutions: these systems are one of better choice for the treatment of agricultural, urban and industrial stormwater, because of their ability to deal with intermittent flows and low concentrations (Kadlec and Wallace, 2009).

FWS are suitable in all climates, including the cold region. However, ice formation can hydraulically preclude winter operation, and the rates of some removal processes are lower for cold water temperature.

The most commonly used species for FW in European countries are:
- ✓ Emergent plants: *Phragmites Australis* (common reed), *Typha latifolia* (Cattail broadleaf), *Typha angustifolia* (Cattail narrowleaf), *Schoenoplectus lacustris* (bulrush), *Juncus effusus* (Rush);
- ✓ Submerged plants: *Ceratophyllum spp* (coontail), *Potamogeton natans* (Floating Pondweed), *Myriophyllum* (parrot feather), *Elodea canadensis* (waterweed);
- ✓ Floating plants: *Lemna spp* (duckweed), *Hydrocharis morsus-ranae* (frogbit).

Pollutant removal capabilities
FW wetlands, like natural wetlands, remove or transform contaminants in water via many different mechanism. These mechanisms can be classified into physical, chemical or biological processes.
Suspended solids removal is usually a fairly rapid physical process. The major removal mechanisms are sedimentation, aggregation and surface adhesion. Settleable organics are rapidly removed in FWS systems under quiescent conditions by deposition and filtration. Attached and suspended microbial growth is responsible for the removal of soluble organic compounds which are degraded aerobically as well as anaerobically. Oxygen is supplied to the wetland water column by diffusion through the air-water interface and via the photosynthetic activity of plants in the water column. FWS treatment wetlands typically have aerated zones, especially near the water surface because of atmospheric diffusion, and anoxic and anaerobic zones in and near the sediments. Nitrogen is most

effectively removed in FWS constructed wetlands by nitrification/denitrification. Biomass decay provides a carbon source for denitrification, but the same decay competes with nitrification for oxygen supply. Plant-uptake contributes to nitrate and phosphorus removal if the vegetation is periodically harvested. Pathogens are efficiently removed thanks to UV radiations, filtration, predation by zooplankton and natural die-off (due to the adverse condition particularly high and variable temperature, dissolved oxygen, redox and pH).

The major removal mechanisms for FW systems are summarized in the following table:

Table 23. Removal mechanisms in FW wetland

Water contaminant	Physical	Chemical	Biologica
Suspended solid	Sedimentation, filtration	/	Microbial degradation
BOD_5	Sedimentation	UV radiation	Microbial degradation
COD	Sedimentation	UV radiation	Microbial degradation
Metals (Ag, AS, Cd, Cu, Cr, Hg, Ni, PB, Se, Zn)	Sedimentation	Precipitation, adsorption, ion exchange	Microbial Up-take, plant uptake
Hydrocarbons	Volatilization	Adsorption, UV radiation	Microbial degradation, plant uptake
Nitrogenous compounds	Sedimentation	adsorption volatilization (ammonia)	Microbial up-take and degradation, plant uptake
P	Sedimentation	Precipitation adsorption	Microbial Up-take, plant uptake
Pathogens	Sedimentation, filtration	UV radiation	Die-off, microbial predation, exposure to biocides excreted by wetland plants

The performance of FW systems are influenced by wastewater characteristics, temperature and the hydraulic retention time. High temperatures positively influence the natural purification processes and in particular denitrification processes.

Removal efficiency in terms of percentage may not be high all the time due to low inflow concentrations in many systems, but FWS constructed wetlands provide effluents with low concentrations of organics, suspended solids, pathogens. Removal of nitrogen and phosphorus is highly variable but usually amounts to about 50%. Removal of fecal coliforms varies between one or two orders of magnitude depending on retention time.

Design and layout recommendations

Pre-treatments

The inlet zone of a FW wetland can rapidly fill with debris, grit and solids if these materials are not removed prior to the inlet in the wetlands. We can use sediment forebay prior to the inlet for storm water wetlands or coarse screen and grit removal for CSO (combined sewer overflow) treatment.

Rooting media
Emergent wetland vegetation requires a suitable rooting media, that can be constituted by organic soil and gravel with a depth of 20-30 cm.

Hydraulic requirements
The bed bottom slope and the width of the channel are designed in order to respect Manning's equations and maintain flow velocity under 0,6 m/s to prevent erosion. The bottom's slope ranges from 1% to 5%.

Length/width ratio of the bed varies from 2:1 to 10:1; allocation of surface area between deep and shallow zones is 40:60%.

The water level within the FWS is controlled at the downstream end of the wetland through the use of a hydraulic control structures. Common outlet structure utilize adjustable weir gate or corrugated metal riser and barrel.

Inspection and maintenance requirements
Bank vegetation maintenance is required with frequency depending on the used plants.

Wetland facilities should be inspected after major storms during the first year of establishment to assess bank stability, erosion damage, flow channelization, and sediment accumulation within the wetland. For the first 3 years, inspections should be conducted at least twice a year.

A sediment marker should be located in the forebay to determine when sediment removal is required.

Accumulated sediments will gradually decrease wetland storage and performance. The effects of sediment deposition can be mitigated by the removal of the sediments.

Table 24. Typical Maintenance Activities for FW (Georgia Storm water Manual)

Activity	Schedule
• Replace wetland vegetation to maintain at least 50% surface area coverage in wetland plants after the second growing season.	One-Time Activity
• Clean and remove debris from inlet and outlet structures. • Mow side slopes.	Frequently (3 to 4 times/year)
• Monitor wetland vegetation and perform replacement planting as necessary.	Semi-annual Inspection (first 3 years)
• Examine stability of the original depth zones and microtopographical features. • Inspect for invasive vegetation, and remove where possible. • Inspect for damage to the embankment and inlet/outlet structures. Repair as necessary. • Note signs of hydrocarbon build-up, and remove appropriately. • Monitor for sediment accumulation in the facility and forebay. • Examine to ensure that inlet and outlet devices are free of debris and operational.	Annual Inspection
• Repair undercut or eroded areas.	As Needed
• Harvest wetland plants that have been "choked out" by sediment build-up.	Annually
• Removal of sediment from the forebay.	5 to 7 years or after 50% of the total forebay capacity has been lost
• Monitor sediment accumulations, and remove sediment when the pool volume has become reduced significantly, plants are "choked" with sediment, or the wetland becomes eutrophic.	10 to 20 years or after 25% of the wetland volume has been lost

Types of wastewater treated in FWS and their diffusion in Europe

As described above, the FWS in Europe are mainly used as tertiary treatment of activated sludge plant or as final stage (polishing stage) in CW hybrid systems.

The main examples in Italy are the CWs for tertiary treatment in Italy are:

- The post-treatment of the municipal centralized treatment plant (a classic activated sludge technology) receiving the wastewater produced by the municipality of Jesi (Ancona): the tertiary treatment consists of 1 ha of HF stage followed by 5 ha of FWS and provides to complete denitrification and disinfection (Masi, 2008);
- The Fusina FW wetlands (100 ha) (Cattaneo et al., 2010, Frank et al., 2010) to polish industrial wastewater, domestic wastewater and storm water coming from the Mestre-Fusina-Marghera area; the treated wastewater will be reused for industrial application, replacing the current supply from Sile river.

Other types of applications are:

- Agricultural wastewater: an example is the integrated constructed wetland (HF+FWS) for farm yards wastes treatment in Anne Valley, Ireland (Carroll et al., 2005);
- CSO treatment (Buts et al., 2005, Balbo et al., 2010);
- Storm water runoff: agricultural runoff (Borin et al., 2001, Borin et al., 2007, Higgins et al., 1993) e urban (Higgins et al., 2000, Pontier et al., 2004);
- Mine drainage with several application also in Europe (Vymazal, 2009).

In contrast to North America and Australia, FWS constructed wetland technology did not spread rapidly throughout Europe (mainly due to the minor land available) and the main technology focus

has been on subsurface flow systems. However, FWS constructed wetlands are in operation in many European countries (Norway, Sweden, Denmark, Poland Estonia, and Belgium). In Sweden FWS systems have been constructed with nitrogen removal as a primary goal but other aims, such a biodiversity and irrigation, are also taken into consideration. More than 2350 ha of wetlands have been created in Sweden in the agricultural landscape between 1996 and 2002; in Denmark about 3200 have created prior to 2004 (Vymazal, 2006).

Longevity of a FWS system
A treatment wetland has a longer life expectancy than concrete and steel equipment

The major factor that influences the lifetime of a FWS is the solids deposition:

- the in-let zone can become filled and no longer provide the needed vertical-settling depth;
- solids accrete internally to the vegetated zones, because of internal wetland generation of solids, reducing the hydraulic retention time and consequently the efficiency of the system.

It is quite difficult to determine the time required to fill the basin to a certain depth, due to the uncertainty on in-let composition, the efficiency of preliminary treatment, the not uniform spatial deposition. In case of complete filling, the wetland must be shut down and the excess material excavated and disposed. However, we can estimate that the medium life of a FWS will be at least 20-30 years.

ANNEX II

Table 25. Reference baseline scenario 2005 calculation (€/year)

absolute values		effective costs	sustainability function 1	sustainability function2
	economy of scale	6,924,529,342.30	10,022,889,694.60	10,913,815,046.17
	50scale50diff	19,657,067,576.32	38,788,270,134.93	39,517,190,289.86
	30scale70diff	24,750,082,869.93	50,294,422,311.06	50,958,540,387.33
	10scale90diff	29,843,098,163.54	61,800,574,487.19	62,399,890,484.80
	differentiated costs	32,389,605,810.34	67,553,619,138.84	68,120,565,533.54
€/km				
	economy of scale	10,421.54	19,801.04	20,597.55
	50scale50diff	26,479.13	69,775.60	70,322.64
	30scale70diff	32,902.17	89,765.42	90,212.68
	10scale90diff	39,325.20	109,755.24	110,102.72
	differentiated costs	42,536.72	119,750.11	120,047.74

Table 26. Business As Usual scenario 2020 calculation (€/year)

absolute values		effective costs	sustainability function 1	sustainability function2
	economy of scale	6,925,438,249.55	9,937,500,439.28	10,848,713,650.76
	50scale50diff	19,653,962,644.12	38,518,034,542.69	39,269,976,481.80
	30scale70diff	24,745,372,401.95	49,950,248,184.06	50,638,481,614.21
	10scale90diff	29,836,782,159.78	61,382,461,825.42	62,006,986,746.63
	differentiated costs	32,382,487,038.70	67,098,568,646.11	67,691,239,312.83
€/km				
	economy of scale	10,423.44	19,524.71	20,510.33
	50scale50diff	26,474.25	69,149.05	69,876.15
	30scale70diff	32,894.58	88,998.78	89,622.48
	10scale90diff	39,314.90	108,848.52	109,368.81
	differentiated costs	42,525.07	118,773.38	119,241.98

Table 27. Manure scenario 2020 calculation (€/year)

absolute values	effective costs	sustainability function 1	sustainability function2
economy of scale	6,888,725,624.77	10,434,878,108.26	11,502,277,502.83
50scale50diff	19,544,858,047.39	40,124,733,063.98	40,998,979,309.18
30scale70diff	24,607,311,016.43	52,000,675,046.26	52,797,660,031.72
10scale90diff	29,669,763,985.48	63,876,617,028.54	64,596,340,754.26
differentiated costs	32,200,990,470.01	69,814,588,019.69	70,495,681,115.53
€/km			
economy of scale	10,378.91	20,255.01	21,978.62
50scale50diff	26,354.14	71,006.46	72,359.75
30scale70diff	32,744.24	91,307.04	92,512.20
10scale90diff	39,134.33	111,607.61	112,664.65
differentiated costs	42,329.38	121,757.90	122,740.88

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Abstract

There is a need to prove the relevance of ecosystem services in economic terms in order to make a comprehensive and compelling case for conservation of biodiversity. Many different approaches and frameworks used so far have proven that there is no economic or monetary estimate of ecosystems or ecosystem services with absolute validity: any valuation exercise is always context related. This report presents a framework for valuing ecosystem services when the specific purpose of valuation is to support conservation policies at regional scale. After a brief review of the foundation of environmental and ecological economics and after showing the difference between economic models and valuation models, the framework for the valuation of ecosystem services related to conservation policies is presented both theoretically and practically through a case study. Theoretically it is shown how the role of economics (i.e. the application of monetary valuation techniques) is instrumental to natural sciences (i.e. the outcomes of models). The case study refers to the valuation of water purification in the Northern Mediterranean region. Throughout the paper, it is strongly outlined the crucial role of working within an interdisciplinary team

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