

A guiding framework for ecosystem services monetization in ecological-economic modeling

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Abstract

Monetary valuation techniques are often used for evaluating the effect of a change in ecosystem services on components of human wellbeing, even though they face several drawbacks. This paper seeks to reconcile monetary valuation techniques with methods that address ecosystem-economy interactions by developing a guiding framework that limits the use of monetary valuation to various market simulations. Simulations of scenarios of environmental measures are carried out with a semi-dynamic hybrid input-output model. The guiding framework ensures that monetary valuation techniques contribute to the understanding of the impact of economic activities on changes in ecosystems services and the feedback impact of these changes on economic activities. The framework operates according to three criteria: *i*) the category of ecosystem components (intermediate products, ecosystem services, benefits obtained from the ecosystem), *ii*) existence of a market, intention to exchange or possibility for restoration or preservation, and *iii*) direct/indirect monetary valuation techniques. The methodology is then tested with a case-study

Keywords: ecosystem services, input-output model, dynamic model, monetary valuation, ecological-economic interactions, monetization limits.

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1. Introduction

The ecosystem services paradigm¹ favors a better apprehension of interactions between the functioning of parts of ecosystems and components of human wellbeing such as leisure time, health, education, income, purchasing power, etc. (Fisher et al., 2009 ; Millennium Ecosystem Assessment, 2005; Carpenter et al., 2006; Sachs and Reid, 2006). It focuses on preserving the ecosystem as a whole rather than on managing specific natural resources and uses. As a result, it provides a policy shift from previous resource- and species-centered visions of environmental preservation towards a new environmental policy vision based on the preservation of ecological functions and ecosystem services.

Monetary valuation techniques are often used for evaluating the effect of a change in ecosystem services on components of human wellbeing as they are a way to guide trade-offs in decision-making processes (Wincler, 2006). Many papers deal with the difficulty of valuating ecosystem services (e.g. Costanza et al., 1997; de Groot et al., 2002) and the complexity to apprehend interactions between ecological functionalities and the production of ecosystem services used by humans (Daily *et al.*, 2009; Polasky *et al.*, 2011). Other authors claim that monetary techniques may not correctly assess interactions between intermediate products provided by the ecosystem (biological structures or processes and ecological functions) and final products (ecosystem services and benefits) (Ackerman, 2004; Venkatachalam, 2007).

The classical methods for valuating non marketed goods are the direct and indirect valuation approaches (Smith et al., 1986). Contingent valuation is probably the best-known direct² valuation technique. Even though every precaution is taken in building up the questionnaire (Carson and Hanemann, 2005; Spash et al., 2009), some authors argue that most individuals would have problems weighing up complex or unfamiliar environmental issues with global

¹ See definitions of the concept of ecosystem service in Millennium Ecosystem Assessment (2005), Costanza *et al.* (1997), Daily (1997), Boyd and Banzhaf (2007) and Fisher *et al.* (2009).

² Stated preference methods (contingent valuation, choice models) are named “direct approaches” because they consist in directly interviewing individuals and ask them the amount they would be willing to pay to restore one more unit of ecosystem service (e.g. create one more hectare of forest, increase marine fish population by one thousand individuals, etc.).

effects occurring over a long period of time and/or large geographical scales (Markandya *et al.*, 2005; O'Connor, 2000; Ashford, 1981). That can partly explain the price differential between environmental intention and action (Rowlands *et al.*, 2003). Indirect³ methods, like hedonic pricing and travel cost, rely on observed behavior in related markets for valuing the ecosystems services.

Another valuation technique is that of benefit transfer (e.g. Plummer, 2009). If collecting primary data on the ecosystem service under consideration is either too expensive or too difficult, it is possible to transfer an existing valuation from an ecosystem (the study site) to a similar site in another location (the policy site). The procedure is to describe the policy site and the possible policy actions, to select existing studies providing a basis for a benefit transfer, to estimate a value for the relevant site and to apply it to the policy site or alternatively to draw up a benefit function relating an individual's willingness-to-pay (WTP) to a set of individual and site characteristics.

Whatever the method used, no one is exempt from criticism and even though a monetary valuation, albeit imperfect, has the advantage to bring about precious indications about ecosystem services, it can never be used as the sole decision making criterion as other social and ecological objectives (many of which may not be adequately captured by money metrics) must be considered as well.

That being said, although monetary valuation of ecosystem services suffers from several limits, this paper proposes a guiding framework for integrating monetary values into a larger approach based on the study of interactions between the ecosystem and the part of human wellbeing that depends on the economy. Carbone and Smith (2013) model the effect of air pollution on ecosystem services and health in a general equilibrium setting accounting for

³ Revealed preference methods such as hedonic pricing are named "indirect approaches" because they estimate the willingness to pay without asking directly to people the amount they would pay for changes caused to ecosystem services. They utilize the fact that some market goods are in fact bundles of characteristics, some of which are ecosystem services (Pearce *et al.*, 2006). By trading these market goods (e.g. houses in a neighborhood), consumers are thereby able to express their values for the ecosystem service (e.g. clean lake in the neighborhood), and these values can be uncovered through the use of statistical techniques used to estimate the price difference between houses located close to the clean lake and those located next to a polluted lake.

both use and non use services. They compute WTP measures of an ecosystem service and show that general equilibrium effects matter. This paper uses a different setting – an Input-Output (I-O) model – and limits the use of monetary valuation to “real⁴” market simulations. Those market simulations, in which monetary values are inserted, are carried out inside a hybrid I-O model (Daly, 1968; Isard, 1968, Miller and Blair, 2009) that focuses on crossed interactions between components of the ecosystem and the economy and that has been semi-dynamized.

The guiding framework ensures that monetary valuation techniques contribute to the understanding of the impact of economic activities on changes in ecosystems services and the feedback impact of these changes on economic activities. It operates according to three criteria: *i*) the category of ecosystem components (intermediate products, ecosystem services, benefits obtained from the ecosystem), *ii*) existence of a market, intention to exchange, or possibility of restoration or preservation, and *iii*) direct/indirect monetary valuation techniques (see Section 4). One advantage of this guiding framework is to consider the critical importance of intermediate products provided by the ecosystem, even if they cannot be easily monetized, as they condition the existence of all other ecosystem services that benefit human life and economic activities.

The remainder of the paper is organized as follows. Section 2 categorizes ecosystem services into biological structures and processes, ecological functions, ecosystem services and benefits. It explains why our approach limits the use of monetary valuation techniques to benefits produced by ecosystem services (except when environmental restoration or preservation activities are considered, in that case, intermediate products and ecosystem services are also monetized). Section 3 presents the semi-dynamic hybrid I-O model. Section 4 describes the guiding framework for the integration of monetization into the I-O model while the last section is devoted to discussion and the conclusion.

⁴ Real market simulations are considered later in the text but stated preference methods and constructed markets are also used in this paper and they obviously do not perfectly represent real market behavior. However, they can be considered as “potential market simulations” since we applied them to ecosystem services that could potentially be exchanged on a market (via a tax representing agents’ WTP for example).

2. Monetary valuation techniques and their limits for a full assessment of ecosystem services

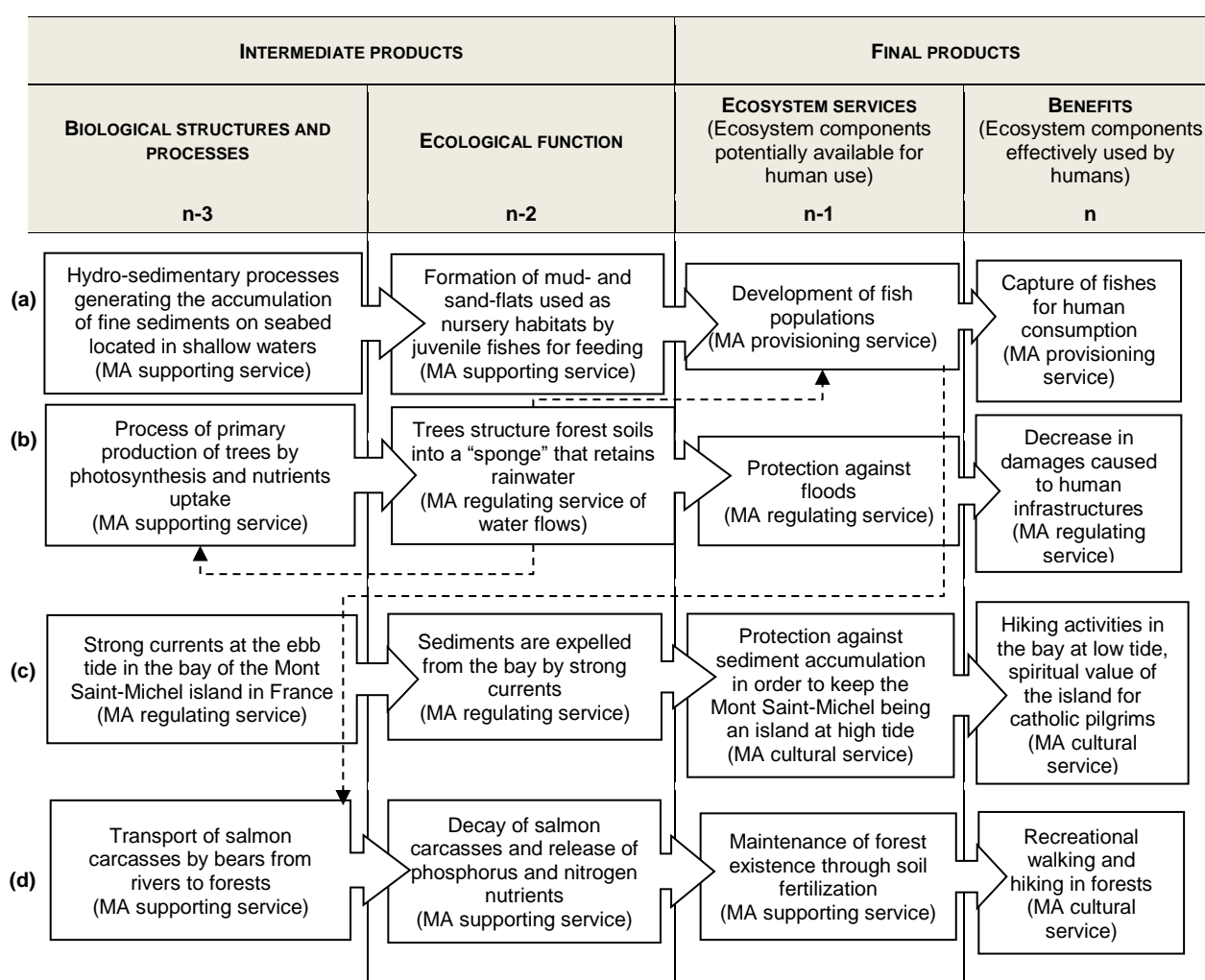
One possible approach to measure the impact of changes in ecosystem services on components of human wellbeing is, after specifying spatial and temporal boundaries, to express their value in physical terms and then convert them into monetary units. De Groot et al. (2002) built a table that offers a good summary to match monetary valuation techniques with the proper category of ecosystem services. But it might also be somewhat misleading, as it shows that the diverse techniques are capable of valuing *all* categories of ecosystem services. This appears to be in contradiction with other scientific contributions (Turner *et al.*, 2004 ; Fisher *et al.*, 2009). These contributions suggest that among the four main categories of ecosystem services defined in the MA (Millennium Ecosystem Assessment, 2005) – supporting, regulating, provisioning and cultural services – the role played by the first two categories inside the ecosystem is not covered by monetary valuation techniques, as they are assumed to be independent of individual preferences. However, monetary valuation techniques are still often used to measure the economic value of supporting and regulating ecosystem services (e.g. the technique of replacement costs for the ecosystem service of water flows regulation) and some researchers attempted to include these aspects in monetary valuation exercises (e.g. Garcia Llorente et al., 2012).

The purpose of this paper is not to assess the impacts of changes caused to ecosystems on aggregated preferences (as monetization approaches do) but rather on the economic production and final consumption. This requires to clearly show the distinction between intermediate products (i.e. biological structures and processes that provide ecological functions) and final products (i.e. the resulting ecosystem services and their benefits). Ecosystem services generate benefits that can modify human wellbeing, economic production and final demand whereas biological structures and processes generate ecological functions that enter the production process of ecosystem services (Figure 1).

The distinction between intermediate and final products is however not new. It can be traced back to de Groot's (1992) distinction between functions – corresponding broadly to what is currently labeled intermediate products – and services. Later, the chain got divided in

additional steps and was thus refined (e.g. Boyd and Banzaf, 2007; Gomez-Baggethun and de Groot, 2010; Fisher et al., 2009). The distinction⁵ presented here is proposed by Haines-Young and Potschin (2010, pp. 115-116) in their cascade model. It is based on the simplified causal chain (Figure 1) going from the initial biological structure or process located in step $n-3$ (it covers the MA categories of supporting and regulating services) to the end result located in step n (i.e. to the benefit to individuals). Nursery habitats, forest soil cover, sediment accumulation in bays and forest soil fertilization are the four examples illustrated in Figure 1⁶.

Figure 1. Classification of ecosystem services based on the Haines-Young and Potschin (2010)'s cascade model.



⁵ Other distinctions do exist in literature (e.g. Fisher *et al.*, 2009).

⁶ Dashed arrows show several examples of feed-back impacts inside a same causal chain and cross-impacts between two causal chains.

The initial biological structure (e.g. soil particles in forests) is the natural biotic or abiotic physical support on which biological processes⁷ take place (e.g. primary production of tree biomass in forests) and includes all components of the physical organization of the environment.

Biological processes (*step n-3* in Figure 1) generate ecological functions, that is, the result of interactions between biological structures and processes that plays a role inside and for the ecosystem (*step n-2*) (e.g. the ecological function played by trees in structuring forest soils into a “sponge” that retains rain water). They are both considered as “intermediate products” provided by the ecosystem since they constitute a preliminary base before generating ecosystem services (*step n-1* of the causal chain) (Turner, 1999).

Ecosystem services are produced by intermediate products and part of the final products’ group⁸ because they are more directly related to individual uses (market and non-market uses) and hence to human wellbeing components and economic activities (e.g. flood protection). It includes what is often called in literature ‘outcome’, ‘goods’ or ‘services’ that are made available by the ecosystem for potential use by humans.

The benefit is the last step in the cascade model and is defined by Fisher et al. (2009) as the point where a natural component of the ecosystem meets human capital (e.g. knowledge) or technical capital (e.g. equipment, tools, machinery, buildings) to generate a good or a service that directly affects human wellbeing, i.e. individuals’ feeling of satisfaction and needs (e.g. damages to infrastructure, needs in recreational activities, ...). As a result, it seems reasonable to assume that this category is the only one that depends on individual preferences and hence, that should be valued in monetary units. As underlined by Fisher et al. (2009), this classification avoids any potential double counting problem.

⁷ A biological process is considered to be any causal chain in which a material resource (abiotic or biotic) or energy plays a role in the production of an identifiable end result.

⁸ A clear distinction between intermediate products and final products is not always possible. For example, provision of clean water by rivers may be used as an ecosystem service (final product) for human consumption purposes and/or as an ecological function (intermediate product) that ensures aquatic life.

Similarly, Turner *et al.* (2004) suggest that the valuation of intermediate products are independent of preferences and hence, are not included in the total economic value (TEV). However, intermediate products may give rise to non use values and accounting for them may result in double counting⁹.

From a more general perspective, the distinction between final products that can be monetized and intermediate products that should not be monetized needs to be tempered. First, few attempts to include ecosystem processes and properties (that is, intermediate products) in monetary evaluation exercises have been undertaken from a theoretical (Kontogianni *et al.*, 2010) and empirical (García-Llorente *et al.* (2011) perspectives. And second, institutional criteria also play a role. As a matter of fact, an ecosystem component that a society considers not salable or, said differently, without intention of exchange, does not need any money measure, independently of its classification. Hence, ecosystem components considered here are supposed to have a positive WTP. Moreover, when questionnaires are well built up, choice experiment (direct monetization approach) may enable to assess the effect on well-being (in terms of WTP) of different environmental management scenarios in different ecosystem components and even in intermediate products such as regulating services (e.g. Takatsuka *et al.* (2011), García-Llorente *et al.* (2012) or Johnston *et al.* (2013)). Nevertheless, this is not without some caveats as underlined in Spash *et al.* (2009) and Johnston and Russel (2011).

The semi-dynamic hybrid I-O model presented in the next Section proposes an alternative approach in which monetary and physical evaluations are combined into a multidimensional guiding framework (Section 4) for assessing the impacts of intermediate product changes on ecosystem services and benefits. This framework i) clearly differentiates the ecosystem components that are effectively measured by monetary techniques from those that are not and ii) ensures that what cannot be measured in monetary units is assessed in physical units by other techniques. It enables various ecological policy scenarios to be assessed and compared

⁹ Following the cascade model of Young and Potschin (2010), for avoiding double counting, the provisioning ecosystem service of fish resources should be considered as a final service obtained from the water flow regulating ecosystem service provided by trees and the supporting service provided by nursery habitats. In that case, water flow regulating and the supporting service should not be valued monetarily because their economic value is already included in the benefit obtained from the provisioning ecosystem service of fish resources.

with each other so that stakeholders can choose the scenario that best suits their needs, desires and projects.

3. Semi-dynamic hybrid I-O models for the analysis of ecological-economic processes

The ecological-economic modeling approach is based on a hybrid I-O model (e.g. Leontief (1970), Victor (1972), McDonald (2005), Lixon *et al.* (2008), Miller and Blair (2009) and others) that has become semi-dynamic. Its semi-dynamic property means that what happens in year $t-1$ has an impact on year t but input technical coefficients remain constant and its hybrid property enables monetary units to be used together with physical units and delivers results in both units. An I-O model describes how economic sectors exchange raw materials, semi-finished products and services between them (intermediate demand of inputs or also called “interindustrial exchanges” as in Figure 2) in order to produce final goods and services sold to households, NGOs, public administrations, foreign demand, etc. (final demand of inputs). In addition, hybrid I-O models give information on the consumption of natural resources and the emissions of wastes and pollutants linked with goods and services produced.

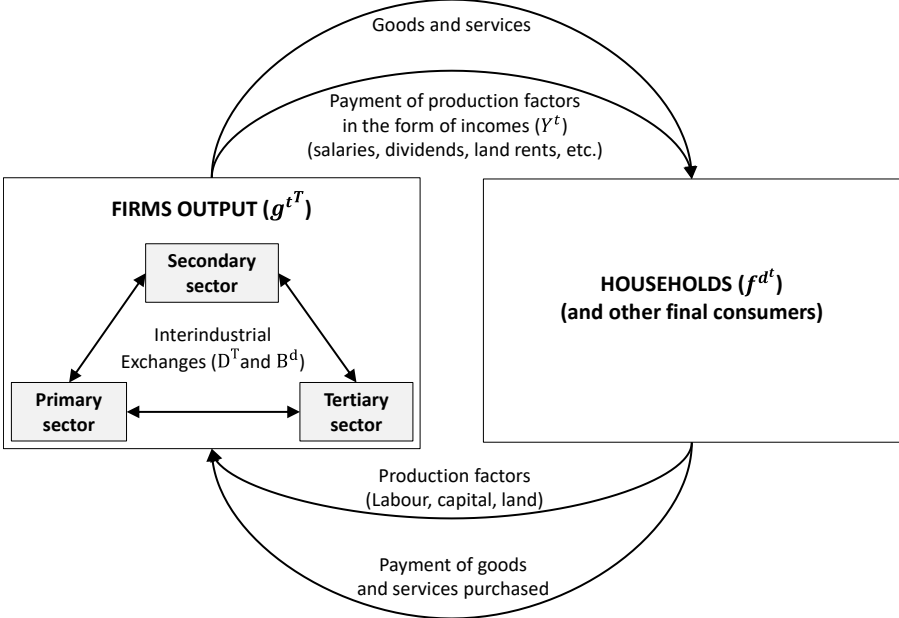


Figure 2. Scheme of the economic system as simulated by input-output models.

The I-O model used in this paper is based on *commodity by industry* tables. They are made of two tables: a Supply table and a Use table. The general Equation (1) of *commodity by*

industry I-O models calculates the direct and indirect impact on the production of output (g) of all other sectors based on changes in final demand (Lixon et al., 2008):

$$g^{tT} = \left[(I - D^T B^d)^{-1} D^T \right] f^{dt} \quad , \quad (1)$$

where g^{tT} is an $m \times 1$ column vector whose elements g_j^t are the total output per sector j produced during year t (exponent T means “transpose”), I is the $n \times m$ identity matrix ; B is a $n \times m$ matrix of *input technical coefficients* b_{ij} , D^T is a $n \times m$ matrix of the *commodity output proportions* d_{ij} , which are technical coefficients defined under the *industry-based technology* assumption. Both technical coefficients, B and D , are calculated respectively on the basis of supply matrix V (see Cordier *et al.* 2011) and the use matrix U , as in Lixon *et al.* (2008). Exponent d shows that consumption concerns inputs used inside the study area, which have been domestically produced in the study area; f^{dt} is an $n \times 1$ column vector representing the final demand in year t where each f_i^{dt} represents the value of regionally produced commodities i consumed by the p categories of final demand k ($k = 1, \dots, p$) that is to say: final consumption by households ($k = 1$), NGO ($k = 2$) and government ($k = 3$), gross fixed capital formation (i.e. investments) ($k = 4$), change in valuables ($k = 5$), change in inventories ($k = 6$), and international and interregional exports ($k = 7$). In other terms, each $f_i^{dt} = \sum_{k=1}^p f_{ik}^{dt}$.

Figure 3 offers a schematic example showing how ecological function, biological structures and processes as well as their indirect feedback impact on the economy are introduced into the I-O model in order to form a hybrid model. The impact of the economy on fish nurseries (a marine natural habitat) is used as an illustration. Quantification is made possible by the following techniques, combined with hybrid I-O modeling (numbers correspond to those shown in the Figure):

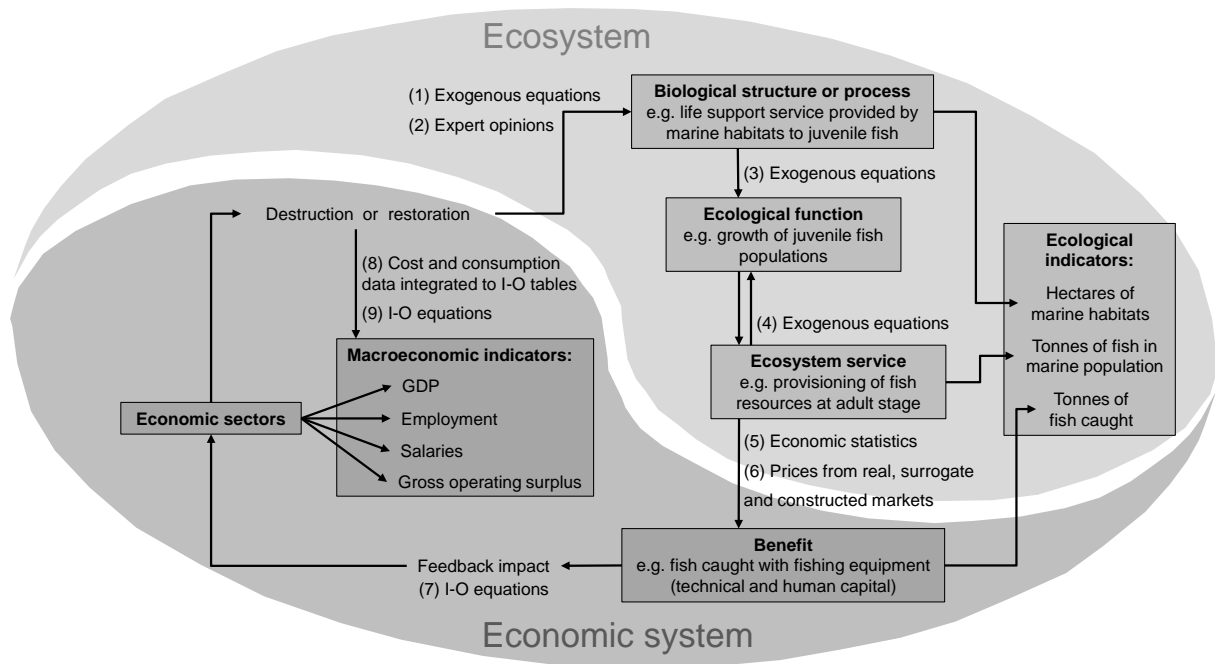


Figure 3. Quantification of relationships between the four steps of the cascade model: Biological structure and process, Ecological function, Ecosystem service, Benefit.

- (1) Exogenous equations calculating the evolution of the stock of biological structure (e.g. surface of marine habitats) due to the activity of Economic sectors – in the illustrative example, they are extrapolated from past trends.
- (2) Expert opinions adjusting results from the exogenous equations¹⁰.
- (3) Exogenous equations calculating the evolution of ecological functions due to changes in biological structures (e.g. equations calculating the link between the evolution of marine habitats and the size of the marine population of fish juveniles).
- (4) Exogenous equations calculating the evolution of ecosystem services due to changes in ecological functions (e.g. fish population equations calculating the quantity of adult fish based on the variation of the population of juvenile fish computed in (3)).

¹⁰ All results from exogenous equations and expert opinions are subject to a sensitivity analysis based on observed data to make sure that they are consistent. If they are not, it is a signal that equations and/or expert opinions must be refined and that further analysis is necessary.

- (5) Economic statistics on the use of ecosystem services (e.g. fishing statistics giving the percentage of fish caught in the total fish population of the study area).
- (6) Prices from real markets, constructed and surrogate markets (e.g. market prices of fish caught and sold on real market by the fishing sector).
- (7) I-O equations calculating the direct economic impact in the fishing sector and the indirect economic impacts on all other sectors that supply the fishing sector with intermediate goods and services (gasoline, fishing equipment, etc.).
- (8) Cost and consumption data integrated in I-O tables (e.g. cost of restoration of marine habitats, subsequent decrease in salaries and employment and effect on final household consumption).
- (9) I-O equations calculating the indirect impact of data integrated in I-O tables in step (8) (e.g. calculation of the impact of final household consumption on all productive sectors of the economy).

Exogenous equations and expert corrections enable us to take into account the idea of Isard (1968) that flows occurring inside the ecosystem (i.e. interactions between ecological functions and biological structures or processes as well as between ecological functions and ecosystem services) should also be included in ecological-economic models. This idea had been rejected by Victor (1972) and then suggested again by Carpentier (1994), although, to our knowledge, not implemented up to now (except for linear interactions such as in trophic chains – see an application in Jin *et al.* (2003)).

Figure 3 shows that there are two stages at which monetary units can be inserted into a hybrid I-O model to represent ecosystem services: *i*) when using market prices and prices from constructed and surrogate markets (arrow 6) and *ii*) when simulating various scenarios of destruction or restoration of biological structures or processes (arrow 8). The next Section presents the guiding framework that details how these monetary units can be inserted inside a hybrid I-O model without eclipsing the interesting advantages of physical units advocated in Section 2.

4. The guiding framework for monetization in ecological-economic models

The reason for integrating monetary values to the semi-dynamic hybrid I-O model is to assess the economic impact of a variation in the supply of an ecosystem service on production sectors and final demand. This differs from other I-O approaches, such as those developed by Cumberland (1966), Hannon (2001) and Grêt-Regamey and Kytzia (2007) who assess the impact of a variation in the supply of an ecosystem service on satisfaction feelings expressed by individuals (through a measure of individual preferences) as is usually the case in conventional cost-benefit analysis.

The guiding framework, illustrated in Figure 4, allows monetary valuation techniques to be inserted into the I-O model, but prevents them overshadowing the existence and the role of intermediate products as well as their interactions with ecosystem services (see Section 2). The framework is based on the partition of ecosystem components detailed in Figure 1 and operates according to three criteria detailed below.

4.1. First criterion: ecosystem components that are good candidates for monetization

This preliminary step enables to judge whether monetization is appropriate.

The ecosystem component is considered an intermediate product if it relates to a component of the ecosystem that participates in the generation of either an ecological function or an ecosystem service. It is considered an ecosystem service if individuals may obtain a direct benefit from a source of matter or energy taken from the ecosystem via a human or a technical capital (knowledge or know-how, equipment, tools, infrastructures, etc.).

If the ecosystem component is neither an intermediate product nor an ecosystem service, it is a benefit obtained from ecosystem services.

4.2. Second criterion: monetization criteria of candidate ecosystem components

If the ecosystem component is an intermediate or an ecosystem service, monetization is normally not considered in the framework (for the reasons mentioned in Section 2) except for restoration or preservation activities (see the third criterion).

But if the ecosystem component is a benefit obtained from the ecosystem, it can be valued in monetary units provided there is a market or intention of exchange. If not, then there is in principle no need for money measure as mentioned before: indeed, a very low WTP may mean that there is little or no intention to exchange (Pearce et al., 2006, Costanza et al., 1997, de Groot et al., 2002). Hence, in Figure 4, we should then select the path that goes toward the box “Evaluation in physical units”.

On the contrary, monetary valuation techniques apply for benefits for which there is a market or intention to exchange. Consequently, the third criterion questions whether the monetary value to be entered in the I-O model is calculated from a direct or an indirect approach.

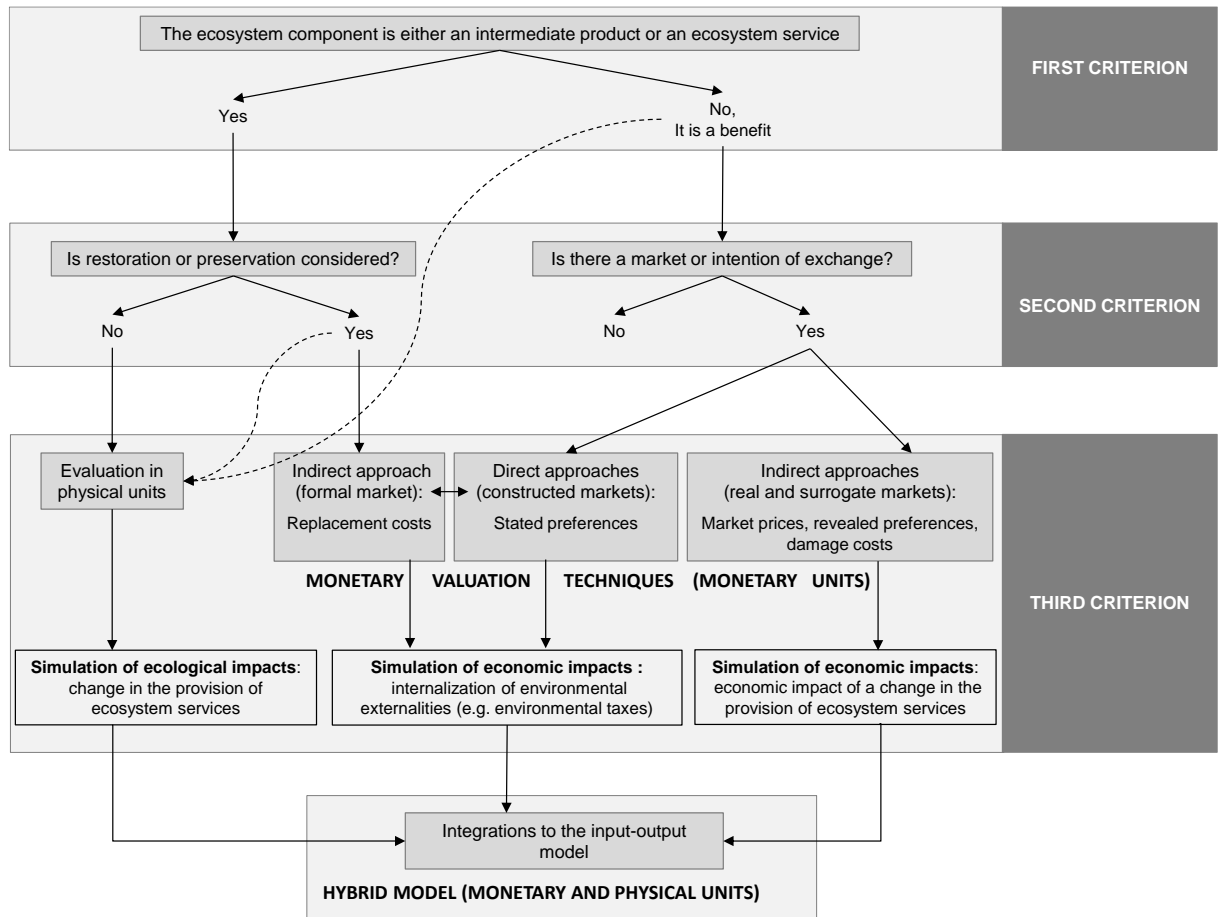


Figure 4. Guiding framework for the integration of monetary valuation techniques into a hybrid I-O model¹¹.

4.3. Third criterion: operational processes for integrating monetized ecosystem components into the hybrid I-O model

The third criterion (Figure 4) suggests three possibilities to integrate monetary values into the I-O model depending on whether the ecosystem component is i) a benefit monetized with a direct approach, ii) with an indirect approach or iii) if it is an intermediate product or an ecosystem service.

¹¹ Dashed arrows show that although monetary valuation is carried out, it is always possible to consider physical unit evaluations in parallel.

4.3.1. Benefits valued with direct approaches (WTP for environmental restoration)

They may be inserted into the I-O model to simulate the impact of the cost of environmental restoration/preservation on disposable income and, therefore, on final household consumption. The impact on gross operating surplus is also simulated further below. The environmental restoration or preservation measure considered in this paper is an environmental tax¹² aimed at funding environmental restoration or preservation activities.

The impact on final household consumption is calculated in Equation (2) assuming that households purchase final commodities ($f_{i,k=1}^{d^t}$) in proportion to their income. We are interested in calculating how WTP for environmental preservation modifies disposable income. This is carried out by transforming WTP values into a hypothetical tax in the model as shown below in Table 1 (note b) and in Appendix 1. This allows us to simulate the direct and indirect impacts of internalizing an externality on the economic system in the case where individuals would actually pay the amount they stated in the contingent valuation or the choice experiment, which reduces household consumption as illustrated in Equations (2) and (4). The calculation of the effect of the hypothetical tax consists first in subtracting from the final household consumption in previous year ($f_{i,k=1}^{d^{t-1}}$), the effect of the variation of disposable income (Y^t) due to the WTP paid:

$$f_{i,k=1}^{d^t} = f_{i,k=1}^{d^{t-1}} \left(1 + e_i \frac{Y^t - Y^{t-1}}{Y^{t-1}} \right) \quad , \quad i = 1, \dots, n \quad (2)$$

where $f_{i,k=1}^{d^{t-1}}$ and $f_{i,k=1}^{d^t}$ are respectively an element of the $n \times 1$ column vector $f_{k=1}^{d^{t-1}}$ (from Equation (1)) representing the consumption of commodity i by households ($k = 1$) at year $t - 1$ and t when the cost of environmental measure is applied; Y^{t-1} and Y^t are respectively the

¹² Introducing a market distortion such as a tax can worsen existing market failures. In this case, our goal is to find a way to take WTP into account and one method is to use a tax whose global return would be the WTP values, as a method for partly internalizing the externality, in the I-O model. But other economic instruments do exist (e.g. payment for ecosystem services, positive incentives such as subsidies, etc.) and it would be interesting to extend the model in order to integrate them in scenario simulations as the one developed in Appendix 1.

income¹³ available for final consumption in $t-1$ and t . And finally, the scalar e_i is the income-elasticity of demand for commodity i (elasticity values are available in Gohin, 2005).

If restoration costs exceeds WTP by a certain amount (Ψ^t), that extra amount is borne by economic sectors responsible for nursery destructions in application of the Polluter Pays Principle. However, it is likely that economic sectors would not accept to bear the full cost through a reduction of their benefits. This is why we arbitrarily assume that they would agree to pay half the cost through a reduction of their benefits (gross operating surplus: \widehat{GOS}) and the other half through a reduction of employment or salaries (Table 1 (note c)).

The Gross operating surplus is thus calculated as follows:

$$\widehat{GOS}_j^t = \frac{\widehat{GOS}_j^{t_0}}{g_j^{t_0}} g_j^t - \frac{1}{2} \Psi^t \alpha_{j_{tot}}, \quad (3)$$

where the gross operating surplus of sector j at time t (\widehat{GOS}_j^t) is calculated differently than in most I-O models: it is computed in proportion of the variation of total output of sector j between time t_0 ($g_j^{t_0}$) and time t (g_j^t) but, in addition, it is reduced as it bears half of the restoration costs, taking into account a sectorial burden-sharing rule ($\alpha_{j_{tot}}$).

This impacts final household consumption (Equation (2)). Disposable income Y^t is thus modified according to Equation (4).

$$Y^t = Y^{t-1} (1 + \rho_{k=1}^t) - WTP^t - \Psi^t \left(\frac{1}{2} + \frac{1}{2} \frac{\widehat{Shares}}{\widehat{GOS}} \right), \quad (4)$$

¹³ Y^{t-1} is proportional to total output per sector (g_j^{t-1}) from previous year: $Y^{t-1} = \sum_{j=1}^m \frac{y_j^{t_0}}{g_j^{t_0}} g_j^{t-1}$ where g_j^{t-1} is calculated from Equation (1) and the income available in reference year ($Y_j^{t_0}$) is calculated with data observed in the year t_0 by subtracting taxes on incomes, savings and social contributions from compensation of employees and adding the share of the gross operating surplus that is paid to shareholders in the form of dividends. And Y^t includes the aggregate WTP, thereby reducing household consumption from previous year.

Yearly income increases exogenously ($Y^{t-1}(1 + \rho_{k=1}^t)$) but households pay for restoration in t (WTP^t). The first term of the addition between brackets, $\frac{1}{2}$, is the first half of Ψ^t that companies pay with the cash released by reduction of salary levels¹⁴ or reductions in employment if salaries remain constant; the second term relates to the other half of Ψ^t that companies pay from their gross operating surplus. The proportion ($\frac{\widehat{Shares}}{\widehat{GOS}}$) of the gross operating surplus (\widehat{GOS}) distributed to shareholders in the form of dividends (\widehat{Shares}) is then reduced by $\Psi^t \frac{1}{2} \frac{\widehat{Shares}}{\widehat{GOS}}$. The value taken for this proportion is $\frac{\widehat{Shares}}{\widehat{GOS}} = 24\%$. It has been estimated with data for aggregated French economic sectors in 2007 (INSEE, accessed in 2010). Hence, Equation (4) shows that the hypothetical tax does not only include the WTP but also the cost of restoration that exceeds the WTP (Ψ^t).

Restoration costs also impact investment. This is calculated in Equation (5) where the positive effect of such investments ($\Psi^t + WTP^t$) on economic activities that are contracted to implement restoration are taken into account:

$$f_{k=4}^{dt} = \left(\sum_{j=1}^m \widehat{GOS}_{inv_j}^{t-1} \right) \widehat{cap} + (\Psi^t + WTP^t) , \quad (5)$$

$f_{k=4}^{dt}$ is an $n \times 1$ column vector of investments in commodities i made in the region in t ; $(\Psi^t + WTP^t)$ is a $n \times 1$ column vector with zero values except in the i^{th} line whose value is the total investment cost in environmental restoration, \widehat{cap} is a $n \times 1$ column vector whose elements \widehat{cap}_j are the fixed capital formation coefficients¹⁵. They are calculated based on the assumption that the share of investment in each commodity i in the total gross operating surplus invested by all sectors j in the reference year t_0 remains constant. $\sum_{j=1}^m \widehat{GOS}_{inv_j}^{t-1}$ is a scalar representing the part of the gross operating surplus that is invested in $t - 1$ by all

¹⁴ Reductions in salary levels may consist in direct monthly income reduction, salary freeze or in lower salaries for new employees.

¹⁵ This calculation of the fixed capital formation coefficients (\widehat{cap}_j) is available from the authors upon request. It enables to take into account the fact that companies bearing important restoration costs in $t-1$ are likely to reduce their investments in t . This is a good technique to estimate investment since fixed capital formation coefficients are quite stable through time, at least over a 13 year-period (verified in the I-O tables for France 1995-2007 published in Eurostat (2009)).

sectors j . It is calculated for each sector by Equation (6). The explanation is similar to that of Equation (3) and it also takes into account the part of the gross operating surplus that is invested $(1 - \frac{\widehat{Share}^{t_0}}{\widehat{GOS}^{t_0}})$:

$$\widehat{GOS}_{invj}^{t-1} = \frac{\widehat{GOS}_{invj}^{t_0}}{g_j^{t_0}} g_j^{t-1} - \frac{1}{2} \psi^{t-1} (1 - \frac{\widehat{Share}^{t_0}}{\widehat{GOS}^{t_0}}) \alpha_{j_{tot}} \quad (6)$$

The direct plus indirect impact of changes in final demand (household consumption from Equation (2) and investment from Equation (5)) on all other sectors of the economy in t is calculated through the general Equation (1). Equations (1) to (6) are applied to a real case study in Appendix 1 to test the integration of WTP into the hybrid I-O model. The Appendix also shows how to introduce replacement costs into the model (one of the indirect approaches listed in Figure 4). Note that these equations rely on the assumption that the increase in production costs, caused by environmental restoration or preservation costs, leads to a decrease in the gross operating surplus and salaries, rather than to a price increase. This might be a correct assumption for studies at regional levels (as is the case in Appendix 1) and for goods and services for which geographical location is not important (the price of such goods and services is influenced by national or international rather than by regional dynamics). Another option would be to consider solely an increase in sales prices. In that case, an I-O price model should be used (see the example for hedonic prices in the next subsection).

4.3.2. Benefits valued with indirect approaches (except replacement cost technique)

This relates to monetization techniques based on real and surrogate markets. Their results are integrated into the hybrid I-O model to simulate the impact of a change in the provision of ecosystem services on the production of economic goods and services by economic sectors. The I-O model enables to assess the direct and indirect effects of this change on the whole economic system through the links that connect an economic sector to its suppliers in raw material and semi-finished goods and services as well as to final consumers. This is carried out by integrating monetary values, that measure the change in the provision of an ecosystem service, inside the matrix F of final demand or the matrix B of intermediate inputs (in Equation (1), the p categories of final demand are summed for each commodity i , which transforms matrix F into a column vector symbolised by f^{dt}).

Table 1 (note (a)) shows that indirect approaches such as hedonic prices can be integrated into the I-O table in order to estimate the impact of a change in the provision of ecosystem services on prices of economic commodities. Based on the I-O model described above, new equations can be set up to obtain an I-O price model (cost push) as developed in Leontief (1974) and Miller and Blair (2009). The model can then be used to calculate relative price changes subject to an exogenous factor price increase, for example, an increase in housing prices (e.g. restoration program of natural habitats that improves landscape beauty and enhances recreational activities) thereby increasing demand for real estate goods.

Equations of the I-O price model require that an increase in the price of real estate goods due to natural habitat restoration programs automatically lead either to a decrease in the value added¹⁶ of the other sectors using such goods as intermediate inputs (e.g. rent of buildings) or to an increase in the prices of the economic commodities the other sectors produce (e.g. if natural habitat restoration brings about an increase in rental prices in a region, lawyers renting an office in that region might increase the price of their services in order to offset the rise in their rent).

The hedonic price information is then added to the value added w_j (Table 1, note (a)) corresponding to the real estate sector (and more specifically to the part of the value added corresponding to gross operating surplus because it is a pure benefit for the real estate sector).

The last step consists in translating the effect of price increases on final demand with the use of coefficients of price elasticity of demand found in the economic literature for each good and service. The subsequent modification of final demand would then be inserted in the column vector f^{dt} and the impact on the total output produced per economic sector in the area studied would be calculated via Equation (1). From that equation, the economic components of the wellbeing measured by employment and Gross Domestic Product (GDP) can be calculated for the whole economy of the study area as well as for each economic sector (see detailed mathematical developments in Victor, 1972). This gives a useful picture of the macro-economy and the sectoral economy of the study area. As an example, if the model shows that the price of basic commodities (food, housing, energy, etc.) would significantly

¹⁶ Because Value added = Receipts – Intermediate input consumptions.

increase when natural habitats are restored, this would be a signal for decision makers that some measures need to be taken: e.g. implementing fiscal instruments to help poorer households bear higher prices on basic commodities, creating a price observatory to impede producers to increase their prices, reducing the restored area to decrease the cost of environmental restoration...

4.3.3. Intermediate products and ecosystem services valued with the indirect approach of replacement costs

The guiding framework allows intermediate products and ecosystem services to be monetarily valued with the replacement cost method only when restoration or preservation activities are considered. Although it is an indirect approach, the technique of replacement cost is integrated into the model for a reason similar to direct approaches described in Section 4.3.1: to simulate the direct and indirect impacts of internalizing an externality on the economic system in the case where inhabitants or companies would accept (or would be forced by law or by economic incentives) to pay the amount required to replace ecosystem services they damaged.

As mentioned before (Section 4.3.1), when the sum of individuals' WTP does not enable to fully cover the restoration costs, then the remaining environmental restoration expenses are borne by economic sectors responsible for nursery destructions. To assess if the sum of individuals' WTP fully covers environmental restoration expenses, it is necessary to estimate the cost of activities and equipment that are required to replace an ecosystem service that has been destroyed.

Table 1. Use table of the hybrid I-O model applied to nurseries (marine habitats) of the Seine estuary.

		Economic sectors ($j = 1, \dots, m$)	Final demand ($k=1, \dots, p$)	Ecosystem	Total
Economic commodities (M€) ($i = 1, \dots, n$)		U u_{ij}	F f_{ik}		q_i
Imports (M€)			m_j		$\sum_{j=1}^{m+p} m_j$
Primary inputs (value added: benefits (GOS), wages and salaries, net taxes) (M€) ($z = 1, \dots, t$)		W W_{zj}	μ_k		$\sum_{j=1}^m y_{zj}$
Total (M€)		g_j			$\sum_{j=1}^m (g_j) + \sum_{k=1}^p (\mu_k)$
Ecological commodities	Nursery (km ²)	r^N		$(e^N)^{Seine}$	$r^N + (e^N)^{Seine}$
	Fish (t)		r_j^S	$(e^S)^{Seine}$	$\sum_{j=1}^{m+p} r_j^S + (e^S)^{Seine}$

a. $\cdots \rightarrow$ **Integration of hedonic prices (indirect approach)** to model the impact of the restoration of natural habitat (Nursery) on final demand.

b. $\cdots \rightarrow$ **Integration of prices from direct approach (contingent valuation, choice experiment)** to model the effect of a new environmental tax set at the WTP value for the restoration of natural habitats (Nursery). The integration is made through Equation (2) and (5)

c. $\cdots \rightarrow$ **Integration of prices from direct approach (contingent valuation, choice experiment)**: the modification consists in taking the share of the cost of restoration of natural habitats (Nursery) that exceeds the amount declared by individuals in direct approaches (b) and subtracting them from the benefits (GOS: Gross Operating Surplus) produced and the salaries paid by economic sectors as developed in Equations (3) and (4).

Legend:

U: use matrix, **F**: matrix of final demand, **W**: matrix of primary inputs (covers all categories of value added: benefits (gross operating surplus), wages and salaries, net taxes), g_j : row vector of total output per sector j , q_i : column vector of the total demand per commodity i , m_j : row vector of interregional and international imports, μ_k : row vector of total input consumed per category of final demand, r_j^S : quantity of fish consumed by sector j or final demand k in the Seine estuary, r^N : total amount of nursery areas destroyed by all economic sectors, $(e^N)^{Seine}$: nursery surface that can be used by the ecosystem of the Seine estuary, $(e^S)^{Seine}$: tonnage of fish that can be used by the ecosystem of the Seine estuary.

5. Discussion and conclusion

This paper proposes an alternative approach for reconciling monetary valuation techniques with methods that address ecosystem-economy interactions. To achieve this goal, we develop a guiding framework that limits the use of monetary valuation to real market simulations and potential market simulations. Simulations of scenarios of environmental measures are carried out with a semi-dynamic hybrid input-output model. The guiding framework ensures that monetary valuation techniques contribute to the understanding of the impact of economic activities on changes in ecosystems services and the feedback impact of these changes on economic activities.

The first criterion of the guiding framework establishes that benefits generated by ecosystem services are measured by monetary valuation techniques while intermediate products or ecosystem services are measured in physical units, except when restoration or preservation is considered, as illustrated by the second criterion. However, papers such as Hannon (2001) and Grêt-Regamey and Kytzia (2007) show that our position is not one shared by all economists. Our guiding framework uses results from monetary valuation techniques to assess the economic impact of a variation in the supply of an ecosystem service on production sectors and final demand while both authors mentioned above use them to assess the impact of a variation in the supply of an ecosystem service on feeling of satisfaction expressed by individuals in direct and indirect monetary approaches used in environmental economics.

To monetize benefits generated by ecosystem services, the third criterion of the guiding framework suggests operational processes for integrating results from monetary valuation techniques to the semi-dynamic hybrid I-O model. The framework also allows the integration of results from natural sciences in physical units. Three categories of impacts are considered: *i*) ecological impact valued in physical units, that is, a change in the provision of ecosystem components (intermediate and final products as defined in Figure 1), *ii*) subsequent feed-back impacts of these changes on economic production and final demand as measured with indirect approaches (real and surrogate markets), and *iii*) economic impacts of the internalization of environmental externalities such as an environmental tax computed from a direct and/or indirect approach (replacement costs).

Dealing with cultural services within the guiding framework is possible but more difficult as these services are not always tangible. As a matter of fact, in the example of the bay of Mont Saint-Michel island in France (Figure 1(c)), cultural service like recreation can easily be quantified in terms of the number of tourists hiking in the bay each year and in terms of euros paid to hiking guides or tourism agencies. However, cultural services such as esthetical service offered by the bay is more difficult to assess.

Even though the guiding framework remains largely theoretical at this stage and should still be tested on additional case studies to that developed in Appendix 1, one originality of the guiding framework lies in that it shifts the traditional focus of monetary valuation from the analysis of impacts of changes in ecosystem services on aggregated individual satisfaction

and preferences towards impacts on production sectors and final demand. Another interesting aspect is the way the framework combines biophysical and monetary data and attempts to differentiate the suitable domain for using either of them. We believe that such distinction is an important one that is often overseen in the literature on ecosystem services valuation. This framework may give natural scientists a better understanding of how to take advantage of economics when analyzing the impacts of interactions between the economy and the ecosystem.

Appendix 1 – Monetary values from direct approaches and the indirect approach of replacement costs

In order to test our methodology, we built two scenarios: a “business as usual” scenario (BAU) and a scenario of nursery restoration in the Seine estuary (located in France’s Haute-Normandie region). Both scenarios have been simulated with a version of the static model of Cordier et al. (2011) that has been semi-dynamized and modified for integrating the WTP resulting from direct monetization approaches. Results are compared with the original situation of 2007 whose values are set to 100.

The BAU scenario describes the annual evolution of the ecosystem and the economy between the reference year 2007 and the horizon year 2018 as it would evolve if current ecological and economic trends were maintained and no restoration of nursery areas were undertaken.

The restoration scenario hypothesizes that 25% of the losses in sandy nursery with high fish density are restored over the period 2008–2018. This consists in a restoration of 2.2 km² per year, or 24.38 km² at the end of the period. The expected impact would increase the capacity of fish populations to regenerate and our simulation results suggest that the growth in fish stock might indeed be significant: the population of common soles (*Solea solea sp.*) born in the Seine estuary would rise from 100.0 to 123.2 in 11 years for the restoration scenario compared to 104.0 under the BAU scenario (Figure 5(a)).

The annual cost of restoration amounts to M€ 386 ($= \Psi^t + WTP$)¹⁷. This value includes all costs incurred in the restoration of subtidal nurseries (nurseries below tide marks, i.e. always under water) of 1m cmh¹⁸ depth located between 500m and 3000m from the shore. The restoration technique consists in dredging sediments from the navigation channel in the Seine estuary and transporting them to the restoration area.

In the guiding framework (Figure 4), WTP resulting from direct approaches are integrated into the model to simulate the impact of the cost of environmental restoration on disposable income (Section 4.3.1). Beaumais et al. (2008) carried out a contingent valuation in the Seine estuary to assess the economic value of wetlands (nurseries included) and computed a maximum yearly WTP across all households of the Seine estuary amounting to M€ 21.89. This value has been integrated into the model as in Equations (4) and (5).

However, the replacement costs technique (Section 4.3.3) shows that the cost of equipment and activities required to replace destroyed nurseries (M€ 386 per year $= \Psi^t + WTP^t$) is much higher than the total aggregated WTP (M€ 21.89 per year $= WTP^{t_0}$). Costs (Ψ^t) of nursery restoration that are not covered by the WTP are also entered into the model but they are borne by economic sectors, not by inhabitants. The impacts of restoration on GDP is illustrated in Figure 5(b): GDP increases from the 2007 base value of 100.0 to 120.8 without restoration (BAU) and 115.7 if restoration activities are carried out. In other words, without restoration the annual average GDP growth over the 2007-2018 period would reach 1.7% whereas with restoration activities, it would reach 1.3%. This negative macro-economic impact is relatively small compared to the sectorial impact shown in Figure 6. This shows that the main issue is not the financial burden of restoration but rather, the distribution burden (who pays for restoration).

¹⁷ all prices mentioned in the Appendix are 2007 prices

¹⁸ cmh : cote marine du Havre (marine reference dimension of Le Havre).

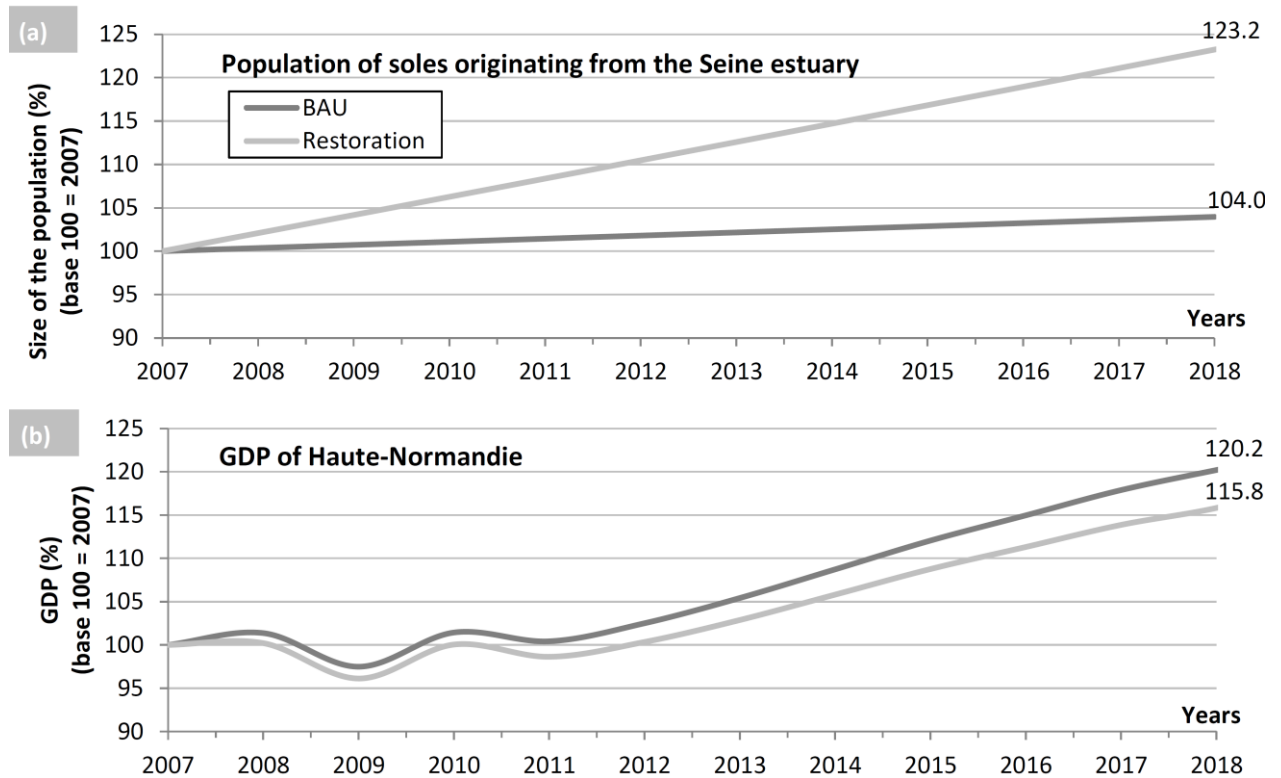


Figure 5. Simulations of ecological and macroeconomic impacts of the Restoration and BAU scenarios.

Figure 6 shows the sectorial economic impact for 4 out of 12 sectors of the regional economy. The remaining sectors are not shown because their graphs are similar to that of the Agriculture sector (Figure 6 (a)) where the gap between both curves is relatively small, which suggests that the losses in terms of benefits caused by restoration activities are relatively small: BAU benefits amount to 117.8 compared to 111.0 for the restoration scenario in 2018. Moreover, the gross operating surplus keeps growing year after year for both scenarios, albeit at a slower rate in the restoration scenario.

Three sectors are nevertheless hit harder in the restoration scenario because they bear the highest responsibility in nursery destructions and the burden sharing applied in this paper allocates restoration costs to economic sectors in proportion to their responsibility ($\alpha_{j_{tot}}$ in Equation (3)). They are the Mining sector, the sector of Manufactures of coke, refined petroleum and nuclear fuels, and the Harbour sector (Figure 6 (b), (c), (d)). The high costs borne by those sectors reduce their gross operating surplus since we assume that they will pay half the cost through a reduction of their gross operating surplus and the other half through a reduction of employment or salaries (Equations (3) and (4)). The mining sector is the most heavily impacted since its gross operating surplus in 2018 rises to M€ 124.4 without

restoration and drops to M€ 79.3 with restoration, i.e. a lower level than in 2007. Restoration therefore represents a loss of 36% in gross operating surplus. This is a result of the burden sharing rule which makes the Mining sector to pay one of the highest shares of the annual restoration costs: $\alpha_{j_{tot}} = 18.3 \%$. The losses in the harbour sector and in the sector of Manufactures of coke, refined petroleum and nuclear fuels are smaller (13.6% and 18.5% respectively) although they also bear high share of restoration costs ($\alpha_{j_{tot}} = 33.4 \%$ and 15.3% respectively).

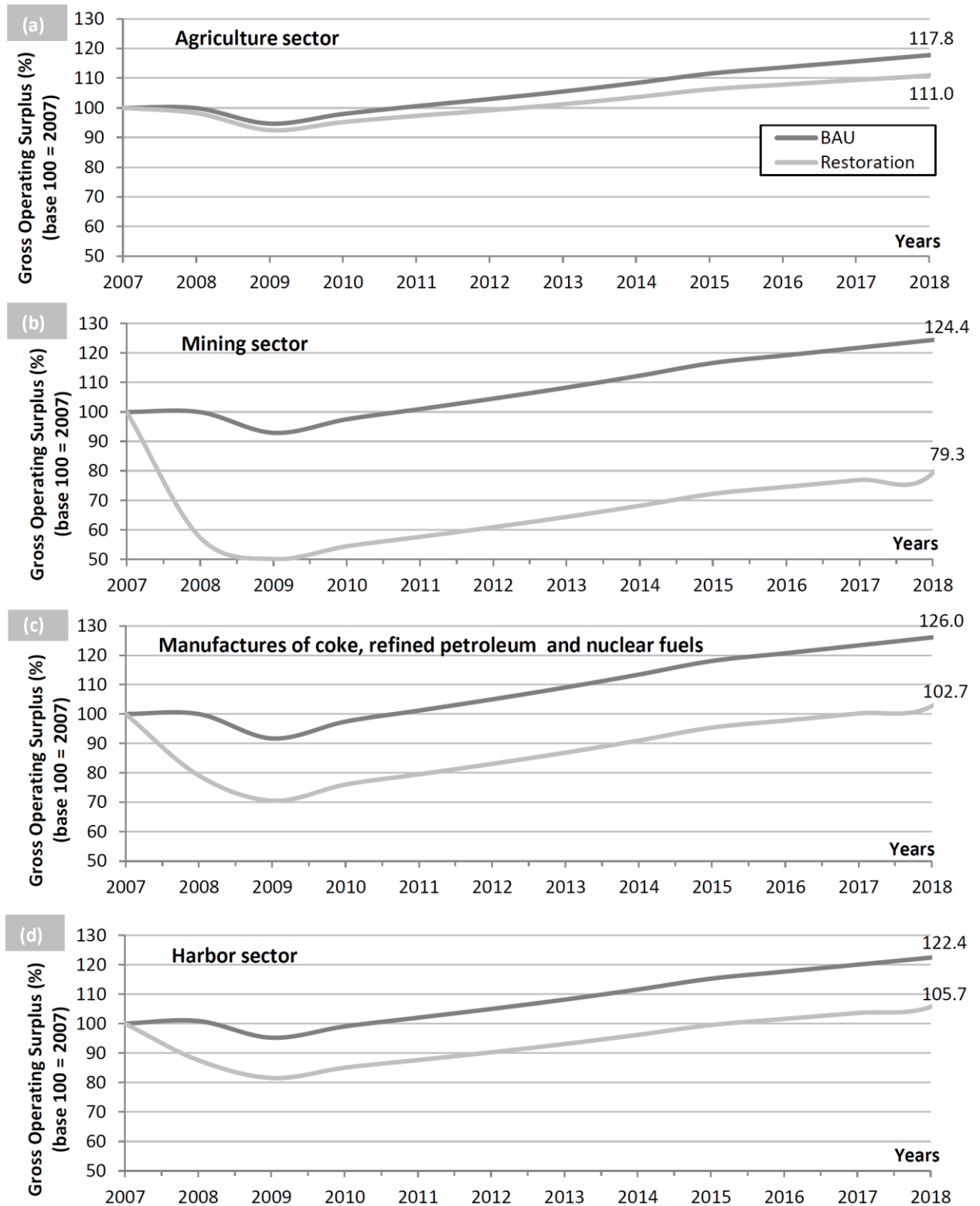


Figure. 6. Simulations of sectorial economic impacts of the Restoration and the BAU scenarios

Acknowledgements

We are grateful to Kate Weir for checking the English. Her help is always appreciated. We are also very much indebted to the three anonymous reviewers for their very interesting and constructive comments and suggestions.

Bibliography

- Ackerman, F., 2004. Priceless Benefits, Costly Mistakes: What's Wrong With Cost-Benefit Analysis? *Post-autistic Econ. Rev.* 25, 2–7.
- Ashford, N.A., 1981. Alternatives to cost-benefit analysis in regulatory decisions. *Annals of the New York Acad. of Sci.* 363, 129–137.
- Beaumais, O., Laroutis D., Chakir, R., 2008. Conservation versus conversion des zones humides: une analyse comparative appliquée à l'estuaire de la Seine. *Revue d'Économie Régionale & Urbaine*, n°2008/4 Novembre, pp. 565-590.
- Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecol. Econ.* 63(2–3), 616–626.
- Carbone, J.C., Smith, V. Kerry, 2013. Valuing nature in a general equilibrium. *J. Environ. Econ. Manage.* 66, 72-89.
- Carpenter, S.R., De Fries, R., *et al.*, 2006. Millennium ecosystem assessment: research needs. *Sci.* 314(5797), 257–258.
- Carpentier, C.L., 1994. Agriculture and the environment: an economic–ecological input–output model of the Canadian economy. PhD thesis, McGill University, Montreal, p121.
- Carson, R.T., Hanemann, W.M., 2005. Contingent Valuation. In K-G Mäler and J.R. Vincent (Ed). *Handbook of environmental economics*, Vol2, Chapter 17, 821-937. Elsevier Publishing, Netherlands.
- Cordier, M., Pérez Agúndez, J.A., O'Connor, M., Rochette, S., Hecq, W., 2011. Quantification of interdependencies between economic systems and ecosystem services: an input-output model applied to the Seine estuary. *Ecol. Econ.* 70(9), 1660–1671.
- Costanza, R., d'Arge, R., de Groot, R.S., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. *Nature* 387, 253–260.
- Cumberland, J.H., 1966. A regional inter-industry model for the analysis of development objectives. *Pap. Reg. Sci.* 17, 64–94.

- Daily, G., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., Shallenberger, R., 2009. Ecosystem services in decision-making: time to deliver. *Front. Ecol. Environ.* 7(1), 21–28.
- Daily, G.C., 1997. Introduction: what are ecosystem services. In: Daily GC (Ed), *Nature's Services*. Island Press, Washington DC, pp. 1–10.
- Daly, H.E., 1968. On Economics as a Life Science. *J. Polit. Econ.* 76, 392–406.
- de Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecol. Econ.* 41, 393–408.
- de Groot, R.S., 1992. *Functions of Nature: Evaluation of nature in environmental planning, management and decision making*, Wolters-Noordhoff BV, Groningen, p. 315.
- Fisher, B., Turner, K.R., Morling, P., 2009. Defining and classifying ecosystem services for decision making. *Ecol. Econ.* 68, 643–653.
- García-Llorente, M., Martin-Lopez, B., Diaz, S., Montes, C., 2011. Can ecosystem properties be fully translated into service values? An economic valuation of aquatic plants services. *Ecol. Appl.* 21:3083-3103.
- Garcia-Llorente, M., Martin-Lopez, B., Nunes, P., Castro, A., Montes, C., 2012. A choice experiment study for land-use scenarios in semi-arid watershed environments. *J. Arid Environ.* 87, 219-230.
- Gohin, A., 2005. The specification of price and income elasticities in computable general equilibrium models: An application of latent separability. *Econ. Model.* 22(5), 905–925.
- Gomez-Baggethun, E., de Groot, R., 2010. Natural capital and ecosystem services: the ecological foundation of human society. In R. Harrison and R. Hester (Ed.). *Ecosystem Services*. Chapter 5, 105-121. RSC Publishing, London, UK.
- Grêt-Regamey, A., Kytzia, S., 2007. Integrating the valuation of ecosystem services into the Input-output economics of an Alpine region. *Ecol. Econ.* 63, 786–798.
- Haines-Young, R., Potschin, M., 2010. The links between biodiversity, ecosystem services and human well-being D. Raffaelli, C. Frid (Eds.), *Ecosystems Ecology: a New Synthesis*, Cambridge University Press, Cambridge (2010), pp. 110-139.
- Hannon, B., 2001. Ecological pricing and economic efficiency. *Ecol. Econ.* 36, 19–30.
- Insee, accessed in 2010. Revenus des ménages - Revenus primaires - Revenus de la propriété.
- Isard, W., 1968. Some Notes on the Linkage of Ecological and Economic Systems. Paper presented at the European Congress of the Regional Science Association, Budapest.

- Jin, D., Hoagland, P., Dalton, T.M., 2003. Linking economic and ecological models for a marine ecosystem. *Ecol. Econ.* 46, 367–385.
- Johnston, R.J., Russell, M., 2011. “An Operational Structure for Clarity in Ecosystem Service Values.” *Ecol. Econ.* 70(13): 2243–2249.
- Johnston, R.J., Schultz, E.T., Segerson, K., Besedin, E.Y., Ramachandran, M., 2013. Stated Preferences for Intermediate versus Final Ecosystem Services: Disentangling Willingness to Pay for Omitted Outcomes. *Agricultural and Resource Economics Review* 42: 98-118.
- Kontogianni, A., Luck, G.W; Skourtos, M., 2010. Valuing ecosystem services on the basis of service-providing units: A potential approach to address the 'endpoint problem' and improve stated preference methods. *Ecol. Econ.* 69:1479-1487.
- Leontief, W., 1970. Environmental Repercussions and the Economic Structure: An Input-Output Approach. *Rev. Econ. Stat.* 52(3), 262–271.
- Leontief, W., 1974. *Essais d'économiques*. Calman Lévy, Paris, p. 316.
- Lixon, B., Thomassin, P.J., Hamaide, B., 2008. Industrial output restriction and the Kyoto protocol: An input–output approach with application to Canada. *Ecol. Econ.* 68, 249–258.
- Markandya, A., Hunt, A., Milborrow, I., 2005. Developments in green accounting. In: Tamborra, M. and Markandya, A. (Eds) *Green accounting in Europe. A comparative study*, vol. 2. Edward Elgar, Cheltenham Publishing Ltd., United Kingdom, pp. 15–33.
- McDonald, G., 2005. Integrating economics and ecology: a systems approach to sustainability in the Auckland region. PhD thesis, Massey University, Palmerston North, New Zealand, p. 597.
- Millennium Ecosystem Assessment (MA), 2005. *Ecosystems and Human Well-being: Synthesis*. Island Press (Ed.), Washington DC, p. 139.
- Miller, R.E., Blair, P.D., 2009. *Input-output Analysis. Foundations and Extensions*. Cambridge University press (Ed.), United-Kingdom, p. 750.
- O'Connor, M., 2000. Natural capital. In: Spash, C.L. and Carter, C. (Eds). *Environmental valuation in Europe*, Policy research brief 3, Cambridge, p. 24.
- Pearce D. W., Atkinson G., Mourato S. *Cost-benefit analysis and the environment: recent developments*, OCDE (Ed.), Paris, France, 2006.
- Plummer, M.L., 2009. Assessing benefit transfer for the valuation of ecosystem services. *Front. Ecol. Environ.* 7(1), 38-45.
- Polasky, S., Nelson, E., Pennington, D., Johnson, K.A., 2011. The Impact of Land-Use Change on Ecosystem Services, Biodiversity and Returns to Landowners: A Case Study in the State of Minnesota. *Env. Res. Econ.* 48, 219–242.

- Rowlands, I., Scott, D., Parker, P., 2003. Consumers and green electricity: profiling potential purchasers, *Business Strategy and the Environment* 12, 36–48.
- Sachs, J.D., Reid, W.V., 2006. Environment — investments toward sustainable development. *Sci.* 312(5776), 1002.
- Smith, V. Kerry, Desvouges, W.H., Fisher, A., 1986. A comparison of direct and indirect methods for estimating environmental benefits. *Am. J. Agr. Econ.* 68(2), 280-290.
- Spash, C., Urama, K., Burton, R., Kenyon, W., Shannon, P., Hill, G., 2009. Motives behind willingness to pay for improving biodiversity in a water ecosystem: Economics, ethics and social psychology. *Ecol. Econ.* 68 (4), 955–964.
- Turner, K., Georgiou, S., Clark, R., Brouwer, R., Burke, J., 2004. Economic valuation of water resources in agriculture: From the sectoral to a functional perspective of natural resource management. FAO, Rome, p. 189.
- Turner, R.K., 1999. Markets and environmental quality. In: Clark G.L., Feldman M.P., Gertler M.S. (Eds). *The Oxford Handbook of Economic Geography*, Oxford University Press, Oxford, pp. 585–606.
- Venkatachalam, L., 2007. Environmental economics and ecological economics: where they can converge? *Ecol. Econ.* 61, 550–558.
- Victor, A.P., 1972. *Pollution: economy and environment*. Georges Allen &Unwin Ltd., Great Britain, p. 247.
- Wincler, R., 2006. Valuation of ecosystem goods and services Part 1: An integrated dynamic approach. *Ecol. Econ.* 59, 82–93.