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Considering Ecosystem Services in Life Cycle Assessment to Evaluate Environmental Externalities

Aurélien Bruel[a,b,]*, Nadège Troussier[a], Bertrand Guillaume[a], Natalia Sirina[a]

[a]Charles Delaunay Institute, Troyes University of Technology,12, rue Marie Curie, Troyes 10010, France
[b]INSPIRE Institute, 11, cours Joseph Thierry, Marseille 13001, France

* Corresponding author. Tel.: +33 351 59 12 77. *E-mail address:* aurelien.bruel@utt.fr

Abstract

Environmental externalities are a typical example of market failures. These market failures could be corrected if decision makers had the “right” information and were aware of the real costs of their activities. Currently, companies consider biodiversity and Ecosystem Services (ES) as environmental externalities. The idea proposed in this paper is to study the links between the life cycle of a product and ES. To achieve that, the paper presents an approach to account for ES in Life Cycle Assessment (LCA). This proposal is illustrated in an industrial case study by means of a bio economic model of ES.

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Keywords: Environmental externalities; Biodiversity; Ecosystem services; Product life cycle; Life cycle assessment

1. Introduction

Environmental externalities are typical examples of market failures. To correct market failures and achieve what, in economic terms, is defined as the allocation of optimal resources, these externalities should be internalized, taken into account in the economic system and thus be reflected in the prices of goods and services [1,2].

Some studies point out that decision makers, product developers, or consumers would take different decisions if they had information allowing them to assess the wider environmental consequences of their choices and if they had to pay the prices corresponding to monetary value of externalities of using matter and energy [2–4]. According to [5], environmental problems are caused by regulation and market failures that could be corrected if decision makers had the “right” information and were exposed to the real costs of their activities.

The purpose of this article is to develop an approach for assessing environmental externalities at product level to support decisions. To obtain this information, it would be interesting to look at the interactions between product, ecological and socioeconomic systems. In this paper, the development of this approach relies on Life Cycle Assessment (LCA), Ecosystem Services (ES) and monetary valuation. It provides a monetary indicator of changes in the supply of ES.

Firstly, the methodology followed in this paper is presented. Then, the literature review used in this paper describes the concepts and tools on which we rely on. The next section displays the proposal. Then, the proposal is illustrated in a case study. Finally, the discussion and conclusion end the paper.

2. Methodology

This research study is situated in the field of Industrial Ecology (IE). Some researchers have mentioned the relationships between IE and environmental externalities [4,5]. The scientific methodology is based on the analysis of the state of the art on the approaches to evaluate environmental impacts and externalities in order to build an original proposal.

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This proposal is illustrated in a case study in order to show the relevance for the addressed research question.

3. Environmental externalities

3.1. Definition

An environmental externality corresponds to negative or positive consequences to a third party, of one or more economic activity of production or consumption [6,7]. This occurs when costs are imposed by one party vis-à-vis another without payment or compensation, or, on the other hand, when one party enjoys the benefits of an economic activity without offering any reward in return. In economics, they are respectively a loss or gain of welfare. Most of the time, environmental externalities are additional costs borne by public authorities (e.g. government, local authorities) and citizens. In particular, environmental externalities result in atmospheric pollution, noise pollution, emissions of greenhouse gases, pollution of water and soil, but also damage to biodiversity and ES. In general, the costs and benefits associated with externalities are not accounted for and quantification turns out to be a key issue [8]. Monetary evaluation in this case could be a useful tool to account for environmental externalities.

3.2. Monetary valuation

Monetary valuation is strictly related to the concept of externalities in welfare economics [9]. However, there are ethical objections to monetary valuation. These stem from a position commonly found in strong sustainability approaches, that some values are non-tradable, and from the misunderstanding that monetary valuation attaches a monetary value to human life or biodiversity in absolute terms. However, the main purpose of monetary valuation is not to put a price on the environment and its component parts, but to estimate the value of marginal changes in the availability of non-market goods [10]. Changes in availability concern both changes in the amount and in the quality of a good and the service that it provides to society. They allow economists and other practitioners to measure the individuals Willingness To Pay (WTP) to avoid the change or Willingness To Accept (WTA) compensation to consent to the change.

Monetary valuation might support decision making, and when it is used to convert the social and biophysical impacts of non-market goods into monetary units, they can be compared against each other and against the costs and benefits already expressed in monetary units. Biodiversity and ES are typical non-market goods for which no market exists.

In this article, we propose to get monetary indicators of changes in the supply of ES to provide relevant information.

4. Ecosystem services

4.1. Definition and conceptual elements

In recent years, the ES concept has become an important research topic addressing the links between ecological and socio-economic systems. The year 1997 was an important one for this concept, with the publication of a book that symbolizes its emergence in academics [11] and an article with a strong media impact [12]. Subsequently this concept has attracted great interest in the scientific community and the international work of Millennium Ecosystem Assessment (MEA) [11] between 2001 and 2005 helped to place the ES concept on the world political agenda. Since the MEA, other studies have been carried out on ES, such as The Economics of Ecosystem and Biodiversity (TEEB) in 2010 [12]. That focused on the economic aspects of biodiversity and ES, and the cost of policy inaction related to the environmental damage occurring in the absence of an effective regulatory framework.

There are different definitions of ES. The European study Mapping and Assessment of Ecosystems and Their Services (MAES) gives one of the clearest from our point of view and is used in this study [13]. Ecosystems are formed by interactions of communities of living organisms with the abiotic environment. Biodiversity plays a key role in the structural configuration of ecosystems, which is essential to maintain basic ecosystem processes and ecosystem support functions. These functions represent the ability or potential to deliver ES. Regarding ES, they are ecological flows derived from ecosystem functions that meet the present or potential demand of future generations. When ES are used or consumed, they generate benefits that result in increased human well-being induced by the satisfaction of a need or a desire [12].

In the study led by the MEA, it was shown that human activities in the last 50 years have altered ecosystems at an unparalleled pace compared to other historical periods, mainly to respond to the growing global demand for food, fresh water, timber, fiber, energy and other materials. The recognized impacts of many industries contribute significantly to the degradation of ES.

4.2. Relationships between industrial systems and ES

Industrial systems are very involved in and responsible for pressures on ES. [11], identified six major challenges directly involving industrial activities: scarcity of fresh water; climate change; habitat degradation; exotic species; overexploitation of oceans and overload of nutrients in water. There are two types of pressures from industrial systems. The first one is related to the dependency they establish to collect natural resources they need for their supply chain. The second one is directly linked to pollutant emissions and the release of wastes in the environment. The dependence and impact of industrial systems can vary between sectors and even in the same sector. They are more or less important depending on the technology used and the context of the production activity. For instance, industrial systems impact sites directly, especially by building production equipment, but also by releasing pollution into ecosystems via the production process. Moreover, they have an indirect impact via their suppliers of raw materials and semi-finished products.

Integrating the links between industrial systems and ES in corporate information systems is a methodological challenge that requires reliable accounting and management systems that supply relevant information and support operational decisions [14]. The idea proposed in this paper is to study the links between the life cycle of product and ES. The article focuses in particular on the environmental impact generated on ES by the product life cycle.

5. Linking the product life cycle and ES

5.1. Life cycle oriented tools

There are many life-cycle oriented tools available for modeling the links between industrial and ecological systems, but none of them take ES into account. Recently a study has compared life-cycle oriented tools which could take into account ES [15]. It includes tools developed by ecologists for quantifying ES; by ecological economists for monetary valuation; life cycle tools, such Life Cycle Assessment (LCA); thermodynamic methods for resource accounting, such as exergy and emergy analysis; variations of the ecological footprint approach, and human appropriation of net primary productivity. Each of these tools has been compared in terms of their ability to take ES into account, following the MEA typology. This study shows that some tools consider ES partially, but that none considers the exhaustive list of ES. The authors emphasize the need to take all ES into account in one tool to avoid providing misleading results and even encouraging wrong decisions. In addition to accounting for all ES, it is also important to consider a life cycle view and find a way of interpreting the results via aggregate metrics. Among these life cycle oriented tools, TEEB in 2012 and [16,17] have suggested taking ES into account in LCA - which is recognized and implemented by companies, public institutions and academics.

5.2. Accounting for ES in LCA

LCA is a widely used environmental assessment methodology, which evaluates the environmental impact of a product throughout its life cycle, from extraction of raw materials to its end of life. Its purpose is to supply information that can be used by governments, companies and consumers to decrease the consumption of natural resources, and reduce emissions and environmental impacts throughout the life cycle of products, services and systems [18] following four different phases [19–21].
Recently, a few approaches have already tried to account for ES in LCA [22–25]. They take ES into account in the LCIA phase of LCA. The table below (Tab. 1) compares these approaches according to a variety of criteria that might influence the modeling choices and information towards decision makers. These approaches are based on traditional LCIA and they calculate potential impacts. They are generalized and aggregated in time and space and thus do not refer to specific situations. Therefore, they do not characterize the loss of welfare associated with degradation of consumption or use of ES by one or more individuals in a specific situation accurately.

Table 1. Comparison of existing approaches

Criteria \ Approach	[23]	[24]	[22]
Type of indicator	Ecoenergetics	Biophysics and monetary	Biophysics and monetary
Accounting for ES in LCA	LCIA	LCIA	LCIA
Type of impact	Generalizable	Generalizable	Generalizable
Classification of ES	MEA	[26]	MEA
Model or method used	Exergy and emergy methods	GUMBO model	LANCA model

6. Proposal

Unlike some existing approaches outlined above, we propose developing an approach to assess changes in the supply of ES in specific or local situations. We suggest relying on the Life Cycle Inventory (LCI) phase of LCA connected to bio economic models of ES. These bio economic models model natural systems and socioeconomic systems. The outputs of the LCI phase are used directly in the bio economic models of ES to evaluate changes in the supply of ES. Then, to obtain monetary indicators of ES, we evaluate the loss of benefits through monetization techniques.

This approach is also different from those presented above because it distinguishes between "intermediate" ES and "final" ES. The "intermediate" ES correspond to the structures and functions of ecosystems that give rise to "final" ES. As for "final" ES, they are directly used or consumed by beneficiaries [27,28]. This distinction helps to avoid problems of double counting when the evaluation is undertaken. For example, counting both the value of pollinators contributing to agricultural output and the value of agricultural output would double count. That is why, in this approach, the evaluation is applied to ES directly consumed or used by one or more beneficiaries.

In addition, it is considered by evaluating ES that biodiversity and the ecosystem functions are included in this estimate. Figure 1 (Fig. 1) shows the relationship between the LCI phase, biodiversity, ecosystem functions, ES and benefits. It highlights the non-bijective relationship between these elements. In fact, the emissions or extraction of substances may affect biodiversity by altering ecological processes or/and biophysical structure of an ecosystem. When biodiversity is affected, ecosystem functions also are and might not deliver the same quantity and quality of ES. The same reasoning applies to a benefit, because it can be provided by several ES and, vice versa, one ES can provide multiple benefits. This approach is complex and requires the relationships between these elements to be characterized before any assessment, in order to specify what is being evaluated and avoid any double counting.

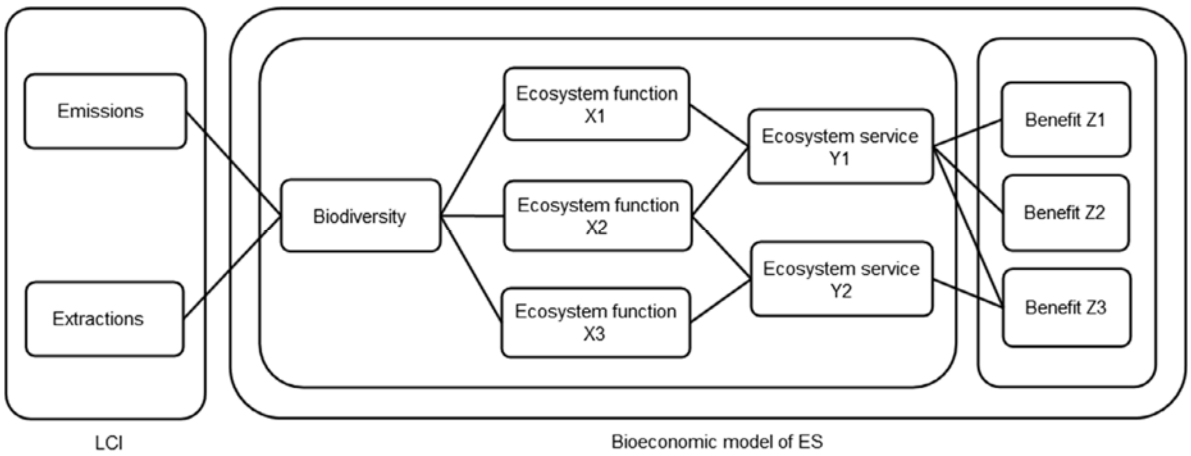

Fig. 1. Linking the Life cycle inventory to bio economic models of ES

7. Illustration of the proposal in a case study

The applicability of this approach is demonstrated by means of an industrial case study with two detergent production systems at the formulation phase. The first one is designed from a traditional process, while the other from an eco-designed one. Both are marketed by Procter & Gamble, which is one of the world leaders in this field. For the LCI output of the two products, the case study relies on data from a study on several Procter & Gamble detergents [29]. In order to illustrate the proposal, we compare the changes in the provision of surface freshwater used for drinking water in a specific situation caused by the product formulation phases of these two products. It provides a monetary indicator of changes in the provision of surface freshwater used for drinking water. It is done by means of a bio economic model of ES developed in this research study.

We suggest to relying on the nutrient enrichment caused by the formulation phase in an aquatic ecosystem. Nutrient enrichment can lead to an eutrophication phenomenon in aquatic ecosystems and is therefore related to several ES. Some biophysical effects of eutrophication have already been studied and lead to many changes in ecosystem structures and their ecological processes [30]. These changes affect many aspects of human well-being. For instance, decreasing water quality affects people in many ways: from drinking water to leisure activities and commercial fishing. Recently some studies did research on the effects of eutrophication on ES provision and the changes in the value of ES. They point out the pressing need to develop better models that are both realist, simple and allow the links between actions from nutrient releases, ES provision and economic valuation to be made [31–33].

In the context of this case study, the pathway modeled is the one between nutrient release by the formulation phases, the ecological structure and functions of the ecosystem affected by pollution, the ES of surface freshwater provision and the drinking water benefit.

Regarding the modeling, formula (3) allows changes in ES provision to be assessed by means of the $\Delta\, ES_s$ parameter. It corresponds to the difference between the water quality at *t1*, recorded $WQ_{s,t1}$ with *s* as nitrate, and the water quality at *t0*, written $WQ_{s,t0}$.

$$\Delta\, ES_s = WQ_{s,t1} - WQ_{s,t0} \tag{3.1}$$

To assess the changes in this ES provision, the concentration (C) of the substance s in the ecosystem serves as a proxy for *WQ*. Thus, the equation (3.1) can be written as:

$$\Delta\, ES_s = C_{s,t1} - C_{s,t0} \tag{3.2}$$

With $C_{s,t0}$ representing the nitrate concentration of the ecosystem. The Eaufrance platform (http://www.eaufrance.fr/index.php) is the unique access point for all information and public data on water and aquatic environments. It inventories different water parameter databases containing the nitrate concentration of many aquatic ecosystems. These data are available thanks to the French water agencies which manage water in six watersheds. To illustrate this case, it is considered that $C_{s,t0}$ is 26 mg/l of NO_3^-. $C_{s,t1}$, is the concentration of nitrate in the surface freshwater after the pollution occurred. The mass of substance *s* that reaches the surface freshwater is the difference between the mass of substance released by the formulation phase system given by MSP_s, minus the mass of substance s treated by the water treatment plant of the company: recorded MSW_s. The mass of substance s that reaches the surface freshwater is then divided by the water flow (WF) of the water course in which the mass is diluted. We can obtain this data from a measurement network managed by the offices of the French Department of Ecology in each administrative area via the platform Eaufrance.

$$C_{s,t1} = C_{s,t0} + \frac{MSP_s - MSW_s}{WF} \tag{3.3}$$

The modeling above gives the changes in *WQ*. In the next part of the modeling, the mathematical formulae calculate the changes in benefits related to substance *s*. Formula (4) is the starting point, focusing on the drinking water benefit. Thus, formula (4.1) is the difference between the drinking water benefit at *t1*, recorded $DW_{s,t1}$ and the drinking water benefit at *t0*, given by $DW_{s,t0}$.

$$\Delta B_s = DW_{s,t1} - DW_{s,t0} \tag{4.1}$$

To assess the changes in benefits, the abatement cost (AC) of substance *s* is proposed as a proxy for *DW*. Thus, the equation (4.2) can be written as:

$$\Delta B_s = AC_{s,t1} - AC_{s,t0} \tag{4.2}$$

AC is related to the water purification chains. Some of them are equipped with water purification systems such as water treatment plants and others use raw water mixing to decrease the concentration of substance *s*. In this case study, it is considered that the water purification chain is a water treatment plant. Thus, $AC_{s,t1}$ is calculated as the difference between $C_{s,t1}$ minus the concentration of a threshold T_x, multiplied by the amount of water pumped (AWP). The whole is then multiplied by the treatment cost (TC) of substance *s* (4.3). The thresholds T_x are defined by the nitrate Directive [34] based on health and environmental criteria. The nitrate Directive set three thresholds for nitrate concentration in drinking water:

- 50 mg/l, the maximum health standard relating to water intended for human consumption and the environmental standard for the quality of surface and groundwater, set at French i.e. and European level T_1;
- 40 mg/l represents the concentration for preventive measures of environmental restoration, intended to characterize the risk of exceeding the standard in the short term T_2 ;
- 25 mg/l, the freshwater concentration likely to influence the water purification chain T_3.

In this case study, we assume that T_x corresponds to T_3.

$$AC_{s,t1} = [(C_{s,t1} - T_x) * AWP] * TC_s \tag{4.3}$$

The equation of $AC_{s,t0}$ (4.4) is the same except for one parameter, the concentration of substance *s* given by $C_{s,t0}$.

$$AC_{s,t0} = [(C_{s,t0} - T_x) * AWP] * TC_s \tag{4.4}$$

Through this approach, characterization factors are obtained from different substances in g NO_3^-eq. by means of the equivalent factors developed by Wenzel et al. (2000).

Table 2. Characterization factors

Substances	Unit in g NO_3^- eq.
NO_3^-	0 ,054615098
NO_2^-	0,073730382
NH_4^+/NH_3	0,198798956

The changes in ES provision at the formulation stage of detergents are presented in a histogram. The histograms show a biophysical indicator of changes in freshwater surface provision and a monetary indicator of the loss of drinking water benefit.

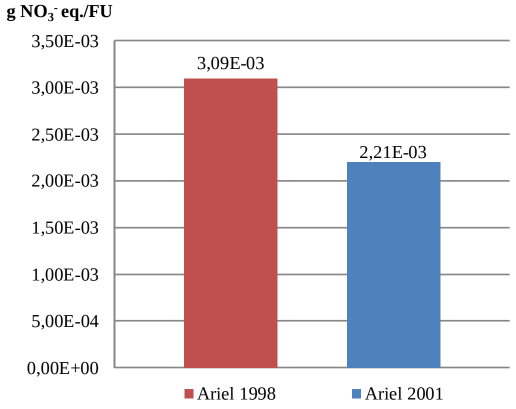

Fig. 2. Biophysical indicator of changes in surface water provision

The monetary indicator of the loss of drinking water benefit is obtained with data about the treatment cost of one unit of nitrogen from a French department of Ecology study [36].

Table 3. Weighting factors

Substances	Unit in €/g NO_3^-
NO_3^-	0,38984

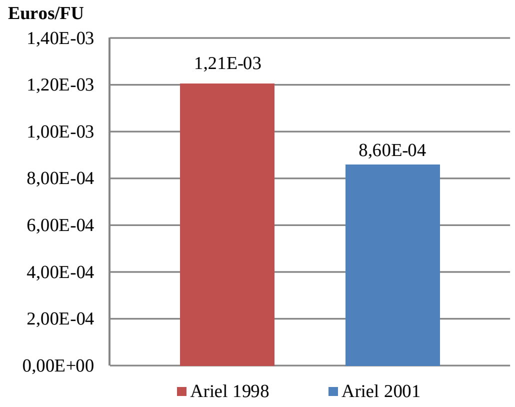

Fig. 3. Monetary indicator of the loss of drinking water benefit

8. Discussion

This research is complex and requires making choices and assumptions when modeling. The illustrative case study shows the changes in ES provision caused by the formulation phase of two detergent. This proposal enables to characterize changes in the provision of ES at a more local level in order to develop a bigger awareness and responsibility of decision makers about environmental impacts and externalities. However, in this article, only one ES is modeled, although for decision making, it would be more relevant to know the trade-offs between different ES. But it is quite a big effort to develop bio economic models for several ES. So, it is necessary to validate the relevance of the proposal in decision making in eco-design processes.

9. Conclusion

This article has developed an approach to evaluate environmental externalities by relying on LCA methodology, ES and monetary valuation. It compares some existing approaches and proposes a new approach to assess environmental externalities at situation specific or local level. Unlike the previous approaches developed, it is based on the two first phases of LCA and connects the LCI phase with bio economic models of ES. Thus, it allows monetary indicator of changes in the supply of ES to be obtained. In order to illustrate the proposal, we compare the changes in the provision of surface freshwater used for drinking water in a specific situation caused by the product formulation phases of two detergents. The first one is designed from a traditional process, while the other from an eco-designed one. Further works are required in order to develop other bio economic models of ES. Research in this field is still in its infancy. However, the need and demand for decision support tools is becoming increasingly important.

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