











# Valuing biodiversity and ecosystem services: why linking economic values with Nature?

Évaluation la biodiversité et des services écosystémiques : pourquoi associer des valeurs économiques à la Nature ?

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#### Abstract

The evaluation of ecosystems and biodiversity has become an important field of inquiry for economists. Although this development has been largely motivated by the search for arguments in favour of more ambitious conservation policies, both the methods and the meaning of the results continue to be controversial. This article aims to clarify the interests and limitations of this works, by revisiting a number of issues, such as the economic qualification of the services that human societies take from nature, the specificities of their contribution to human well-being, or the consequences of a valuation of biodiversity based on ecosystem services. We conclude with a discussion of the purposes of evaluations: improving public policies or creating new markets?

#### Résumé

L'évaluation des écosystèmes et de la biodiversité est devenue un domaine de questionnement à part entière pour les économistes. Bien que ce développement ait été largement motivé par la recherche d'arguments en faveur de politiques de conservation plus ambitieuses, les méthodes mises en œuvre et les résultats obtenus continuent à faire l'objet de controverses. Cet article vise à préciser l'intérêt et les limites de ces travaux, en revisitant un certain nombre de questions, telles que la qualification économique des services que les sociétés humaines se procurent auprès de la Nature, les spécificités de leur contribution au bien-être humain, les conséquences d'une évaluation de la biodiversité à partir des services écosystémiques. On conclut par une discussion des finalités de ces évaluations : amélioration des politiques publiques ou création de nouveaux marchés ?

Key words: biodiversity, evaluation, methods, ecosystem services, economic theory

Mots-clés : biodiversité, évaluation, méthodes, services écosystémiques, théorie économique

## 1 Introduction

"To say that we should not do valuation of ecosystems is to simply deny the reality that we already do, always have and cannot avoid doing so in the future". (Costanza et al., 1998).

"The world's ecosystems are capital assets. If properly managed, they yield a flow of vital services, including the production of goods (such as seafood and timber), life support processes (such as pollination and water purification), and life-fulfilling conditions (such as beauty and serenity). Moreover, ecosystems have value in terms of the conservation of options (such as genetic diversity for future use) (1). Unfortunately, relative to other forms of capital, ecosystems are poorly understood, scarcely monitored, and (in many cases) undergoing rapid degradation and depletion. Often the importance of ecosystem services is widely appreciated only upon their loss". (Daily et al. 2000)

Earth ecosystems provide valuable services supporting human life. Since before the development of agriculture, thousands of years ago, they have been modified and managed to satisfy human's needs and desires. It does not imply that they can or have to be economically valued, and the quantification and economic valuation of economic services remain controversial (Sagoff, 2011). Nevertheless, the exploitation of natural systems, including efforts to modify and manage them, implies to confront the tradeoffs between real and potential services, and effects upon their resiliency. More generally, innumerable choices made every day by billions of people impact ecosystems and many will result in biodiversity losses and to various extend of social costs.

The concepts and methods to value ecosystems and biodiversity have progressively emerged and their roots can be found in the core of economic theory of value (Daily et al., 2000; Gomez-Baggethum et al., 2010). The recent enthusiasm for these analyses appears to have been mostly initiated by the needs of conservationists that were looking for stronger reasons for more ambitious policies aiming at protecting the nature and biodiversity (Balmford et al., 2002). The current situation can then be characterized by a worrying gap between the perceived importance of improving our understanding of the dependence of ours economies and societies upon the maintenance of well functioning ecosystems (Daily, 1997; Costanza et al., 1997; Balmford et al., 2002; Braat and Ten brink, 2008; Reid et al., 2005, Sukhdev, 2008; TEEB, 2009) and the theoretical and practical unresolved difficulties to build consistent and reliable analysis of this dependency.

This paper proposes a rather general overview of the potential and difficulties of the economic approach of the valuation of ecosystems and biodiversity, and suggests a more personal view on a few issues. Section 2 clarifies some conceptual issues raised by these studies. Section 3 discusses technical aspects related to methods and objects through which the existing studies try to approximate the value of biodiversity and nature's services. Section 4 comes back to the significance of the results obtained by a few large-scale studies. The article concludes with a discussion of the purpose of these evaluations, mobilized both to inform public policies and to discuss the introduction of market mechanisms.

## 2 Practical and conceptual issues

Most economists involved in the biodiversity valuation debate entered this area, sometimes reluctantly, after they have been invited to do so by conservationists who hoped to find in economic analysis strong advocacy to stop biodiversity losses. And many economists were very cautious entering this debate since they knew how poorly equipped they were to build convincing and reliable arguments (Hanley et al, 1995; Nunes et van der Bergh, 2001; Hanley et Shogren, 2002...).

## 2.1 Why is biodiversity so important to human societies?

The biodiversity concern has become so widely pervasive in our societies that addressing the question of the importance of biodiversity appears somewhat outmoded. It may nevertheless remain useful to insist upon the diversity of reasons that can advocate for more ambitious conservation policies.

#### 2.1.1 Evidence

Several of these reasons are easy to understand since biodiversity or, more precisely the ecosystems that express the diversity of life, offers a large variety of goods and services that support human life: provision of food; fuel and construction materials; purification of air and water; stabilization and moderation of global climate; moderation of floods, droughts, extreme temperatures and wind forces; generation and renewal of soil fertility; maintenance of genetic resources that contribute to the variety of crops and animal breeding, medicine and other products; recreational, aesthetic and cultural benefits (MA, 2005).

Apart from these actual benefits, biodiversity plays a significant role as an insurance or a safety net in our changing world, especially for the most vulnerable human populations whose well-being depend often more directly upon productive ecosystems (Quaas et Baumgärtner, 2008). On a global scale, biodiversity must be considered in connection with major issues as for example poverty reduction (Dasgupta et Mäler, 1993), food security and fresh water availability, economic growth, conflicts over the use and ownership of resources (Baland et François, 2005), human, animal and plant health, energy and climate change<sup>1</sup>.

There are two massive evidences upon which most analysts strongly agree. Two centuries after the so-called industrial revolution and despite major changes in agriculture, manufacturing, mining, transportation, and technology, our societies remain strongly dependent on well functioning ecosystem for life support, production inputs or amenities (Daily, 1997; Diaz et al, 2006). On the other hand, there is strong evidence that human activities threaten ecosystems and biodiversity (MA, 2005; GBO-3, 2010). Biodiversity thus appears both as the source of goods and services whose degradation threatens human well-being, and something at stake in many human and social choices.

#### 2.1.2 Freedom of choice

The Millennium Assessment synthesis report (MA, 2005) highlighted the links between ecosystem services and the elements that contribute to human well-being in a now widely published and used general scheme. This scheme identifies four categories of constituents of well-being (security, basic material of good life, health, and good social relations) and what appears as a background category: the freedom of choice and action, defined as the "opportunity to be able to achieve what an individual values doing and being".

The suggested symmetry between the supporting functions, which make possible the ecosystem services to human societies, and the freedom of choice that render human being able to draw benefits from this services is the clearest lesson from this graphic. It emphasizes the statement that, without the possibility of choosing their action, the question of the value of ecosystem services has no meaning. The question of the economic value can only be addressed when there is a choice, and we will see that the nature of economic value implies, at least implicitly, the existence of alternative option of choice.

<sup>&</sup>lt;sup>1</sup> There is in fact an economic controversy on a possible ambiguous role played by the dependance of the poor populations upon forest and other common property natural resource, between safety net and poverty trap (see Delacote, 2009), but the insurance mechanism provided by well functionning ecosystems is widely regognized.

## 2.2 The value of nature and the nature of economic value

Values are norms that allow judging, individually or collectively, if something is good, beautiful, true, useful, moral, etc. Value analysis can be addressed in many conceptual frameworks that can be structured by opposing the objectivist approaches that tend to establish the basis of a universal hierarchy among things, and the subjectivist ones that relate the value of the thing on their relative desirability. The economic conception of value is often summarized in the idea that economics values things according to their utility and scarcity. Without restating what Daily et al. (2000) already clarified, several points have nevertheless to be made explicit here.

#### 2.2.1 The nature of economic values

The economic perspective is anthropocentric; only the point of view of human beings can be taken into account. This statement does not mean that only human direct interests can be considered by economic analysis. It means that only the effects that affect human psychology and can impact human well-being will be part of economic analysis.

Economic values are consequentialist. The judgment on the economic value of choices and actions is not related to their concordance to deontological principles, but only to their consequences on human well-being. Intrinsic values have no economic meaning and the things get economic value according to their contribution to human well being; this is why economic valuation is said instrumental and in most cases utilitarist. Once again, it does not mean that more sophisticated reason to conserve nature cannot be considered, but these consideration are apprehended through the filter of human preferences.

Despite recurrent attempts to establish the foundations of an objective approach of value, subjective approaches are now widely prevalent, which is obviously problematic for things for which agents do not have a clear awareness of the benefits they receive. Since human beings, especially in developed economies, are used to consider that the production processes, which supply most part of their consumption, are no longer dependants upon well functioning ecosystem, it cannot be expected that they spontaneously value biodiversity very high.

Finally, the standard approach of value is marginalist. Valuation does not aim at absolute measures, but relies on marginal rates of substitution in order to determine how much of an increase in B can compensate the utility loss due to the reduction of one unit of good A. Since most economic analysis relies on ordinal instead of cardinal approaches of utility, the meaning of valuation is less to measure than to compare. This statement puts the emphasis on the fact that economic valuation refers explicitly to individuals' preferences.

A value approach based on comparisons raises the question of its universality: Is everything comparable? Economically, comparing the contribution of things to social well-being means that, at some level; one thing can substitute another. The idea that everything can be substituted is certainly not spontaneously accepted. This intuition must be discussed because the concept of substitution refers to different levels of analysis: replacing the object, substituting the service, maintaining the well-being. The difficulty to replace technically some assets does not necessarily mean that the benefits society derives from its use or existence cannot be compensated in terms of well-being by some other elements.

#### 2.2.2 Biodiversity as a commodity?

From the perspective of an economist, biodiversity is of interest for two reasons. On the one hand, biodiversity is valuable to society: the greater the biodiversity, the better off we are, and if we lose some biodiversity, we judge ourselves to be worse off (Diaz et al., 2006). On the other hand, choices made by society have had and are continuing to have impacts on biodiversity: some of these choices, including without having had the intention or even the awareness, have adversely impacted biodiversity. Clearing land or draining wetlands for

agriculture or development, harvesting timber from primary forests, overfishing, for example, have caused biodiversity losses. These statements led the economist to consider that biodiversity is a scarce and valuable resource, and thus to some extend regarded as a commodity. G. Heal (2000) lists several reasons for this perspective: biodiversity provides or enhances ecosystem productivity, insurance, knowledge, and ecosystem services. Can biodiversity, however, be regarded as an economic good?

Biological diversity is a characteristic of sets or systems, such as populations, ecosystems or landscapes, which themselves only imperfectly satisfy the properties of rivalry and exclusion that describe the standard economic goods. The benefits that an agent receives as a result of the beauty of a landscape or the existence of a gene for the synthesis of a molecule of pharmacological interest are not reduced by the fact that another agent receives benefit from the same asset. And it would be difficult to deny access to a service such as the regulation of some pollution or local climate that certain ecosystems produce and of which humans beings benefit even when they don't have any direct interaction with them. Biodiversity appears therefore as having some properties of a public good. But most uses, whether they involve taking or are related to aesthetic and scenery, can suffer of congestion or excessive demand can create rivalry. The most appropriated economic qualification thus appears to be that of common property resources, which generally are not efficiently produced or maintained solely through the market mechanisms.

As a characteristic of ecosystems enhancing their social value, biodiversity appears both as a local and a global common. Perrings and Gadgil (2003) showed that, since biodiversity conservation provides benefits at different organisation levels, a well-designed management framework should integrate the importance of articulating local and global public benefits. But, more generally, E. Ostrom (1998) argued in favour of a polycentric governance of biodiversity based on the "law of requisite variety", which states that any regulative system needs as much variety in the actions that it exists in the system to be regulated. She agues that complex resource systems and biodiversity can successfully be maintained by complex, polycentric, multi-layered governance systems through a variety of mechanisms. A central issue is here that at each level, there must exist a sufficient congruence between the effective management rights and the awareness of the social values at stake.

#### 2.3 Biodiversity, values, and public policies

#### 2.3.1 Is Nature substitutable?

Since ecosystems and biodiversity appears as valuable resource, economist have developed a conceptual framework, which aims at measuring the Total Economic Value (TEV) of these assets. Since the seminal paper of Krutila (1967) and the large debate that was raised by its first implementations in the 1980's, this concept has been widely discussed and used. The TEV is typically defined as the integration of direct and indirect use values, option values and non-use values in a common framework or, in a simpler contrasting presentation, as the sum of use and non-use values.

The various use values, albeit they can sometimes be difficult to identify in real situation or to measure practically, don't really raise conceptual issues today. Social debate may, nevertheless occur when it comes to the social distribution of these values (Opschoor, 1998). Thought it can be of a real importance for decision-making, this issue is in no way specific of the biodiversity field, since it is a general critic of utilitarism that only refers to the sum of the interests at stake, and not to their distribution.

The case of non-use values is quite different. The controversies that emerged with their introduction in the economic analysis are not totally clarified. There are still interrogations on their economic meaning, or even on their economic nature. Among the economists who have discussed the utilitarian basis of economic analysis A. Sen proposed a dualist view of the individual, both a consumer who seeks to satisfy his preferences and a citizen who makes

judgements on objectives that may exceed his own interests. Within these interests "for others", Sen distinguishes "sympathy" which is reflected by the existence of altruistic arguments in the utility function, and "commitment" expressing ethical principles which may make the individual approve changes that reduce his utility.

The relation to nature is particular since it appears impossible to avoid it: human beings are the product of their co-evolution with other living beings, irrespectively to the progress of human autonomy, human life and survival remain dependent on ecosystems functioning. Some works (Turner, 1993; Ekins et al, 2003; Neumayer, 2010 among many others) have explored the hypothesis that there is a critical level below which a decrease of "natural capital" could be replaced by additional of manufactured capital or human, but instead resulted in a decrease in their effectiveness. If such a threshold exists, then, as suggested by D. Pearce (2007), the economic analysis of biodiversity is relevant only insofar as this limit is not crossed. When an option of the choice may lead to cross it, the question of substituting Nature or even nature's services becomes too difficult to handle and, beyond non-use values, confront the limits of our scientific understanding of our biophysical dependence (Daily et al, 2009), and possibly fundamental ethical issues (Maris, 2010).

#### 2.3.2 Biodiversity as a merit good?

This kind of distinction can relate to the opposition between instrumental, including non-use values and intrinsic values. Turner et al. (2003) provided a framework on the foundation of the value of Nature that goes far beyond the usual economic categories and the sole interests of human beings. They especially distinguish anthropocentric values and non-anthropocentric values, related to the interests that biological diversity presents for other species and the ecosystems themselves. Is the utility for human beings the only or even the main reason for conserving nature? Answering this question would go far beyond the scope of this paper. Many authors have contributed to this debate, and economics is certainly not the best conceptual framework to explore all its dimensions.

A possible road to introduce the issue is to oppose the case when human point of view remains of interest and those when it is not acceptable to rely on human judgement. In the first case, the point is to define deontological principles that will possibly allow to make practical choices and avoid too many decisions to become tragic or "awkward choices" (see Hanley and Shogren, 2002; or in a different perspective Sagoff, 2004; 2008). In other cases it seems necessary to consider more general ethical frameworks that go beyond human sole interests, such as the moral right to existence of any life form (for a recent synthesis see Maris, 2010).

Economic analysis is not comfortable with the case when the preferences of individuals are not a usable base for decision making, which refers to what economists qualify as "paternalism". There are strong arguments that, when it comes to choices involving biodiversity, individuals' preferences are not adequate for efficient decision-making. The understanding of the complex biophysical relations and regulation that drives biodiversity is clearly not appropriately rendered in the preferences of economic agents or political citizens. This inaccuracy, which should diminish as people becomes increasingly aware of these questions, leads us to pay special attention to the notion of "merit goods" (Musgrave, 1987) for which an evaluation resulting from the agents' preferences cannot be used directly to justify collective choices. The merit goods dimension of biodiversity can be explained by the agents' lack of familiarity with the "goods" and by the difficulty of explaining the ways in which they are "useful" to them. Therefore, the appropriate level of protection or conservation implies the intervention of an authority, not only due to the public good characteristics of several elements that contribute to the value of biodiversity and the ecosystem services, but also due to an incomplete or biased perception of that value, related to the indirect and incomplete perception of the services provided<sup>2</sup>.

Confronted with this difficulty, the idea of constructing a measurement of the goods to be produced or preserved and to choose it commensurate with the other economic goods could however form a useful step towards taking these goods into account in decision making and a practical means of socially managing their production. This statement addresses nevertheless a challenge both to economic valuation methods that remain largely dependant upon observable costs or behaviours, and to the choice of the objects that will materialize the concept of biological diversity for the purpose of evaluation.

## 3 Material and methods

Although facing severe conceptual issue, the valuation of biodiversity has become a widely developed area of research, and we have to come back to the methods and techniques that economists have build in attempts to circumvent these obstacles.

#### 3.1 On the valuation techniques

During the last decades, a large scientific and administrative literature has repeatedly reviewed the available methods that enable to produce practical measurements of ecosystem services values and it must be questioned if there is still anything new to be said on valuation methods. The first point is to restate that, despite its misleading name, the so-called total economic value does not pretend at estimating an absolute value of ecosystems, but its purpose is to allow the addition of the multiple economic reasons (Balmford et al., 2002) that underlie the social values of ecosystem services or threatened ecosystems. The valuation consists of comparing different situations and provides a measurement of changes in human well-being between these situations. For this purpose, evaluation must be built from observations, which raises the question of what is observable. Ultimately, these are the individuals' preferences that must be made observable, which may involve helping individuals to build them.

#### 3.1.1 What is observable?

Aside from the recurrent debate on the conceptual and methodological issues raised by the economic valuation of non-market goods, valuations methods have to deal with empirical data. Following textbooks, these techniques can be introduced according to the nature of the data used: effective technical costs, observable behaviours and choices, statements when confronted to questionnaires. These data have then to be treated in order to obtain price-equivalent, under the assumption that individuals' preferences, which are admittedly the foundation of economic value, are adequately summarized in willingness to pay or to accept compensations.

Table 1 gives a conventional presentation of valuation techniques that opposes vertically methods based on observations directly related to natural assets and methods which use observations on goods or activities, which are associated as complementary to the natural assets (like travel costs to scenery viewing). Horizontally, it contrasts methods in which preferences are revealed through choices and behaviours, and methods in which individuals

<sup>&</sup>lt;sup>2</sup> Merit goods, for which agents are not able to express reasoned preferences, are sometimes related to public goods, for which the problem is not that preferences are biased but that there is no incentive to translate preferences into behaviour ("free riding" issue). The two categories often overlap (Fiorito and Kollintzas, 2004) and this is the case for biodiversity. The agents' relationship with the public authority can therefore be considered to be a delegation of choice. Concrete public decisions often rely on expert knowledge, and in the best case on the benevolence of the policy-makers. As a matter of fact, the expertise process related to biodiversity is mostly managed through specialised NGO.

are invited to state their preferences in the frame of *ad hoc* questionnaire that lead them to express directly willingness-to-pay or indirectly choices between alternatives whose cost is one of the arguments.

	Revealed preferences	Stated preferences	
Direct methods	Monetary valuation at market prices Avoided costs, productivity effects Costs of restoration, replacement	Contingent valuations	
Indirect methods	Prevention or protection expenditures Travel costs Hedonistic prices	Contingent ranking Comparison by pairs Joint analysis: choice experiment, choice modelling	

Table 1. A diversity of valuation methods for non-market goods and services

Source: adapted from Chevassus-au-Louis et al., 2009.

The valuation of the benefits of biodiversity conservation raises two main problems. The first one has already been addressed when we proposed to consider biodiversity as a merit good. Many people are poorly informed about the meaning and issues related to biodiversity. Their preferences do not constitute a satisfactory basis for evaluating biodiversity, complicating the use of stated preferences techniques, because information must be given to the respondents, while avoiding to directly influencing their preferences. Hanley et al. (1995) found that willingness to pay for biodiversity protection increases with the level of information provided. The second point is often described by the fact that individuals' preferences for biodiversity protection may be lexicographic<sup>3</sup> rather than utilitarist. If this were true for many individual, the cost-benefit analysis would become invalidated as a guide for decision making. Despite evidence that below a certain limit (unfortunately difficult to determine<sup>4</sup>) Nature's services become less substitutable, multiple observations have shown that individuals have the ability to make trade-offs between biodiversity and other assets when faced with situations of choice.

#### 3.1.2 Stated preferences: helping individuals to build their preferences

When individuals are faced with choices involving numerous variables, the standard assumption of pre-existing and stable preferences for all possible situations may be totally unrealistic. The *constructed preference hypothesis* suggests that preferences do not exist prior to choice situation but rather are created at the moment of choice (Slovic, 1995). This theory predicts that preferences will be malleable to the choice environment, such as framing or anchoring effects. On the other hand, the discovered preference hypothesis (Plott 1996) considers that preferences exist but that they need to be uncovered through a process involving practice, repetition and experience. This process will result in stable preferences that are consistent with economists' standard beliefs. It is more and more considered that market behaviour does not reveal underlying *true* preferences but rather *context-dependent* preferences. The growing evidence of a design effect in stated preference methods is consistent with this theoretical framework.

Regarding stated preference methods there is indeed a long lasting debate on the designdependence of the results. Since these techniques are not based on observable facts, many questions arise, many of them are related to the so-called bias affecting contingent valuation

<sup>&</sup>lt;sup>3</sup> Preferences are said "lexicographic" when individuals do not consider the possibility of compensating the loss of biodiversity through an increase in the availability of another good or service. As in a dictionnary, the value of the first item is what it is, and can not be changed by varying the following.

<sup>&</sup>lt;sup>4</sup> The existence and practical identification of this limit has led to a considerable debate on the notion of "critical natural capital" which is at the core of the opposition between strong and weak conceptions of sustainability (Ekins et al., 2003; Neumayer, 2010).

(CV) method, namely the question of embeddedness which means that the statements of the respondents are not really related to the precise hypothetical scenario described, but to some wider inclusive conceptions of what is at danger or what their willingness-to-pay will allow to protect.

In the last decade, growing interest has gone to procedures that could help individuals construct reasoned preferences, namely through collective and deliberative process (Wilson et Howarth, 2002; Alvarez-Farizo et Hanley, 2006; Spash, 2007). As long as the stated preferences remain the most elaborated methods to address the social value of ecosystem, any methodological evolution that may lead to improve the informational basis that supports these approaches has to be studied seriously.

The elicitation format is at stake also. After the *NOAA Panel* identified the dichotomous choice as the only methodologically acceptable elicitation format for contingent valuation, this technique appears limiting since it implied larger sample sizes and appeared poorly able to handle scenario involving multi-dimensional changes. For both these reasons, valuation analysts were increasingly interested in choice modelling techniques (Hanley et al., 2001). Moreover, choice modelling appeared to show a better sensitivity to scope since they give larger values than contingent valuation for the overall bundle and smaller values for individual components (Foster and Mourato, 2003). More generally, it is widely acknowledge that stated preference methods results are design-dependant, especially when related to non-use values.

#### 3.2 On which basis can biodiversity be valued?

Biodiversity is not an object, but a characteristic of sets of objects, such as ecosystems at various scales. The question of what can be valued is then not trivial and the literature brings many different answers, from theoretical approaches to more observable objects. Since the Millennium Ecosystem Assessment, valuing biodiversity from ecosystem services has become a kind of standard approach, but several points remain questionable.

#### 3.2.1 Theoretical objects

Since markets and even human preferences can be inadequate for decision-making, there is a need for alternative information. Biodiversity index, build by scientists to compare situations, appears mostly inappropriate for social management. Nevertheless, many analyses, considering the absence of relevant information on individual preferences and collective interests, are trying to use them to build usable indexes.

Weitzman (1992) proposed to value biodiversity from a measure of dissimilarity between taxonomic units that he calls "species" and which can stand for any reproductively isolated group. The index is build from dissimilarity between two units and can be recursively extended to any set of "species". In "The Noah Ark problem", Weitzman (1998) build on this index a cost-effective approach which results can be useful 'as a criterion to rank project. Metrick and Weitzman (1998) applied this ranking criterion to assess the cost-efficiency of the public expenditures related to the Endangered Species Act. They created proxies for the main variables and found that the most suitable dependent variable was the expenses for each action and the most significant explanatory variable was the size of the species.<sup>5</sup> As noted by Weikard (2002), the practical calculation of the index requires impractical amounts of information, but the conceptual model could be applied to ecosystem functions. Despite some provocative titles (Christie et al., 2006), empirical measures of the value of biodiversity and ecosystems are in fact on concrete objects and varied enough so that the choice is often motivated by the purpose of evaluation.

<sup>&</sup>lt;sup>5</sup> This unexpected result was in fact consistent with recent knowledge on the biological traits of species.

#### 3.2.2 Empirical objects

Because the assessment of ecosystems and biodiversity has become an area of collaboration between scientific disciplines, which have to communicate about empirical objects, or simply because many studies were initiated by social demand or toward policy-making, a large body of the scientific literature on biodiversity valuation focuses on biophysical objects. These objects can be very different and were generally chosen in order to represent one or another aspect of biodiversity. These objects can be briefly enumerated and a few representative studies mentioned: species (Richardson et Loomis, 2009); genes (Goeschl et Swanson, 2007; Sarr et al., 2008), ecosystems and habitats (Costanza et al., 1997; Amigues et al., 2002), ecological functions (Allen et Loomis, 2006; Maltby, 2009); landscapes (Bonnieux et Legoffe, 1997; Swanwick et al., 2007).

Whatever the object that existing analysis pretend valuing, it must be clear that valuations are related to changes in well-being associated with modifications in the characteristic of these objects and, more precisely, in their availability as a source of goods and services. This is certainly why since early publications, such as Westman (1977), the notion of services was more and more used to introduce the beneficial uses of ecosystems in the utilitarian framework.

#### 3.2.3 Ecosystems services

Ecosystem services are usually defined as the benefits people obtain from ecosystems. These include provisioning services, such as food, clean water or raw material; regulating services, such as regulation of floods, drought, and in some cases disease; cultural services, such as recreational, spiritual, and other nonmaterial benefits; and supporting services, such as soil formation and nutrient cycling. This notion has become especially widespread with the Millennium Ecosystem Assessment (MA, 2003; 2005), which stated authoritatively that ecosystems were experiencing serious degradation in regard to their capability of providing services, and, at the same time, the demand for ecosystem services was rapidly increasing as populations and standards of living increase. The Millennium event stated that, if ecosystem services were becoming increasingly scarce, it was "partially due to the lack of valuation because it is impossible to manage what we do not value" (MA, 2005), but it might be considered as somewhat self-serving.

Although an important moment, the Millennium Ecosystem Assessment did not initiated the current enthusiasm in ecosystem services. The idea has a long history that Mooney and Ehrlich (1997) initiate with the 1970 Study of Critical Environmental Problems (SCEP, 1970), which first used the term '*environmental services*'. The list of identifiable services remained very variable with the authors' choice. Westman (1977) was among the first to explicitly refer to the value of '*nature's services*', and finally Ehrlich and others used the term 'ecosystem services' in the early 1980s (Mooney and Ehrlich, 1997).

Considering the relationship of ecosystem to societies as "services" has become a meaningful qualification, clearly inspired from the services produced by humans. Comparing the issues related to maintaining ecosystems with other dynamics related to human activities might indeed be a dangerous concept, which, focusing on the sole final services that benefit humans, can obscure the complexity of ecosystems functioning (Ghazoul, 2007). Moreover, a major difference between social and ecosystem services lies in intentionality. Unlike human productions and business, ecosystems do not aim at fitting human needs. This is the responsibility of human societies to adapt their organization to ecosystem services".

#### 3.2.4 Quantification and valuation of ecosystem services

There is a strong on-going debate on the possibility of quantifying and evaluating ecosystem services. Ecosystem services are the conditions and processes through which natural ecosystems and the species that make them up allow and sustain human life. There are thus

many reasons to consider that ecosystems have both utilitarian and intrinsic values. The limited substitutability of Nature addresses difficult challenges for any valuation attempt and, in fact, for the quantification of these services.

Following Gomez-Baggethum et al. (2010), it can be said that the monetary valuation was probably the main cornerstone that led in the 1980's to cleave the economic approach of society-nature interaction between:

- environmental economics that favour an extension of monetary valuation techniques to non-market natural assets,
- ecological economists that addressed the substitutability of natural capital and preference-based valuation as controversial issues.

This contrast echoes the opposition between the weak and strong conceptions of sustainability (Baumgartner, 2010). The former assumes a large substitutability that allows a monetary valuation of non-market natural assets, while the latter focuses on the importance of preserving a "critical natural capital" whose measurement remains nevertheless a problem (Ekins et al., 2003). This enduring opposition is known as the "incommensurability debate" and often considered as the single most controversial issue in CBA (see Aldred, 2006). Indeed, if no substitution is possible, the very principle of economic valuation fails and the meaning of any valuation no longer holds, except on the basis of the subjective judgments of the subjects involved.

Recently, M. Sagoff (2011) relies on the analysis of F. Hayek on markets as information processes to ague that, in the absence of markets, the subjective nature of economic value renders the perception of ecosystem value by different economic agents incommensurable. The economists will recognize the old issue of the impossibility of aggregating agents' preferences. Although, it appears somewhat specific here since there are limited possibilities to quantify physically ecosystem services and human preferences might be the only way to get a measurement. In other words, for such heterogeneous things as ecosystem services, quantification and valuation are not really distinct steps. And the alleged impossibility of the former implies the impossibility of the latter. This paper offers a meaningful enlightening on a recurring controversy, but, like many others, it does not allow to draw useful lessons for practical decision-making.

#### 3.3 Ecosystems and human well-being

The abundance or the quality of natural resources does not constitute a guarantee of improved well-being. The economic literature shows a series of works on the "resource curse", that is to say, the paradox that natural resource-rich countries tend to have less economic growth and worse development outcome than less well endowed countries. The contribution of ecosystems and biodiversity to human well-being is therefore contingent on the ability of societies to value these assets. The link between ecosystems and human well-being therefore depends on the reality or, at least, the potential services that societies know how to get from them, but also on their capability to do without, and to rely on alternative resources (including human creativity).

This complex relation can be represented in three steps:

- identification of ecosystem ecological functions
- description of the beneficial uses that societies get from these ecosystems
- analysis of the economic value of these services, according to available alternatives.

Several recent publications follow the view developed by Haines-Young and Potschin (2010), which introduces a supplementary step. They split the concept of beneficial use in, firstly, a biophysical description of the service and, secondly, an analysis of how that service benefits to humans. This benefit is finally valued in economic terms.

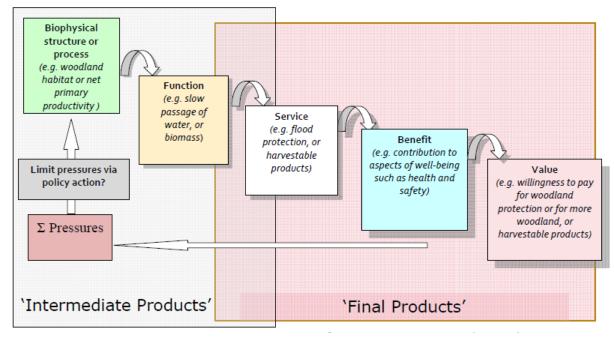


Figure 1. From ecosystem structure and function to ecosystem services, benefit and value

Source : adapted from Haines-Young and Potschin (2010)

The important point here is to understand and make readers, users or policy-makers that there is no direct link or proportionality between the bio-physical phenomenon and the social value. Between these two level of observation, conceptualising and analysis, several parameters strongly influence the final results, among which: available technologies, cultural preferences and, as it was emphasized above, freedom of choice, which determines in fact the final well-being that humans will be able, or not, to draw from the actual ecosystems.

## 4 Results

As it was emphasised above, applying the general CBA framework to biodiversity issues raises so many conceptual and technical difficulties that despite early calls (Krutilla, 1967; Helliwell, 1969; Westman, 1977; Randall, 1988) little practical works were achieved before the last two decades. On the contrary, many voices spoke against the possibility or the legitimacy of doing so (Ehrenfeld, 1988; Norton, 1988; Hanley et al., 1995). Nunes and van der Bergh (2001) tried to separate the meaningful achievable analysis from less credible attempts, and Hanley and Shogren (2002) still considered the idea of using CBA for nature conservation arbitrage as "awkward choice" situations. Despite several important clarifications, the reliability and the relevance of the economic valuation ecosystems and biodiversity remains strongly debated.

It is not possible to provide here a review of the diversity of results obtained by the hundreds of studies that have evaluated one or another aspect of biodiversity. This section is thus organized into three points: the economics of endangered species, the valuation of ecosystem services and, finally, a brief discussion of the distinction between general and remarkable or unique biodiversity. We conclude with some dangers associated with the assimilation of the value of biodiversity with that of ecosystem services.

## 4.1 The economics of endangered species

There is a large body of studies on the value of species, whose existence is mainly due to the U.S. Endangered Species Act (1973), which led to this type of work being done from the point of view of budgetary rationalisation (see Brown and Shogren, 1998; Metrick and Weitzman, 1998). The contingent evaluation method is well suited to the evaluation of willingness-to-pay (WTP) for the conservation of endangered species, especially for charismatic species. A lot can thus be found in the scientific journals, with a limitation that could be described as "publication bias": each study must be presented in a limited format that leads authors and referees to put more interest on the methodological issues rather than the numerical results. Thus, the results are generally presented in the form of an econometric regression whose explained variable is a household's willingness-to-pay (usually yearly). The concerned population, which has to be estimated in order to calculate the social WTP, to be compared to the policy objective (to preserve the species in the world, the area, or a given specific surface), is in most case not even mentioned.

Most studies concern charismatic or emblematic species. It should be stressed that this concept does not only depend on ecological aspects but also integrates the socio-cultural context and may thus vary according to the places and the generations. The change of "status" of the lynx, the bear or the wolf, formerly driven out, and today "made into heritage", is undoubtedly not entirely due to them becoming rare, which is already old news, and this phenomenon illustrates this well. By limiting our interest to modern times and to the Western cultural context, we gathered in table 2 a number of studies related to threatened vertebrate populations, for which the willingness-to-pay per household and per year was estimated for various protection measures. Most of this data resulted from a meta-analysis by Loomis and White (1996) relating to 22 studies carried out between 1983 and 1993 (here only the average values in current currency is given).

Group	Species	Place	WTP (\$)	Reference
	Rhinoceros	UK for Namibia	54-65	OCDE, 2001
Mammals	Wolf	Sweden	126	Loomis et White, 1996
	Grizzly bear	USA	46	ld
	Sea otter	USA	29	ld
	Grey whale	USA	26	ld
	Bighorn Sheep	USA	21	ld
	Caribou	Canada	14-98	Anielski et Wilson, 2005
	Northern spotted owl	USA	70	Loomis et White, 1996
Birds	Whooping cranes	USA	35	ld
	Red cockaded woodpecker	USA	13	ld
	Bald eagles	USA	24	ld
Reptiles	Sea turtle	USA	13	Loomis et White, 1996
Fishes	Pacific salmon	USA	63	Loomis et White, 1996
	Cutthroat trout	USA	13	ld
	Atlantic salmon	USA	8	ld
	Squawfish	USA	8	ld
	Stripped shiner	USA	6	ld

Willingness-to-pay for diverse charismatic vertebrate species (in dollars per household and per year)

Loomis and White (1996) proposed a generalisation of this data, through a predictive multiple regression model with a basic amount of 11 dollars per year for the residents of the site, to which it is necessary to add 47 dollars if the species is a mammal, 33 dollars if it is a bird, 23 dollars if the person is a visitor and not a resident (following a proximity paradox which states

that residents value heritage assets less than visitors) and 42 dollars if a single payment is proposed (which in fact corresponds to a reduction in the total value). Brown and Shogren (1998) showed that this model resulted in a willingness to pay which, if extended to all American households, would amount to devoting 1% GDP to protect 2% of the threatened species; they consider this result excessive.

Richardson and Loomis (2009) updated the results obtained by Loomis and White (1996) by adding the work published after 1995 in new meta-analysis of the contingent evaluations related to the TEV of rare and endangered species. The two groups of studies are treated separately and the recent studies obtain in general higher willingnesses-to-pay, for which the significant explanatory variables are: change of size of the populations, the type of species, whether it belongs to the "charismatic megafauna", the existence of non-use values; but, also, the year of the study, the type of subject questioned, the survey method, the rate of answers, and the frequency of the payments are also significant variables.

These methodological refinements do not eliminate the problems involved in the use of stated preferences methods, in particular because of the importance of variables related to the design or the conditions of carrying out the study, beside variables related to the evaluated asset. In addition, the use of these values often brings a problem of disproportion, even of disconnection between the location and the sometimes limited character of the territory to be protected, because of the endangered character of these species, and the size and location of the population considered as likely to pay, because of the charismatic character of these species and the importance of their option value or non-use value.

Despite the large number of available studies, and the fact that many among them have been achieved in a cost-benefit analysis perspective (as required for the ESA), they do not bring usable information for decision making unless there are explicitly aimed at valuing action programs (see Chevassus-au-Louis et al., 2009). The same comment applies to the less numerous or empirically detailed studies related to genes or even habitats.

#### 4.2 The economics of ecosystem services

The concept of ecosystem services has obtained an increasing attention as a mean to communicate about the societal dependence on ecological life support systems (Daily, 1997). A set of studies framed the beneficial use of ecosystems functions as services with the aim of increasing public awareness and policy-makers interests in biodiversity conservation (Gomez-Baggethum et al., 2010).

The term "ecosystem services" refers to a set of benefits that fall into three distinct economic categories (Barbier, 2007): (i) "goods" (products obtained from ecosystems for direct consumption or as inputs for industry, such as resource harvests, and genetic material); (ii) "services" (recreational and tourism benefits or certain ecological regulatory functions, such as water purification, climate or pollution regulation, erosion control); and (iii) cultural benefits (scientific knowledge, spiritual and religious feelings, heritage...). Early studies had to elaborate ad hoc lists of the services to be valued, according to what appeared to be mainly at stake in the study areas, mixing sometimes ambiguously services for human and ecological functions

Costanza et al (1997) have established a general framework in their attempt to evaluate the world's ecosystems. They identified and proposed estimates for 17 categories of services, for all terrestrial and marine environments. The value of just the coastal environments, including estuaries, coastal wetlands, plant communities and algae fields, coral reefs and continental shelves, represent 43% of the total, even though they only cover 6.3% of the surface of the globe. This weight seems to be related to the role that these environments play in the regulation of nutrient cycles, both terrestrial and marine, whose monetarisation seems however to be particularly tricky.

The final results were provocative enough to offer them the recognition they received: the value of services rendered annually was estimated between one and three times the value of the world gross product, due for a large part to coastal and littoral ecosystems. Among the many critics this work received, one was related to the meaning of valuing assets monetarily at a higher level than the global wealth, which assumed implicitly a conception of the wealth that go far beyond income.

Extending the study of Costanza et al. to 23 "functions" (regulation, habitat, goods and services, information) De Groot et al. (2002) gives the value ranges for all the world ecosystems. Without considering here all these figures, it must be mentioned that they can take values ranging from a few dollars to tens, hundreds and often several thousands of dollars per hectare and per year. The importance of the variations can be explained mainly by variations in the quality of the ecosystems and variations in the intensity of the uses, but also by the evaluation method because the different techniques do not capture the same attributes.

The *Millennium Ecosystem Assessment* (MA) established of a conceptual framework for documenting, analysing, and understanding social–ecological systems, which has had wide influence in the policy and scientific communities (MA, 2005). The process resulted in a list of 22 ecosystem services, organized in four major categories: the provisioning services, the regulating services, the cultural services and the so-called support services, which are in fact interactions within and between ecosystems, that do not directly contribute to human well-being, but maintain the possibility of the other services categories.

Despite the fact that the MEA remains the reference framework, the definition and the classification of the ecosystem services remains an open and discussed question (Boyd and Banzhaf, 2007; Wallace, 2007; Costanza, 2008; Fisher et al., 2009). Among the difficulties, the following must be mentioned: the mixed public goods character (public-private); the difficulties in understanding the spatial and temporal dynamics, the "joint production" character of several services by the same ecosystem; the complexity of the interactions between the structures, functions and services; the fact that the agents only identify as services those from which they benefit (Boyd and Banzhaf, 2007).

*The Economics of Ecosystems and Biodiversity* (TEEB) study<sup>6</sup> aimed, for its scientific part (TEEB-D0), to build a synthesis of "the latest ecological and economic knowledge to structure the evaluation of ecosystem services under different scenarios, and to recommend appropriate valuation methodologies for different contexts. It also aims to examine the global economic costs of biodiversity loss and the costs and benefits of actions to reduce these losses" (TEEB website).

TEEB proposes a typology of 22 ecosystem services, defined as "the direct and indirect contributions of ecosystems to human well-being". Relatively to the MA definition, it introduces a distinction between services and benefits in order to explicit that services can benefit people in multiple and indirect ways, and it omits supporting services such as nutrient cycling and food-chain dynamics, which are seen as ecological processes. A "habitat" service has been identified as a separate category to highlight the importance of ecosystems to provide habitat for migratory species (nursery service) and gene-pool (namely for commercial species).

It would be quite presumptuous to pretend give a critical point of view on the many outputs of this encompassing study. But we can come back in a few words on the key messages of the large synthesis related to the Ecologic and Economics Foundations (de Groot et al., 2010). TEEB puts clear emphasis on: the importance of linking biophysical aspects of ecosystems with human benefits through the notion of ecosystem services for assessing the trade-offs (ecological, socio-cultural, economic and monetary) involved in the loss of ecosystems and

<sup>&</sup>lt;sup>6</sup> The TEEB study (2007-2011) is hosted by UNEP with financial support from the European Commission, Germany, the United Kingdom, Netherlands, Norway, Sweden and Japan.

biodiversity ; the need to make ecosystem assessment spatially and temporally explicit at scales meaningful for policy-making since both ecological functioning and economic values are context, space and time dependant; the necessity of evaluating within the context of contrasting scenarios since both the values of ecosystem services and the costs of actions are best measured as a function of changes between alternative options; be attentive to include in the assessment the total bundle of ecosystem services provided by different conversion and management options; staying aware of the cost side of the equation, as focus on benefits only ignores important societal costs like missed opportunities; integrate an analysis of risks and uncertainties, acknowledging the limitations of knowledge on the impacts of human actions on ecosystems and their services and on their importance to human well-being.

Apart from the very last words that sacrifice for the ideology of transparency, assuming it is both feasible and appropriate, these recommendations appear of real interest when compared to the content of numerous existing studies. In a pure economist perspective, the determination of the biophysical basis of the services is not generally required. But it is here of importance, once acknowledged that some of the services might be poorly perceived or understood by individuals. The integration of the biophysical basis plays as a safety belt that maintains the analyst on her way in order not to shorten her checklist.

## 4.3 General and unique biodiversities

In conservation policies, the distinction between general and unique biodiversity is widely used to justify specific protection measures for unique or extraordinary elements of ecosystems. This distinction is not independent of human judgement: what is exceptional nature? Beautiful landscape, endangered species, remnants of disappearing ecosystems?

In a large meta-analysis of the valuations of wetlands (Brander et al., 2006) an uncommon result can be observed: the mean value of the "biodiversity" characteristic (whatever it may mean) was more than thousand times its median value (17000\$ to some 15\$/ha.year). The simplest explanation is that among the panel of studies used for the meta-analysis, some were related to exceptional ecosystems with unique biological diversity and were then highly valued by the individuals, when other sites were of non-remarkable diversity and poorly valued for that purpose. Assuming a majority of sites with limited value for diversity, and a significant minority with very high diversity value, the statistical results is no longer amazing. And it may be the better way to oppose scientifically general and unique biodiversities.

Valuing unique assets is always a difficult task, and it gets worse when the asset is considered a heritage. The analyst is then faced with a series of additional challenges: how to reason substitutability, how to handle the social-ecological dynamics of the asset, which confidence give to the non-use values?

The economic concept of substitutability goes far beyond the technical notion of replacing an object by an equivalent object, if not in the form, at least in its function. This approach would not in fact be satisfactory for unique assets (which object could possibly replace the Mona Lisa if it happened to be destroyed ? A copy ?). The economic substitution must be understood as a possibility of compensation in terms of final services, or even of well-being. The destruction of the Mona Lisa would be a great misfortune, but it seems unlikely that it would cause the end of the world, not even of the world as we know it. For each of us, even this great loss would probably be compensated in one way or another. It would be quite different if we were talking of the destruction of all existing artwork. In that case, the world as we know it would be radically and permanently changed, and there is probably no way to evaluate this change.

Ecosystem dynamics is a central issue for an adequate analysis of the social stakes related to biodiversity conservation. In economic valuations, time is usually taken into account through "discounting", which allows effects occurring at different times in the future to be compared by converting each future euro into a common currency of equivalent present euro

by multiplying it by a discount factor<sup>7</sup>. The choice of the discount rate is of particular importance for projects involving long time horizon because in such situations even tiny changes in the discount rate can drastically modify the results of the valuation. These questions have become an especially acute issue in the economics of climate change, but ecosystems and biodiversity might address at least as complicated questions. In a few words (see Chevassus-au-Louis et al. (2009) for an extensive presentation):

- there are strong arguments in favour of a lower discounting rate for natural assets than for manufactured goods of at least 1%, due to different evolution of their anticipated future prices,
- the uniqueness of these assets creates option values that must be added to their preservation value,
- if these assets are seen as irreplaceable (it's a matter of judgement) then they should be evaluated as exhaustible resources ("Hotelling rule").

Evaluating unique assets gives logically a larger weight to non-use values than for more common ecosystems or habitats. Unfortunately, despite their indisputable importance to integrate consideration less directly related to usefulness in the evaluation, non-use values raise many problems, both conceptual (are they economic values?) and methodological (how to get reliable information on their "importance", how to define the population concerned; etc.). It can then not be expected that evaluations of unique biodiversity assets will produce robust results. Stated preferences method allow to get results, but most analysts agree that these results must be replaced in a deliberative framework that allows all the issues and stakeholders to express their views.

In the report of the French Centre d'Analyse Stratégique that aimed at producing "reference values" for public socio-economic assessment of the impact of human settlements on ecosystems (Chevassus-au-Louis, 2009), the latter considerations led to skip the case of "remarkable" biodiversity, considering furthermore that "reference values" for unique assets would be a contradiction in the terms.

#### 4.4 Valuing ecosystem services *or* biological diversity

Recently, Norgaard (2010) expressed his concern that assimilating nature's diversity to a provider of services, although an "eye-opening metaphor", may become a "blinder" of the complexity of the ecological processes that render these services available to human beings. This concern encompasses several of the above debate : intrinsic vs. utilitarian values, bias related to incomplete information and understanding of ecosystem functioning, etc. But this concern can also be interpreted as reflecting the existence of very real differences between the categories underlying concepts biodiversity and ecosystem services. This separation is evident in the meta-analysis on the value of wetlands mentioned above (Brander et al., 2006) in which the "biodiversity" appears as a particular argument among other ecosystem services. In the same way, when Christie et al. (2006) pretend "valuing the diversity of biodiversity", it makes clear that, in this particular case, the distinction was problematic.

Although it is repeatedly stated that diversity is a characteristic of ecosystems that increases their ability to provide services, this assertion is true only in general, and in each case, the precise relationship between biodiversity and ecosystem services must be analyzed. We can not ignore situations in which an increase in diversity can result in degradation of servces. It is the case of certain human diseases whose prevalence is reinforced by the proximity of ecosystems such as forests or wetlands. And productive activities that operate artificialized environments such as agriculture (Zhang *et al.*, 2007), forestry or aquaculture, although their excesses have led to massive losses of diversity, must continually find equilibria that involve

<sup>&</sup>lt;sup>7</sup> For readers not familiar with economics, the discount factor is generally build like a simple composite interest calculus, based on an annual discount rate.

a minimum level of control against over-diversification of farmed environments (weeds, pests, etc.). More generally, it must be accepted that the dependence of ecosystem services upon biodiversity varies widely with the nature of the service.

That is the conclusion that Braat and Ten Brink (2008) illustrated in Figure 2 which shows that if the regulating services are fairly systematically promoted by diversity, provision services instead pass through a maximum when diversity is managed appropriately. The case of cultural services has led the authors to distinguish between scientific and spiritual services that are maximal in wild nature, and recreation or tourism which benefit from certain facilities

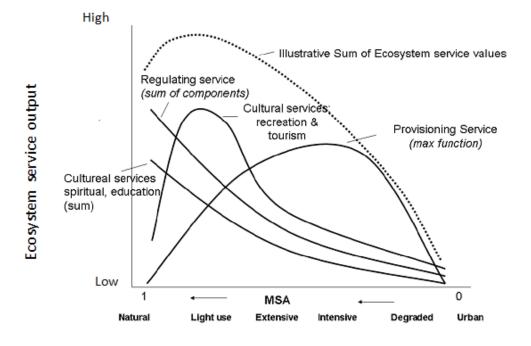


Figure 2. A general perspective on ecosystem services and human transformation of nature

Land use intensity gradient

Source : Braat et Ten Brink, 2008

## 5 Discussion

"The total value of biodiversity is infinite so having a debate about what is the total value of nature is actually pointless because we can't exist without it." (Robert Scholes, ecologist)

The legitimacy of evaluating biodiversity remains at stake. It can be suspected that behind these criticisms, there is some misunderstanding of what economic evaluation really means. The issue is not to put an economic value on nature, which would indeed be pointless, but to translate the value of losses from the destruction of some ecosystems in terms that allow comparison with other societal issues. Scholes assertion is actually based on confusion between two different aggregation issues: the economic value of the whole biosphere, which is obviously a non-sense, and the sum of all the economic reason to conserve or preserve ecosystems. And even this more limiter assessment can not be done in absolute terms, bat

has to be related to a precise context of what threatens this ecosystem and how : "a default value of zero for a difficult-to-measure ecological value, as is used (explicitly or implicitly) in a number of cost-benefit analyses, is no more defensible scientifically than a default value of infinity. But this only underscores the need for evaluating ecological services in context" (Toman, 1998). To go further in this direction, we will briefly review what should not be done and what should be tried, before turning to the contexts in which the evaluation of ecosystems can have a real meaning.

### 5.1 Values of the services vs. preservation cost: looking for efficiency

Confronted to the many uncertainties and controversies that follow the economic valuation of ecosystem, an alternative is often proposed: instead of evaluating the cost of the destruction or degradation of the threatened ecosystems, which is of course the same thing as the value of the ecosystem services, why not to estimate the preservation or restoration costs?

This solution, which seems very comforting, is actually a false good idea. Apparently, the evaluator may believe that he is shifting from an assessment based on preferences too difficult to assess, in a more reliable estimate based on the cost of technical supply. In practice, it provides a measure of cost that it might be totally irrational to bear, because it can be much higher than the value of lost services. Following this fallacy, the analyst has lost the very principle of economic valuation: the search for efficiency.

There are nevertheless cases when this approach can be adequate. If the definition of objective appears out of reach of any economic analysis (and this is definitely the case in many situations), the search for efficiency should lead to achieve the goal determined on other bases at lower cost, and analysing the costs of preservation, conservation, restoration or replacement, whichever is deemed possible, then is the basis of this analysis.

Ideally, economists may wish to put their analysis in a cost-benefit framework, which is the only one to provide a measure of efficiency, but this would require being able to estimate the all the costs, market and non-market, associated with every possible situation to determine what would be the best situation. At the optimum, the marginal value of services and the marginal costs of conservation would be equal. But making all these calculations is generally not realistic.

## 5.2 Evaluating potential

Most existing evaluation relate to services actually rendered by ecosystems here and now. If the objective is to assess whether the expected profits from the destruction of these environments will be greater than that of the lost ecosystem services, one must be careful to compare two trajectories of the same nature, and not the reality of ecosystems that may have suffered other damage, with the uncertain promises of idealized projects. Since there are few ways to make developers to abandon their optimism about their projects, there is no reason to value biodiversity according to the present state of ecosystem services, but rather such they should be at the relevant time horizon, if the rules favourable to their maintenance are applied.

Since good practices in evaluation are to measure the differences between contrasting scenarios, it is of utmost importance that the values of ecosystem services taken into account were that which would be effective in a favourable scenario. The word "favourable" should not be construed as referring to a pristine wilderness that will not exist anymore, but as the result of changes in real ecosystems at a specific horizon, assuming that reasonable choices for maintaining them were made.

## 5.3 Valuing ecosystems for better decision-making

On the one hand some economists and ecologists are convinced that economic analysis is an adequate framework for improving decisions involving conservation aspects (Heal et al., 2005). On the other hand, other ecologists or policy analysts consider this task as unrealistic and misleading (see Chee, 2004; Sagoff, 2008). In a recent article, Wainger et al. (2010) were therefore legitimate to address the somewhat iconoclastic question: "Can the concept of ecosystem service be practically applied to improve natural resource management decision?".

#### 5.3.1 Decision without explicit valuation

It may appear somewhat surprising to read so many warnings about the unreliability of economic evaluations, particularly when they relate to ecosystem services, whereas, in real decision-making, namely in France, economic evaluation appears to have so little influence on the final choices. The question therefore remains somewhat theoretical, or refers to foreign contexts. It is of course complicated by the fact that agents-citizens-voters have, in this case, a biased perception of the real issues.

The position of a policymaker that is to make a choice in an area where people have wrong beliefs has in fact been studied by economists (Salanié et Treich, 2009). This case can be considered as a simple extrapolation of a situation in which there is no reliable or recognized assessment. Without going into this interesting model, we will recall the three situations identified and characterized by the authors:

- dictatorship, when the decision is made according to the sole preferences of the Prince, or of the interests he may serve,
- populism, when the decision is made according to the preferences of people, or of the best organized lobby, even though the policy-maker knows that their beliefs are wrong,
- paternalism : when decision relies on expert knowledge to serve the real interests of people, even if they do not realize the dangers they are protected in this way.

There is no need to enter in technical aspects to make understandable that these three situations do not compare favourably with the case when an explicit valuation allows the policy-maker to make choices according to the real interests of the population. It is admitted here that these interests are not easily identifiable, and much less quantifiable or economically assessable. But do not even try is probably worse.

#### 5.3.2 Valuation as a guide for decision making

Since ecosystem services are not fully captured in commercial markets or adequately quantified in terms comparable with economic services and manufactured products, they are often given too little weight in policy decisions. There is then a need, like for many assets involving public good aspects, to implement public policies for efficiency considerations. In this perspective, the evaluation of ecosystems as a means of improving information for decision-making is a recurring proposal of the groups established to consider this issue (Bingham et al., 1995, Heal et al, 2005; Chevassus-au-Louis et al., 2009 TEEB, 2009; Daily et al., 2010). A body of academic works have explored some of the technical or informational difficulties that such approaches are meeting and identified more or less realistic ways to treat them .

When reviewing a set of these works, it is readily apparent that most are pursuing the same goal: defining cost-efficient, least-cost, or efficient conservation policies (Ando et al, 1998; Murdoch et al., 2007; Polasky et al., 1999; Polasky et al., 2001; Turpie et al., 2003; Underwood et al., 2008; Wätzold et Schwerdtner, 2005; Wu et Bogess, 1999). Apparently, no

published study aim to characterize an optimal policy of conservation, with the purpose of justifying the objectives of conservation relatively to other legitimate social goals.

This latter objective, which would be the most consistent with the purpose of economic analysis, is probably unrealistic given the quality of results. And we must recognize that most studies that attempt to situate their subject in a clear theoretical framework, do not go further than declarations of intent. And one must acknowledge with the authors that the quality and incompleteness of information, notwithstanding possible conceptual difficulties, can not a priori to do better than the construction of indicators of values that can be incorporated into the economic assessment of the projects that affect biodiversity.

Several authors have nevertheless stated that it would be possible to go further, and considered that valuation might be a first step toward a "commodification" of Nature.

#### 5.4 Valuation as a prerequisite for institution building?

A body of literatures tends to consider economic valuation as a simple step toward institutional innovation, namely the creation of Payment for Ecosystem Service (PES) schemes (see Gomez-Baggethum et al, 2010) and possibly of new property rights.

#### 5.4.1 Designing policy instruments

The payment schemes for ecosystem services have mostly appeared in Latin America, namely Costa Rica (Pagiola, 2008), as a practical way to raise money for conservation. They have further sometimes be seen a universal policy instrument, up to the point that Engel et al (2005) felt the need to clarify that: "*PES is not a silver bullet that can be used to address any environmental problem, but a tool tailored to address a specific set of problems: those in which ecosystems are mismanaged because many of their benefits are externalities from the perspective of ecosystem managers.*" (Engel et al, 2005). The PES are however a rather broad category, in which can probably be classified some of the mechanisms of the Second Pillar of the CAP.

It must be clear that there is no direct link between the value of ecosystems and the prices the PES mechanisms can create as an incentive for ecosystem services preservation or enhancement. Like in the case of cost-efficient policies, the price in a PES mechanism is supposed to reflect the opportunity costs of the farmers or any other social category that become beneficiary of such a mechanism in exchange of fulfilling some ecological target or implementing some constraint in terms of ecosystem use.

Muradian et al. (2010) devote special emphasis to institutional and political economy issues, related to the PES. They consider there is a gap between the Coasean approach that dominates the design of PES in the economic literature and what can be practically implemented. According to their analysis, PES must, on the contrary, be analysed taking into account complexities related to uncertainty, distributional issues, social embeddedness, and power relations in order to understand the variety of contexts and institutional settings in which PES operate. Even if the reality of PES is usually very far from an efficient market, implementing PES schemes means to some extend designing new property rights

#### 5.4.2 Designing new property rights?

Can it be both possible and appropriate to go beyond the implementation of incentives policy instruments and to use economic valuation for defining of new property rights? This would be the standard Coasean answer to the externalities issues. And, while being more cautious with the realities, this way was proposed by some positive studies, such as Chichilnisky and Heal (1998), Heal (2000; 2003) or McCauley (2006), which explore the potential to manage biodiversity by bundling it with marketable assets like agricultural products, pharmaceutical bio-prospecting contracts or eco-tourism.

Is this way really so promising? It is of course difficult to state definitely on potentials, and the economic theory of property rights has stressed since early papers (Demsetz, 1967) that new property rights potentially emerges when there is more benefits in appropriation then transaction costs for their implementation. New property rights may even create new responsibilities and appropriate incentives but, some basic principle shape their potential. Demsetz clearly distinguishes between the goods and the related rights that "When a transaction is concluded in the marketplace, two bundles of property rights are exchanged. A bundle of rights often attaches to a physical commodity or service, but it is the value of the rights that determines the value of what is exchanged" (Demsetz, 1967). This statement remains useful today to understand that the values revealed by any market are no more (and no less) than the value of what the prevailing property right regime defines as being marketable. Faced with the complexity of ecosystems and biodiversity, it appears

Faced with the complexity and uncertainty attached to biodiversity and ecosystem functioning, it seems unlikely that a comprehensive system of rights should never be able to cover all issues. The conclusion is ineluctably that even if efficiency gains can be expected from the definition of new rights allowing to expand the social management of the benefits that humans obtain from ecosystems, some aspect of these benefits will not be supported appropriately by these mechanisms and public policies, relying on a broader and shared understanding of issues, will remain necessary.

## 5.5 Final words

In a literature and issues review, Turner et al. (2003) stated that "*In the last 30 years or so, valuation of environmental change and services has become one of the most significant and fastest evolving areas of research in environmental and ecological economics.*" Many non-specialists might have believed that this field was mostly developed after 2003. Turner et al. relate this massive interest to the motivation to build a better and more comprehensive informational base for the policy formulation and decision making process.

When related to ecosystems and biodiversity, so as when related to about anything else, valuation is not a solution or an end in itself, but firstly a conceptual and methodological framework for organizing information as a guide for decision-making. Daily et al. (2000) went further: "*It is one tool in the much larger politic of decision-making. Wielded together with financial instruments and institutional arrangements that allow individuals to capture the value of ecosystem assets, however, the process of valuation can lead to profoundly favourable effects.*"

This last point remains today questionable. In the current state of the art, is the valuation of ecosystem services able to support a relevant "internalisation" of the non-market benefits of conservation? The answer is unfortunately negative. And the explanation does not necessarily lie in some failure of the conceptual framework, but rather in the weakness of our practical information and, possibly, of the valuation techniques, which have to struggle with poorly motivated preferences to achieve price-equivalent consistent with the common monetary metric usable with competing uses. Alternative approaches to build a value concept on objective information have not really succeeded to produce a usable framework that links conceptually empirical observations with normative social objectives<sup>8</sup>.

The political will to continue developing a conceptual and methodological framework that has not established its capacity to handle the complexity of the natures-societies interaction can certainly be related to the growing evidence that we are living in a world of increasing scarcity (Baumgärtner et al., 2006). In this coming world – this world may already be the one we are living in – the situations when choice between competing uses of ecosystems might

<sup>&</sup>lt;sup>8</sup> None of the most promising proposals, such as energy, "emergy" or ecological footprint, can be clearly related with normative concept compatible with collective decision-making in other fields of human activities.

become more and more numerous. And explicit analysis methods may become a necessity to legitimate brutal choices.

Finally, the choice is not between valuing or not valuing, it is between valuing with explicit and contestable methods and valuing implicitly and referring to general principles that have in many cases been analysed as manipulated, like in some well studied situations (see Johansson-Stenman & Konow, 2010), in which the equity argument is used in a self-serving objective (known as "equity bias").

Even if the non-use values remain difficult to assess in a strict economic framework, the combination of evaluation methods with more deliberative approaches, favourishing the formation of consistent and reasoned preferences, could allow some reconciliation of instrumental and intrinsic values (Norton & Noonan. 2007). At stake here is nothing less than the design and implementation of biodiversity policies that articulate the goals of conservation with those of preservation.

To emphasize that economic evaluation should remain a means and never an end, we let the last word to D. Ehrenfeld (1988), who in the seminal book of Wilson, after a plea against the idea of putting a value on biodiversity, concluded: "*I cannot help thinking that when we finish assigning values to biological diversity, we will find that we don't have very much biological diversity left*". Ecosystem services valuation would remain meaningless if it does not help to make better practical choices and actions.

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